3.3 Vadose Zone Studies

D. G. Horton

This section summarizes the activities and results of several technical studies done at the Hanford Site in fiscal year (FY) 2004 to better understand the vadose zone sediment, vadose zone hydrology, and contamination. These studies are designed to result in new, innovative methods for cleanup and monitoring at the Hanford Site. These studies include experiments supporting in situ gaseous reduction of contaminants in the vadose zone and infiltration and recharge studies. Also, laboratory analyses were done on vadose zone sediment, bentonite well seal material, and perched water from the vadose zone to determine the reason for corrosion of relatively new well casing at Waste Management Area A-AX. Finally, development continued in 2004 to create a database of vadose zone geologic and geophysical data for use in subsurface characterization, monitoring, and modeling and for performance assessments.

Vadose zone studies are designed to result in new, innovative methods for cleanup and monitoring.

3.3.1 Investigation of Accelerated Casing Corrosion in Wells at Waste Management Area A-AX

C. F. Brown and R. J. Serne

The U.S. Department of Energy's (DOE's) Vadose Zone Project managed by CH2M HILL Hanford Group, Inc. asked Pacific Northwest National Laboratory to investigate the cause of accelerated corrosion of type 304L stainless steel casing in two *Resource Conservation and Recovery Act* (RCRA) groundwater monitoring wells located within Waste Management Area A-AX. Geochemical and selected physical characterization tests were performed on archived vadose zone sediment recovered during the installation of four RCRA monitoring wells 299-E24-19, 299-E24-20, 299-E24-22, and 299-E25-46; sidewall core samples collected during the decommissioning of corroded wells 299-E24-19 and 299-E25-46; a sample of Wyoming bentonite; and a sample of perched water collected during the recent installation of well 299-E24-33, located near the corroded wells. The locations of Waste Management Area A-AX and the sampled wells are shown on Figure 2.10-1 in Section 2.10. A photo of the corroded casing in well 299-E24-19 is shown in Figure 2.11-7 in Section 2.11. This section summarizes the casing corrosion investigation. The complete work will become available in FY 2005.

Casing corrosion occurred between 90.7 and 91.2 meters below ground surface in well 299-E24-19 and from 90.0 to 91.4 meters below ground surface in well 299-E25-46. Laboratory tests were conducted to characterize the sediment and identify water-leachable constituents. Testing consisted primarily of 1:1 sediment:water extractions, which were used to calculate the elemental concentrations of water soluble constituents in the solid and estimate in situ pore-water chemistry conditions (specifically with regard to chloride). Additionally, 8 M nitric acid extractions and x-ray diffraction analysis of the solids were used to provide a measure of the total leachable elements and any new crystalline phases that may have formed during the corrosion process.

The moisture content profile of the archived sediment samples collected during installation of the four RCRA groundwater monitoring wells correlated nicely with the lithology described in the well logs from the respective boreholes. The primary region of interest was the silt lens located between 88.6 and 91.9 meters below ground surface, which had an elevated moisture content ranging from 5.5% to 25.1%. The large degree of variability in the moisture content of the material collected from within the silt lens was a direct result of sample preservation. Unfortunately, several of the sample containers were not sealed tightly; therefore, evaporation likely occurred during the archive period, resulting in moisture content calculations that were not indicative of in situ conditions. Sediment

samples collected above and below the silt lens were comprised primarily of coarse-grained sand, with moisture contents ranging from 0.4% to 3.7%. By contrast, the sidewall core samples collected at the time of well decommissioning had elevated moisture contents ranging from 30.4% to over 50%. The elevated moisture content in the sidewall core samples correlated well with, and has been attributed to, the composition of the material collected, namely, varying percentages of bentonite and silt lens material (confirmed via x-ray diffraction analysis).

The casing in two wells at Waste Management Area A-AX is believed to have corroded due to crevice corrosion and stress corrosion cracking in a high chloride environment facilitated by perched water.

The pH profile in water extracts of the archived sediment samples ranged from a low of 7.37 in well 299-E24-20 at 86.9 meters below ground surface to a high of 7.91 at well 299-E24-19 at 88.6 meters. The measured pH in extracts from the sidewall core samples was quite different, with an average solution pH for the seven samples of 2.2. An acidic pH in this environment indicates either the presence of acid as a subsurface contaminant, or more likely, is the result of the hydrolysis of metals during the breakdown/corrosion of the well casing. Not surprisingly, a water extract of the Wyoming bentonite test sample had a solution pH of 7.93. However, it was interesting to note that the dilution corrected pore-water electrical conductivity of the pure bentonite sample (Wyoming bentonite) was nearly a factor of two greater than any of the sediment samples tested. These data suggest that the bentonite material (backfill and seal) acted as a source of salinity for vadose zone pore water in the vicinity of the stainless steel casing.

During the course of this research effort, an emphasis was placed on determining the chloride content of the sediment samples. Two primary mechanisms that lead to stainless steel corrosion/failure in chloride containing environments are (1) crevice corrosion and (2) stress corrosion cracking (Sedriks 1996). Crevice corrosion occurs as a result of dissolution of metal from within the crevice followed by hydrolysis of the dissolved metal ions (Sedriks 1996). The end result of this process is a microclimate of pH <2 within the crevice with ambient (near neutral) solution outside the crevice. Stress corrosion cracking is characterized by the failure of stressed alloys in corrosive environments. This phenomenon requires static mechanical loading (caused by forming or welding of the material or perhaps stresses caused by pounding the casing into the sediment formation) in conjunction with an aggressive environment (Lacombe et al. 1993). In the neutral pH environments typically found in the vadose zone at the Hanford Site, 100 mg/L chloride is the critical threshold concentration beyond which stainless steel experiences pitting or stress corrosion cracking problems (Sedriks 1996).

The archived sediment samples, collected during installation of the four RCRA monitoring wells, had calculated pore-water chloride concentrations ranging from 28.8 to almost 600 mg/L. However, due to sample preservation problems, the measured moisture content of samples was artificially low and resulted in an exaggeration of the true pore-water chloride concentration. To confirm this hypothesis, the mass of chloride that was water-extractable from the archived sediment samples was corrected by field moisture logging data from similar lithologic units/depths collected during the installation of well 299-E24-20. Performing this calculation yielded calculated pore-water chloride concentrations in the archived sediment samples ranging from 25.5 to 95.5 mg/L. The one exception was in an archived silt lens sample from well 299-E25-46, which had a calculated pore-water chloride concentration of 127 mg/L. Therefore, it is unlikely that any of the archived sediment samples tested could generate pore waters with sufficient chloride content, with the exception of the silt lens sample from well 299-E25-46, to initiate corrosion of the well casing.

Interpretation of the laboratory data indicated that the Wyoming bentonite test sample was capable of generating localized vadose zone pore water with chloride concentrations in excess of 700 mg/L. The sidewall core samples from well 299-E24-19, which were comprised of a mixture of bentonite and silt lens material, had an average pore-water chloride concentration of 376 mg/L. The sidewall core samples collected from well 299-E25-46 had calculated pore-water chloride concentrations ranging from 1,200 to more than 10,000 mg/L. Therefore, the Wyoming bentonite test sample, as well as all of the sidewall core samples

tested, were capable of generating pore waters with sufficient chloride concentrations to cause corrosion of the stainless steel well casing.

Analysis of the sidewall core samples yielded a clear relationship between chloride concentration and well casing corrosion. The sidewall core samples containing the greatest amount of chloride, 3,000 μg/g of sediment, came from the well that experienced the longest length of casing failure (1.38 meters in well 299-E25-46). All of the sidewall core samples from both of the decommissioned wells contained more chloride than the Wyoming bentonite test material. However, the chloride constituted a trace constituent of the sample and its presence in varying concentrations could be a result of the source of the bentonite and/or processing of the material by the vendor.

The sidewall core samples from well 299-E25-46 were very important to this study because they provided a lateral glimpse of the zone of corrosion adjacent to the casing. The concentration of chloride in the three samples was greatest in the sample collected closest to the degraded well casing, and decreased with increasing distance from the corroded casing. This finding implies that the annular seal material contained the source of chloride in the sidewall core samples. Testing is currently underway to determine if the bentonite material, or a contaminant contained therein, or a process performed during the installation of the now failed wells could has acted as the source of chloride. Regardless of the source of chloride, it is believed that the advanced well casing corrosion found at wells 299-E24-19 and 299-E25-46 was caused by chloride facilitated crevice corrosion and stress corrosion cracking. Furthermore, it is possible that the casing in wells 299-E24-19 and 299-E25-46 was damaged at the time of installation, which would have accelerated the stress corrosion cracking process. Additionally, the silts lens, located between 88.6 and 91.9 meters below ground surface, likely exacerbated this process by providing a continual source of moisture in contact with the chloride source, which generated localized pore waters with high chloride concentrations.

As a result of this study, cement grout was recently used instead of bentonite to seal the annulus of two new wells in zones with high moisture content.

3.3.2 Gaseous Reduction and Reoxidation Characteristics of Hanford Formation Sediment

E. C. Thornton, L. Zhong, and M. Oostrom

In situ gaseous treatment of sediment with diluted hydrogen sulfide could provide a way to immobilize various contaminants in the vadose zone. Reduction of metals and selected radionuclides, such as chromium, technetium, and uranium, is a possible approach to control contaminant transport through the unsaturated zone and, thus, protect the underlying aquifers. Laboratory testing has shown that sediment treated with diluted hydrogen sulfide in nitrogen reduces ferric oxides to ferrous oxides and ferrous sulfide. Ferrous sulfide, in particular, is a strong reducing agent that provides a long-term potential for maintaining a reduced environment. Therefore, treated sediment could be used to establish a permeable reactive barrier in the vadose zone that would reduce and precipitate contaminants entering the treated zone via infiltration of contaminated solutions from surface waste sites.

Laboratory studies provide a basis to predict gas consumption and treatment time for in situ gaseous treatment of contaminants in the vadose zone.

Recent laboratory research and development activities related to in situ gaseous treatment have focused on Hanford formation sediment and re-oxidation characteristics. This has provided information regarding consumption of hydrogen sulfide during treatment, which is needed to support field design and barrier installation activities. Since re-oxidation of the treated sediment will eventually occur, sediment re-oxidation testing activities have also been undertaken in the laboratory to provide a basis for estimating the lifetime of a vadose zone permeable reactive barrier.

3.3.2.1 Gaseous Treatment of Hanford Formation Sediment

Laboratory column tests were completed for the purpose of measuring the consumption of hydrogen sulfide during interaction with Hanford formation sediment. Conventional columns, measuring 30.4 centimeters long by 2.6 centimeters in diameter, were packed with Hanford formation sediment. Each column was treated with diluted hydrogen sulfide (200 ppm H_2S in N_2) until hydrogen sulfide breakthrough was complete (i.e., greater than 180 ppm H_2S detected in the column effluent). Gas flow rates were varied in four column tests from 200 to 500 millimeters per minute. Figure 3.3-1 illustrates that the breakthrough time increased as the flow rate was reduced.

Two tests were also conducted in a larger column measuring 158 centimeters long by 20.3 centimeters in diameter. The purpose of the larger column tests was to provide breakthrough data in a system where the linear velocity of gas flow is comparable to that expected in the vadose zone during in situ gaseous treatment. Nine sampling ports were located along the length of the larger column at a spacing of 15.2 centimeters. This permitted monitoring of the hydrogen sulfide reaction front along the length of the column as well as measuring breakthrough at the column outlet. Flow rates of the gas mixture were 1,800 and 4,000 milliliters per minute in the two larger column tests.

The relationship between the velocity of the reaction front and pore velocity for the conventional and large column tests is presented in Figure 3.3-2. The reaction front velocity is the time required for the column effluent concentration of hydrogen sulfide to attain 50% of the inlet concentration (i.e., effluent concentration is equal to 100 parts per million [ppm] hydrogen sulfide). The figure shows that the relationship is nonlinear, indicating that reaction kinetics are an important aspect of gaseous treatment. The empirical relationship presented, however, provides a basis for predicting gas consumption and treatment time requirements since pore velocity can be calculated given the geometry of the treatment cell and injection flow rates.

3.3.2.2 Reoxidation Behavior of Treated Sediment

Laboratory column tests also were done to evaluate the re-oxidation of gas treated sediment. This information is needed to determine the reductive capacity of treated sediment, which provides a basis for assessing the potential effectiveness of the permeable reactive barrier and estimating the lifetime of the barrier with respect to re-oxidation.

Four tests were conducted using the smaller columns mentioned above (30.4 centimeters long by 2.6 centimeters in diameter). The columns were packed with Hanford formation sediment and the sediment treated with 200 ppm H_2S/N_2 gas mixture at a flow rate of ~32 hours for each of the tests. The treated soil was then re-oxidized using deionized water equilibrated with a 21.5% oxygen in nitrogen gaseous mixture. The water was pumped through the treated soil at a different flow rate in each test over the range of 0.096 to 0.71 millimeter per minute and re-oxidation monitored by measuring the oxygen concentration of the column effluent. Re-oxidation was fast initially (i.e., no oxygen was observed in the effluents) and oxygen breakthrough was observed to begin in each test after ~20 column pore volumes were pumped through the column. The breakthrough curves were initially steep, but gradually flattened out with time and complete oxidation could not be achieved. The re-oxidation phase of the experiments was terminated when the oxygen concentration of the effluent was 80% or more of the concentration of the influent.

The relationship between the number of pore volumes of water pumped through the column to achieve initial oxygen breakthrough versus the linear pore velocity of water was determined using a power function (R^2 = 0.9948). The resulting curve is almost linear and indicates increased reductive capacity at lower flow rates (Figure 3.3-3). These results suggest that a higher percentage of the total reductive capacity would be utilized in maintaining an anoxic environment in a vadose zone barrier, since flow rates through the barrier would probably be much lower than employed in these tests. Application of these test results can

provide a basis for estimating oxygen breakthrough characteristics of a vadose zone barrier at the flow rates expected to occur within a barrier.

3.3.2.3 Modeling Activities Conducted in Support of the In Situ Gaseous Treatment Approach

Periodic rejuvenation of gases in an in situ reactive barrier may be necessary to maintain a permeable reactive barrier in the vadose zone.

Calculations were done to estimate barrier lifetime. These calculations assume that the lifetime of the barrier is related to re-oxidation of iron in the treated zone and is an equilibrium process. Re-oxidation of iron in the barrier is thus related to oxygen flux into the barrier. This flux consists of two components – diffusion of oxygen from the surface to the treated vadose zone interval through the unsaturated portion of the sediment pore space and by vertical downwards flow of aerated pore water.

Initial calculations suggested that re-oxidation would require a very long period of time (hundreds to several thousands of years). A recent reassessment of these calculations indicates that barrier lifetime may be on the order of only several years. The primary reason for this discrepancy appears to be related to estimation of the oxygen diffusivity coefficient. The value of this parameter is fairly well known for oxygen gas (0.21 cm^2/s), but appears to be poorly known in partially saturated environments. A literature review is being conducted to further clarify this issue. However, a conservative approach is being taken at this time and a relatively short barrier lifetime is being assumed. Thus, periodic rejuvenation of the barrier through injection of H_2S/N_2 mixtures may be a necessary part of maintaining a vadose zone permeable reactive barrier.

3.3.3 Recharge Estimates Using Chloride Mass Balance at the Hanford Site

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Lysimeters measure drainage of liquids such as precipitation through soil.

The results of a multi-year study of recharge at the Hanford Site using chloride mass balance became available in 2004. The chloride-mass-balance method has been used extensively to estimate recharge in arid and semi-arid environments. This method was tested at the Hanford Site against 26 years of drainage from a 7.6-meter-deep lysimeter at a simulated waste-burial ground where removal of vegetation has increased recharge rates. This section describes those experiments.

In the chloride mass balance method, measurements of chloride in pore water are used to estimate the recharge rate when both precipitation and chloride inputs are known. The chloride mass balance for a soil profile at steady state can be written as:

$$P\,(Cl_p) = R\,(Cl_s)$$

where P = average annual precipitation (millimeters per year)

Cl_p = average chloride input from all sources, including wet and dry fallout (mg/L)

Cl_s = average chloride concentration of pore water below the root zone (mg/L)

R = average annual recharge rate (millimeters per year)

Key assumptions are (1) steady influx of water and chloride; (2) steady, vertical efflux of chloride below the root zone; (3) no soil sources or sinks for chloride; and (4) piston flow of chloride such that point measurements of solute concentrations can be used to represent a true spatial average of the soil chloride flux. With proper assumptions about average annual precipitation and chloride inputs, the only direct measurement required is the volume-averaged chloride concentration in the pore water.

Chloride mass balance has been applied for years to estimate recharge. However, in situations where there has been land-use change and recharge has increased, there have been few attempts to compare chloride mass balance with any independent or direct estimates of recharge. These experiments tested the chloride mass balance method for predicting recharge in a semi-arid climate setting, where land use has changed (i.e., soil disturbed and vegetation removed) and where the chloride flux could be quantitatively checked with lysimetry. Lysimetry is a method that can be used to directly measure the percolation of water through soils and determine both the flux rate and soluble constituents removed in the drainage (Gee and Hillel 1988; Gee et al. 2003).

For the past 25 years, the precipitation at the Hanford Meteorological Station has averaged 180 millimeters per year, about two thirds of this amount coming in winter months. For undisturbed sites, with shrub-steppe vegetation, actual evaporation is approximately equal to annual precipitation, so little drainage is expected. Chloride measurements made at the Hanford Site in areas of undisturbed shrub-steppe vegetation growing on coarse soil have shown significant bulges of high chloride (>100 mg/L) at shallow depths with corresponding recharge rates estimated to be much less than 1 millimeter per year (Prych 1995; Murphy et al. 1996; PNNL-13033). In contrast, for disturbed sites with little or no vegetation and coarse soil, actual evaporation can be less than two-thirds of the annual precipitation (Gee et al. 1992) resulting in drainage rates that exceed 50 millimeters per year. The corresponding chloride distributions are expected to be low at the disturbed sites containing coarse soils.

Chloride concentrations in drainage waters from a test lysimeter were in good agreement with those required in chloride mass balance estimates of lysimeter drainage rates.

Precipitation at the lysimeter test site was previously found to be ~6% more than at the Hanford Meteorological Station (PNL-6403), so precipitation was estimated to be 190 millimeters per year for the past 26 years at the lysimeter site. Chloride input from wet and dry fallout has been studied extensively by Murphy et al. (1996) and found to range from 0.22 to 0.23 mg/L. For the purposes of this study, 0.225 mg/L was selected for the chloride input at the lysimeter site.

Soil samples were taken at a simulated waste burial ground ~5 kilometers north of Richland from two sand-filled, 7.6-meter-deep lysimeters that were kept vegetation free for the past 26 years. The lysimeter soil is classified as 1% gravel, 95% sand, 3% silt, and 1% clay using the United States Department of Agriculture system (PNL-3304). Chloride analysis performed on samples of lysimeter soils taken at the time of construction (1978) showed the initial chloride concentration of the pore water in the lysimeter soil was 88 mg/L.

Lysimeter drainage was measured in two ways: (1) from the inception of drainage in 1981 until April 2000, drainage was measured by periodically collecting water in tared containers and weighing the containers in the laboratory and (2) in April 2000, the drainage collection was switched to an automatic tipping spoon (Pronamic Ltd.) rain gauge. During the past 20 years, the lysimeter drainage from the bare soil has averaged 34% of the total precipitation or ~55% of the winter precipitation.

In December 1996, soil cores were taken from the lysimeter at ten depths, from the surface to a depth of 7.1 meters. Each soil sample was analyzed for water content and then a 1:1 (solution:solid) extract was prepared and analyzed for chloride. Similarly, samples were taken in March 1998 and then again in September 2002. All samples were analyzed using similar equipment.

Table 3.3-1 shows the chloride concentrations for the soil samples and drainage water collected at three dates. (Note that fewer samples were taken in March 1998 than were taken at the other two sampling times.) Recharge estimates from soil cores are presented in Table 3.3-2 along with recharge estimates based on chloride in the drainage water. These values are compared to the measured drainage rate. It is apparent that the soil-core estimates underestimate the recharge by a factor of 5 or more for samples taken in 1996 and 2002 and by a factor of 2 or more for samples taken in 1998. The explanation for the differences in the 1998 data and the other two sampling events is that the chloride in the March 1998 samples

was measured at field moisture, and the samples were not oven dried. The other sample sets were oven dried, which is the typical protocol for chloride mass balance analysis.

There are a number of uncertainties associated with the chloride mass balance method, one of which is the total chloride input. Chloride input was not measured directly but was derived from estimates made by Murphy et al. 1996. Table 3.3-2 shows that if the chloride input is increased 22%, from 0.225 to 0.275 mg/L, agreement is achieved between the 26-year average lysimeter drainage rate and that estimated from the chloride mass balance method using the drainage-water chloride as an estimate of the true pore-water chloride. While this modification can explain the difference between the chloride mass balance estimate using the drain water, it does not explain the large underestimation of recharge using the soil water extracts, because (1) it would take an unreasonably high chloride input value of above 1 mg/L and (2) if such a high chloride input value was used, it would make the drain water correspondingly to high, i.e., above 3 mg/L.

To investigate the discrepancy between the soil chloride concentrations and the observed drainage flux, the transport was modeled using a mobile/immobile water transport approach (van Genuchten and Wagenet 1989). Simulations were conducted with CXTFIT (Toride et al. 1995) under conditions of steady drainage (65 millimeters per year) over the first 16 years of drainage. Previous laboratory data (PNL-3304) suggested that the ratio of mobile water to total water content was ~0.65. Several simulations using this ratio failed to produce any similarities to the observed drainage and soil sample concentrations. The first simulation, which assumed a very slow exchange coefficient and a ratio of mobile water to total water of 0.65, was able to match the drainage water concentration, but soil water chloride (resident) concentrations were much higher than observed. Decreasing the exchange coefficient in subsequent simulations produced higher chloride concentrations in the drainage water than were observed, and also resulted in chloride concentrations varying down the length of the lysimeter at the end of the 16-year simulation period. Additionally, the chloride concentrations in the drainage water changed significantly with time; as opposed to the observed concentrations that are essentially unchanging in time between 1996 and 2002. Decreasing the mobile to total water content ratio produced much higher soil water chloride concentrations than were observed.

Slow mineral dissolution may explain elevated chloride concentrations in soil pore water compared to chloride concentrations in lysimeter drainage.

Only when the ratio of mobile to total water content was increased to 0.9 and an extremely slow exchange between the mobile and immobile phase (α=1 x 10^{-5} $years^{-1}$) was chosen could simulated soil water and drainage water concentrations be well matched to those observed in the 1:1 dilutions and drainage water. Under this simulation, the chloride in the immobile phase is essentially unavailable for transport. These simulation results produced fairly uniform pore-water and drainage water chloride concentrations that changed only very slowly over time.

The parameters used above suggest that a small portion of the chloride in pore water is inaccessible to the recharge waters. The lack of significant temporal change in either the soil water chloride or drainage water, combined with the fairly uniform vertical distribution of soil water chloride suggests that the majority of pore water is mobile yet disconnected from a small mass of soil water chloride.

The small mass of immobile water is consistent with an incomplete flushing of the vadose zone by infiltration and is consistent with that seen by Jolly et al. (1989) following a change (increase) in recharge. The nature of the incomplete flushing could be either microscopic, with remnants of chloride remaining in dead end pores that are ubiquitous, or as large unflushed areas resulting from large-scale preferential flow. As will be discussed below, large-scale preferential flow is very unlikely to be occurring in the lysimeter. Rather, the findings of elevated soil water chloride, as compared to the drainage water, is consistent with widely dispersed storage of small amounts of residual chloride originally present when the soils were emplaced in the lysimeters.

Preferential flow probably is not an explanation of the high chloride concentrations found in the soil samples. The chloride concentrations of side port samples taken in 1996 were nearly the same as those found in borehole samples taken from the center of the lysimeter in 2002. If there were wall effects or other preferential flow occurring, it should have shown up as marked differences in the two tests.

Mineral dissolution is a possible mechanism for some of the high chloride numbers. PNNL-13033 reported chloride concentrations in Hanford formation minerals ranging from 100 to 230 ppm. If 0.1% of the chloride leached from minerals, this would account for a soil-solution concentration ranging from 0.10 to 0.23 mg/L. Such levels of chloride could possibly have leached from the samples, particularly after they had been dried and subsequently wetted with deionized water. The fact that the samples in 1998 were not oven dried and gave the best results, suggest that oven drying may release chloride from the sediments tested. While this hypothesis is not exhaustively tested, mineral dissolution may contribute to the high chloride values obtained with the 1:1 extracts. Additional chloride contamination from outside sources at the very low levels of chloride is also possible.

These observations raise concerns about using soil pore-water sampling in the chloride mass balance method for recharge rates much above a few millimeters per year at the Hanford Site and possibly at other locations where slow mineral dissolution or other sources of chloride release can confound the low concentration values of the soil chloride. It is further recommended that minimum soil-water dilutions be used when measuring chloride concentrations in soils to reduce the impacts of analytical errors and possible dissolution. Dilution errors may be reduced by using less than 1:1 extract ratios or by centrifuging field-moist samples (when the sample is wet enough), or by obtaining pore water directly in the field using solution sampling or wick lysimetry (Gee et al. 2003). This study indicates that at soil pore water concentrations below a few milligrams per liter, chloride sources other than from precipitation and fallout may contribute to errors in estimating recharge using the chloride mass balance method.

In summary, chloride concentrations found in drainage waters from the test lysimeter, were in good agreement with those required in chloride mass balance estimates to accurately predict lysimeter drainage rates. In contrast, the soil pore-water chloride was always elevated with respect to the drainage water. Slow dissolution of mineral chloride is a possible explanation for the observed elevated soil pore-water chloride and consistent with the 0.9 mobile (0.1 immobile) chloride transport analysis.

3.3.4 Recharge Predictions at Hanford Waste Sites

G. W. Gee, J. M. Keller, and A. L. Ward

Lysimeter studies were used to evaluate the drainage at some Hanford waste sites.

Studies of drainage, which leads to recharge in desert soil like that found at the Hanford Site, have been ongoing for more than two decades (PNL-6403; Gee and Hillel 1988; Gee et al. 1992, 1994; PNNL-13033). These studies have dispelled the myth that hot, dry desert conditions prevent deep drainage. In reality, significant drainage can occur under desert conditions, even when potential evaporation rates greatly exceed precipitation, particularly at locations where soil has been disturbed and vegetation removed. An analysis of drainage under conditions where surfaces are primarily coarse-textured and barren became available in FY 2004. That analysis compared two methods for estimating drainage rates from bare soil surfaces at the Hanford Site, namely Darcy's Law (or Richard's equation) and an empirical model approach.

Three sites with extensive drainage records were selected for the study: 300 North Lysimeter Site, Field Lysimeter Test Facility, and Solid Waste Landfill.

The 300 North Lysimeter Site, located ~6 kilometers due north of the 300 Area, contains two large drainage lysimeters (2.7-meter diameter, 7.6-meter deep) that have been monitored

periodically for the past 26 years. Soil taken from the 300 North lysimeters were analyzed (PNL-6488) for unsaturated hydraulic conductivity using (1) the particle-size method of Arya and Paris (1981), (2) the steady-state column method of Klute and Dirksen (1986), (3) the Guelph Permeameter method of Reynolds and Elrick (1985), and (4) the instantaneous profile method of Watson (1966), where water contents were measured by neutron logging and pressure profiles measured with a nest of tensiometers. More recently, Gee and Ward (2002) reported values of the unsaturated conductivity of the 300 North lysimeter soil using the ultracentrifuge method of Nimmo et al. (1994). The soil-water pressure profile continues to be monitored in both lysimeters using tensiometers (Sisson et al. 2002) and soil-water content is monitored by capacitance methods and gravimetric sampling. In addition, field tests were run in the lysimeters to measure the dependence of unsaturated hydraulic conductivity on water content and soil-water pressure.

A soil texture model was developed to predict recharge in a range of soil types at the Hanford Site.

The Field Lysimeter Test Facility, located adjacent to the Hanford Meteorological Station near 200 West Area, contains a number of lysimeters designed to measure water balance using a wide range of surface cover materials, from coarse gravels to silt loam soil. Four of the lysimeters were used to measure drainage of coarse sediment under natural precipitation conditions.

The Solid Waste Landfill, located ~4 kilometers southeast of the 200 East Area, is instrumented with a 6.5-meter deep, basin (pan-type) lysimeter with a capture area of 85 square meters (HNF-7173). The lysimeter was placed at the bottom of the landfill trench and drainage has been collected from this site since July 1996 (HNF-7173).

The texture of the surface soils was determined using both dry and wet sieving followed by hydrometer analysis (Gee and Bauder 1986) for all three sites. Table 3.3-3 lists the lysimeter facilities used for the analysis and the soil and key textural characteristics for each of the lysimeters reported. Precipitation was measured at the Hanford Meteorological Station.

3.3.4.1 Drainage Estimation Methods

Deep drainage at waste burial sites on the Hanford Site is best analyzed by assessing the complete water balance of the surface soil. A simple water-balance model was developed for surface soil at the Hanford waste sites. The model assumes the following conditions:

1. Winter precipitation dominates the net infiltration process.
2. Water runoff and run-on volumes are negligible because most waste sites are relatively flat or bermed.
3. Annual water storage changes are negligible.
4. Soil texture (e.g., particle-size distribution) controls the amount of water retained in the surface and influences the overall evaporation rate.
5. The soil surface remains unvegetated (upward water movement is by evaporation only).
6. Water storage in the bare soil is largely confined to the top meter of soil.

Based on the above assumptions, the surface water balance can be written as:

$$D = (P1 + P2) - (E1 + E2)$$

where P1 = the winter (November through March) precipitation

P2 = non-winter (April through October) precipitation

E1 = the winter evaporation, and E2 is the non-winter evaporation.

Combining terms leads to the following expression:

$$D = P1 - Ef \quad [4]$$

where $Ef = (E1 + E2) - P2$ is an evaporation factor dependent on soil texture and precipitation.

The equation seems simplistic but does estimate evaporation. The equation is empirical and only works for conditions where there are no plants, hence only evaporation and not evapotranspiration is considered. The equation is designed to estimate evaporation rates from bare surfaces, such as tank farms and other waste sites, that have been kept free of vegetation to minimize biointrusion and contaminant uptake by plants.

Based on an analysis of lysimeter records (Gee et al. 1992; PNNL-13033; PNNL-14744) and subsequent measurements made at the lysimeter sites through April 2004, the impact of precipitation and soil texture on drainage was determined for a variety of surface conditions ranging from clean, washed gravels to fine-textured silt loam soil.

Relationships between the evaporation factor (Ef) and grain-size were evaluated using drainage data from the lysimeters listed in Table 3.3-3 with the exception of the Solid Waste Landfill, which was used to test the texture model. The evaporation factor (Ef) was calculated for each of the four lysimeters using the average lysimeter drainage data from January 1, 1995, to April 1, 2004, and the average winter precipitation. Three different analyses of the relationship between Ef and grain-size were performed, including (1) Ef versus percent fines, as was done by Gee and Ward (2002); (2) Ef versus percent fines multiplied by the soil's D10 value (the diameter of sieve openings that will retain 10% of the soil); and (3) a multivariate analysis using percent fines, geometric mean particle diameter, d_g, and geometric standard deviation, σ_g, as the independent variables.

Darcy's Law Estimates of Drainage. Table 3.3-4 shows Darcy's Law drainage estimates from unsaturated hydraulic conductivity derived from the five different methods and the measured drainage for the 300 North lysimeter sandy soil. Field-measured soil-water pressures confirm that unit gradient conditions exist in the 300 North lysimeter (PNL-6403; Sisson et al. 2002) so use of the unsaturated conductivity value at field water content and soil-water pressures is justified. The five methods used to generate the unsaturated conductivity functions resulted in differences of more than three orders of magnitude. Compared to lysimeter (direct) measurements, the instantaneous profile method of Watson (1966) provided the best estimate of drainage, while the Guelph Permeameter provided the worst estimate.

The texture model predicted drainage for a bare sand site with a precision of better than 15%. The model over-predicted drainage by ~30% at a vegetated site.

Texture Models. The types of soil used to develop the Ef functions range from fine silt loam to coarse sandy gravel. Three of the soil samples contain similar percent fines (fraction less than 50 µm), yet are markedly different when comparing the total grain-size distribution. For example, d_g for the sand is 0.506 mm, which is appreciably smaller than the d_g for the sandy gravel at 7.529 mm (Table 3.3-3).

The cumulative drainage for the 300 North and Field Lysimeter Test Facility lysimeters used for the model calibration and the cumulative drainage for the Solid Waste Landfill is shown in Figure 3.3-4. Table 3.3-5 shows the average winter precipitation, as measured at the Hanford Meteorological Station, the average drainage and the calculated evaporation factor (Ef) for the five lysimeters during the time period used for model calibration. In addition, average winter precipitation, average drainage, and the calculated Ef for the sand lysimeter are shown from January 1982 to March 1993. The similarity of the calculated Ef for both the early and later time sand data sets illustrates the stability of Ef with differing precipitation conditions.

Figure 3.3-5 shows the relationships between Ef and two separate soil textural descriptors, with the developed Ef functions presented in Table 3.3-6. The relationship between Ef and percent fines used by Gee and Ward (2002) does not illustrate the difference in observed drainage from the sand and sandy gravel soil. While both types of soil possess the same fraction of fines, the presence of coarser soil particles in the sandy gravel decreases its overall storage capacity. To correct for the presence of a coarser soil fraction, two approaches were explored: (1) multiply the percent fines by the soil's D10 value and (2) perform a multivariate analysis using percent fines, d_g, and σ_g as the independent variables. From Figure 3.3-5, use of D10 in combination with percent fines allows the sandy gravel soil to

be better represented. This improves the Ef relationship, yet retains the influence of the fine soil fraction in the overall evaporation rate. From Table 3.3-6, the use of multivariate analysis results in a r^2 value of unity due to the degrees of freedom of the regression being equal to zero. While this essentially renders statistical evaluation of the regression model meaningless, the model is included in the overall analysis to demonstrate the potential of using different statistical models to estimate Ef values. It is expected that as data from additional sites become available the regression models will be strengthened by their addition to the calibration data set. Future Ef regression models should include analysis such as the Akaike Information Criterion (Sakamoto et al. 1986; Yu et al. 1997) to identify when the addition of independent variables is offset by a significant reduction in regression error, while still retaining enough degrees of freedom to evaluate the regression model.

Results from the texture model using the two separate Ef functions for the 300 North site sand drainage during a 12-year period were not included in the calibration data set and for the independent drainage data from the Solid Waste Landfill are presented in Table 3.3-6. For the 300 North sand, drainage was accurately predicted using the multivariate Ef function and, to a lesser degree, from the percent fines and D10 Ef function. Comparison of the developed Ef functions versus the independent drainage data from the Solid Waste Landfill shows that for both relationships, the model under predicts the observed Ef of 70 millimeters per year, resulting in an over-predicted drainage. The Ef function developed using multivariate analysis provided the best estimate of Solid Waste Landfill drainage, calculating a drainage of 66 millimeters per year compared to a measured drainage rate of 51 millimeters per year. The Ef function using the percent fines times D10 descriptor performed similar to the multivariate analysis, estimating a drainage of 70 millimeters per year.

No attempt was made to keep the surface of the Solid Waste Landfill vegetation free, resulting in the potential for water uptake by plants. Indian wild rice (*oryzopsis hymenoides*) has invaded the site and while sparse, it is a perennial growth form that has a relatively deep rooting depth and most likely has contributed to the lower drainage rates observed at this site. Because the Ef functions were developed using drainage data from non-vegetated soil, the presence of vegetation would result in the under prediction of Ef as was seen in the results. Figure 3.3-6 shows the Solid Waste Landfill Ef from 1996 through 2004 calculated from measured drainage and precipitation. A general increase in Ef with time is observed, correlating with a decrease in drainage. The trend is likely because drainage will continue to decrease with further establishment of vegetation at the Solid Waste Landfill. These data illustrate the significance of vegetation in influencing water loss rates from surface soils. Where surfaces remain barren, such as at radioactive waste sites at the Hanford Site, the texture model is expected to provide reasonable predictions of drainage. The texture model was designed to be used at waste sites where the percent fines range from a few to more than 60% (i.e., coarse gravel to silt loam surfaces).

Not presented here are data from lysimeters containing coarse rock fragments or gravel with little or no fines. Open, coarse sediment with no fines are conditions at the Hanford Site that allow for significant advective heating during late spring through fall because of high thermal loading of the soil surface (Gee et al. 1997; Ward and Gee 2000; PNNL-13143).

There has been some effort over the past 10 years to stabilize the surface of Hanford Site waste sites for worker protection against radioactivity and for ease of access. The surfaces of a number of the tank farms have been recovered with commercial "road-base" material. While no exact specifications are available for the size distribution of this gravelly material, the major specification is that it contains what is called 3/4 minus material, meaning that the majority of the material passes through a 1.9-centimeter square sieve. Because retrieving material from within the tank farms is difficult, some road base material was collected from adjacent to the Field Lysimeter Test Facility; the texture analysis from this material was used to predict the average annual drainage rates for such materials that might exist over current waste sites at the Hanford Site. Table 3.3-7 shows the size-distribution statistics

and expected drainage rates for a bare soil with 1.9-centimeter road-base and two types of synthetic soil consisting of road-base material with the addition of 10-percent fines and 20-percent fines. Predictions indicate that drainage would be substantial at all sites where road-base surfaces now exist over Hanford Site waste sites. However, modest increases in the percentage of fines significantly reduce the drainage. An increase in percent fines from 4% to 14% reduces the predicted drainage by as much as a factor of 4. An additional increase in percent fines to 24% further reduces the predicted drainage to less than 20 millimeters per year. Calculations of drainage from the synthetic soil assume that the thickness of the synthetic materials is at least 1 meter. As a first approximation, if the synthetic layer is less than 1 meter, the storage would be altered in proportion to the layer thickness; storage and, hence, drainage are functions of layer thickness, which is implicit in the empirical models presented here.

The two empirical texture models have obvious limitations. These include restriction to bare surfaces and assumptions of local climate conditions and soil layer thicknesses. In spite of these limitations, the models do surprisingly well in predicting drainage from bare waste sites. Where more site data are available, more sophisticated models may be justified.

Estimates of drainage rates from Darcy's Law using hydraulic property data may have large uncertainties, unless those estimates are tempered by calibration with actual field drainage data. Direct measures of drainage either at waste sites of concern or adjacent to them but with similar surface soil conditions will be useful in calibrating both simple models, such as those described here, or more complex models that require well-defined hydraulic properties. Whereas the texture models developed here are specific to the Hanford Site, the general approach may be applicable to other sites where similar conditions exist (i.e., bare soil surfaces where drainage is dominated by winter precipitation events).

Recharge studies such as this have important applicability to numerical models of vadose zone moisture and groundwater. The studies provide necessary input values for some models that predict fate and transport of contaminants in the subsurface. They also supply information to optimize surface cover design for contaminant remediation.

A new computer tool is being developed to provide Hanford researchers a user friendly system to archive, retrieve, and interpret borehole geologic data.

3.3.5 Borehole Geologic Information System

G. V. Last

DOE's Groundwater Remediation Project managed by Fluor Hanford, Inc. is developing an integrated borehole geologic data management and interpretation system to maximize the value and ease of use of geologic data. Borehole data are the cornerstone of subsurface characterization, monitoring, and performance assessment projects. These data often take great effort and expense to generate, yet, historically they have been managed in an ad hoc fashion, using a wide variety of formats (generally non-digital) and scattered across individual project records. The types of data included in the Hanford Borehole Geologic Information system include geologists and drillers logs, particle size distributions, calcium carbonate and moisture contents, geophysical logs, mineralogy, bulk rock chemical composition, and geochronology information.

The Hanford Borehole Geologic Information System (HBGIS) is a secure online web application supported by Microsoft SQL Server as a back end database (PNNL-SA-43003). HBGIS is designed to support the Hanford Site community with a user friendly graphical interface that provides a comprehensive information management system for archival, retrieval, and interpretation of data. As of September 2004, over 130 boreholes had been incorporated into this system. The HBGIS's unique feature is its ability to connect directly to different databases to obtain relevant borehole information rather than storing duplicate data available in the other Hanford Site databases. Existing databases currently accessed by the HBGIS include the Hanford Well Information System, the Hanford Environmental

Information System, the Virtual Library, and the Pacific Northwest National Laboratory geophysical log library. Links to other existing databases including the DOE, Office of Legacy Management's (formerly the Grand Junction Project Office) geophysical logging database may also be possible. The HBGIS data transformation option allows exporting data into graphical data processing software such as LogPlot and SoilVision.

For information about and access to the HBGIS, contact the Fluor Hanford, Inc., Groundwater Remediation Project, in Richland, Washington.

In parallel with the development of the HBGIS for accessing raw borehole geologic data, an apriori database of interpreted geologic contacts was assembled. Of the ~570 boreholes drilled within the 200 Areas, major geologic contact information was assembled for ~475, or 83%, of these boreholes (WMP-22817). Geologic contact information is used for geohydrologic and performance assessment modeling of the subsurface. Therefore, it is imperative to use a consistent set of the most accurate contact picks available for input into these models. As the raw data become more available through the HBGIS and as interpretations of these data are made, this geologic contacts database will continue to evolve.

During FY 2004, data packages providing vadose zone and geographic information were compiled. This data will be used for performance assessments.

3.3.6 Data Packages for the 2004 Composite Analysis and 2005 Integrated Disposal Facility Performance Assessment

G. V. Last and S. P. Reidel

The DOE Groundwater Remediation Project managed by Fluor Hanford, Inc. developed two vadose zone related data packages to support performance of the 2004 Composite Analysis. The vadose zone data package (PNNL-14702) describes the geologic framework, physical, hydrologic, and contaminant transport properties of the geologic materials and deep drainage (i.e., recharge) estimates. Much of the data and interpreted information were extracted from existing documents and databases.

The *Geographic and Operational Site Parameters List (GOSPL) Data Package for the 2004 Composite Analysis* (PNNL-14725) describes a number of key input parameters needed to simulate the performance of over 1,000 waste sites included in the 2004 Composite Analysis. Parameters described in the data package include information on individual waste sites (e.g., size, coordinates, type of site, operational dates), effluent types, release models, hydrostratigraphy, remedial actions, and infiltration assumptions. The data package describes the data fields, including the source(s) of data, and provides the resulting inputs assembled to facilitate the generation of keyword input files containing general information on each waste site, its operational/disposal history, and its environmental settings (past, current, and future).

A geologic data package (PNNL-14586) supporting the 2005 Integrated Disposal Facility performance assessment was released in March 2004. That data package compiles existing geologic data from the Integrated Disposal Facility area from both surface and subsurface geologic sources. The data quality and uncertainties in the data are discussed. Several cross sections, a fence diagram, and isopach maps are included in the report. The seismicity of the area and the results of hydrologic testing also are included.

Table 3.3-1. Chloride Concentrations (mg/L) Found in Soil and Drain Water Taken from the Lysimeter at Selected Times

Soil Sample Depth (m)	December 1996	March 1998	September 2002
0.0–0.5	7.0	--	--
0.5–1.0	5.0	--	--
1.0–1.5	3.2	1.5	6.1
1.5–2.0	4.8	1.6	5.6
2.0–2.5	5.8	--	3.9
3.0–3.5	6.7	--	6.7
4.0–4.5	5.7	1.5	4.6
5.5–6.0	7.8	1.6	4.9
6.5–7.0	7.4	--	4.7
7.0–7.5	4.5	--	5.2
Drain water	1.2	0.8	0.9

Table 3.3-2. Comparison of Measured and Estimated Drainage Rate (mm/yr) from Chloride Mass Balance Recharge Estimates Using Drainage Water and 1:1 Extract (pore water) Data Sets[a]

	Drainage rate (mm/yr)			
	December 1996	March 1998	September 2002	Average
	0.225 mg/L Chloride Input			
Soil samples	7	25	8	13
Drain water	37	53	48	46
Measured	64	66	61	62[b]
	0.275 mg/L Chloride Input			
Soil samples	10	37	11	19
Drain water	50	74	66	62
Measured	64	66	61	62[b]

(a) Assumes 190 mm/yr average precipitation with 0.225 mg/L average chloride input or 0.275 mg/L average chloride input.
(b) Twenty-six year average.

Table 3.3-3. Calculated Grain-Size Statistics for the Lysimeter Soil and Sediment

Lysimeter Location	Soil	% Fines (< 0.05 mm)	% Sand (0.05 – 2 mm) (%)	% Gravel (2 – 152 mm)	D10[a] (mm)	d_g[b] (mm)	σ_g[b] (mm)
300 North	Sand[c]	3	96	1	0.192	0.506	0.554
FLTF	Loamy sand[c]	18	79	3	0.021	0.219	0.226
FLTF	Silt loam[c]	63	37	0	0.004	0.031	0.319
FLTF	Sandy gravel[c]	3	16	81	0.697	7.529	0.217
SWL	Gravelly sand	4	36	60	0.183	4.687	0.092

(a) D10 = Diameter of sieve openings that will retain 10% of the soil.
(b) d_g and σ_g are geometric mean particle diameter and geometric standard deviation, respectively.
(c) Used in fitting of the evaporation factor, E_f.
FLTF = Field Lysimeter Test Facility.
SWL = Solid Waste Landfill.

Table 3.3-4. Darcy's Law Drainage Estimates from Unsaturated Hydraulic Conductivity Functions for 300 North Sand

Method	Soil-Water Pressure Head (mm/yr)	Unsaturated Hydraulic Conductivity (mm/yr)	Drainage (mm/yr)
Particle Size (Arya and Paris 1981)	--	3,700	3,700
Steady State Column (Klute and Dirksen 1986)	32,000	20	20–32,000
Guelph Permeameter (Reynolds and Elrick 1985)	189,000	--	189,000
Ultracentrifuge (Nimmo et al. 1994)	--	25,000	25,000
Instantaneous Profile (Watson 1966)	45–55	45–55	45–55
Direct Measure (drainage lysimeter)	--	--	56

Note: Assumes a unit-gradient condition at a soil-water content of 0.09 m^3 m^{-3} and a soil-water pressure head of -0.40 meter.

Table 3.3-5. Average Winter (November through March) Precipitation, Average Drainage, and Calculated Evaporation Factor from Lysimeter Records

Lysimeter Location	Soil	Observation Period	Average Winter Precipitation (mm/yr)	Average Drainage (mm/yr)	Evaporation Factor (mm/yr)
300 N	Sand[a]	1995-2004	129	73	56
FLTF	Loamy sand[a]	1995-2004	129	32	97
FLTF	Silt loam[a]	1995-2004	129	0	129
FLTF	Sandy gravel[a]	1995-2004	129	111	18
SWL	Gravelly sand	1996-2004	121	51	70
300 N	Sand	1982-1993	109	54	55

(a) Used in fitting of the evaporation factor (E_f).
Note: Measurements for the sand soils are from the same lysimeter but over different time periods.
300 N = 300 North Lysimeter Site.
FLTF = Field Lysimeter Test Facility.
SWL = Solid Waste Landfill.

Table 3.3-6. Relationship Between Evaporation Factor and Soil Textural Descriptors, Goodness of Fit, Predicted 300 North (1981-1993) Sand, and Solid Waste Landfill Drainage from Developed Evaporation Factor Functions, and the Corresponding Measured Drainage

Method	Relationship	Goodness of Fit (r^2)	Predicted 300 North Sand Drainage (mm/yr)	Measured 300 North Sand Drainage (mm/yr)	Predicted SWL Drainage (mm/yr)	Measured SWL Drainage (mm/yr)
1	$E_f = 0.6609\ (\%fines \times D10)^{-0.8833}$	0.997	46	54	70	51
2	$E_f = 102.331 + (83.935 \times \%fines) - (9.270 \times d_g) - (80.224 \times \sigma_g)$	1.000	53	54	66	51

Note: The 300 North sand data set is from a 12-year record not used in the model calibration.
SWL = Solid Waste Landfill.

Table 3.3-7. Grain-Size Statistics and Predicted Drainage for Commercial Road-Base Material, Representing Tank Farm Surface Cover, and Two Synthetic Types of Soil

Soil	% Fines	% Sand	% Gravel	D10 (mm)	d_g (mm)	σ_g (mm)	Predicted Drainage (mm/yr)	
							Method 1	Method 2
Road-base	4	24	72	0.267	5.471	0.162	93	87
Synthetic A[(a)]	14	23	63	0.023	2.701	0.060	24	45
Synthetic B[(a)]	24	20	56	0.006	1.049	0.033	0	19

(a) Consists of road-base material with an addition of 10 and 20 percent fines.

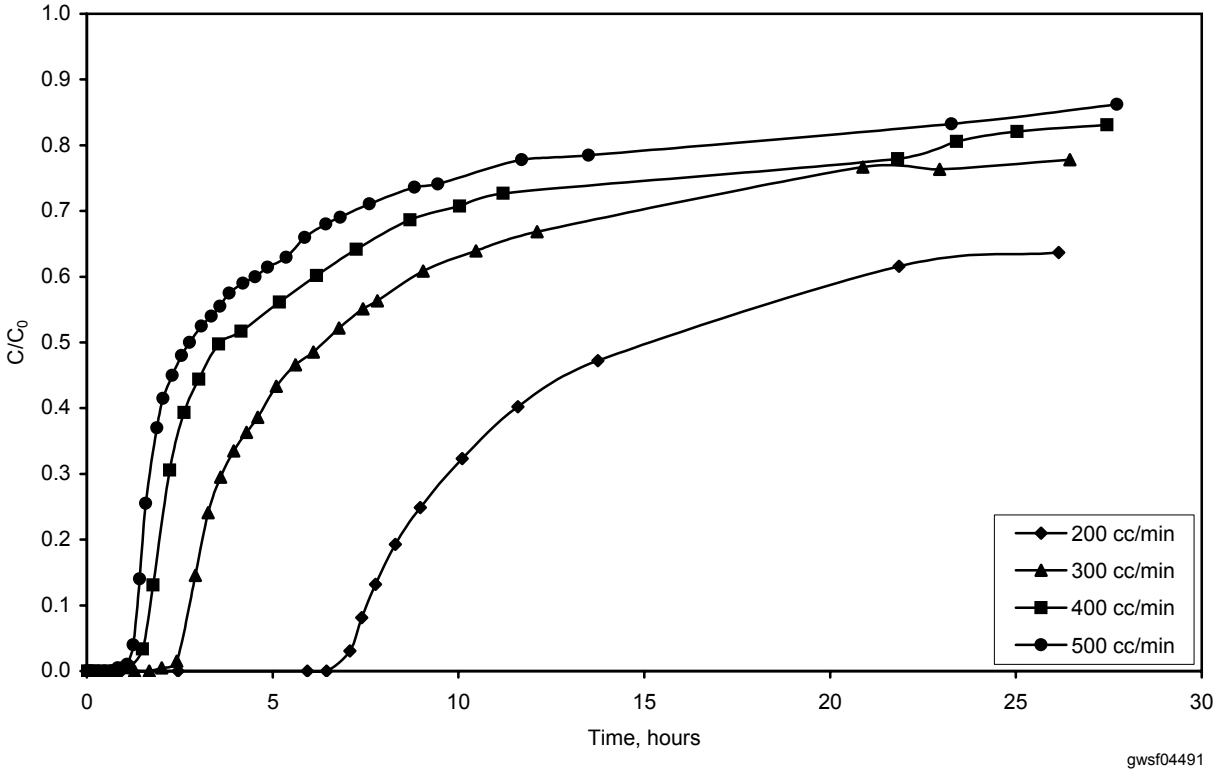


Figure 3.3-1. Hydrogen Sulfide Breakthrough Curves during the Initial 30 Hours of Small Column Treatment Tests with Hanford Formation Sediment

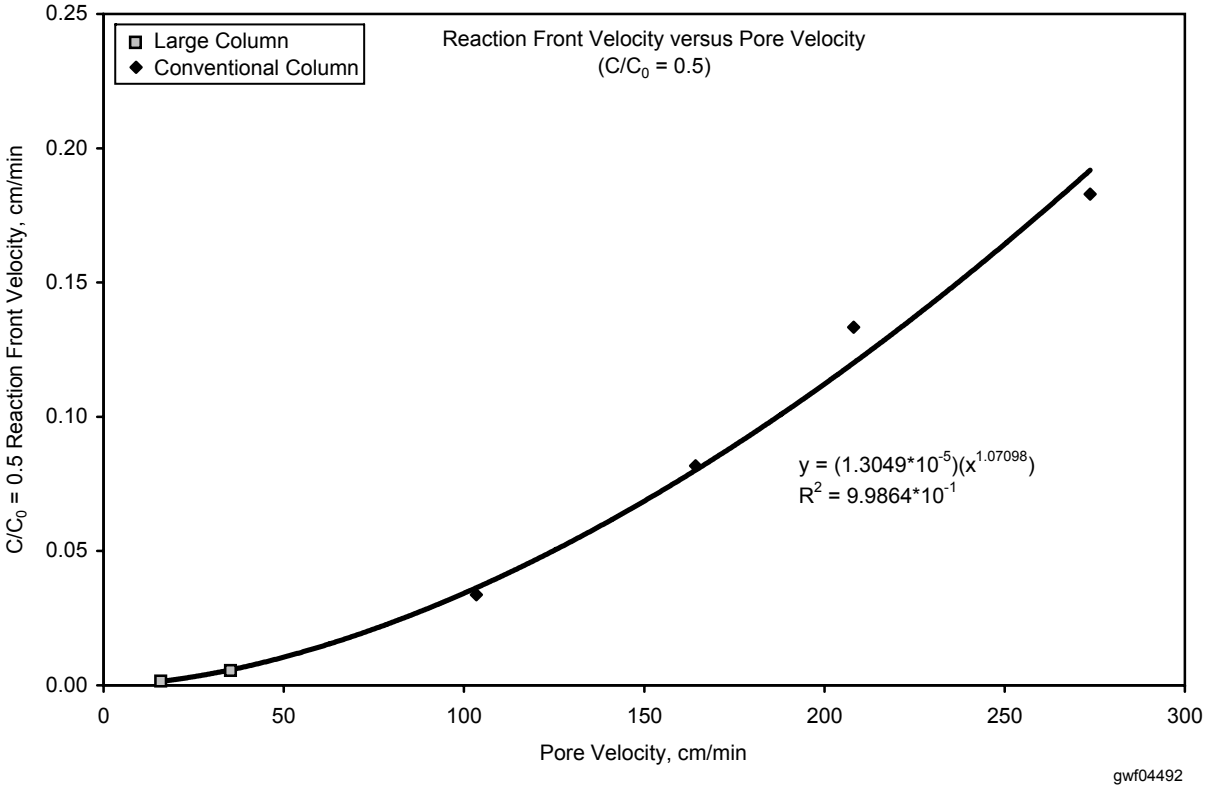


Figure 3.3-2. Reaction Front Velocity versus Pore Velocity for the Conventional and Large Column Gas Treatment Tests

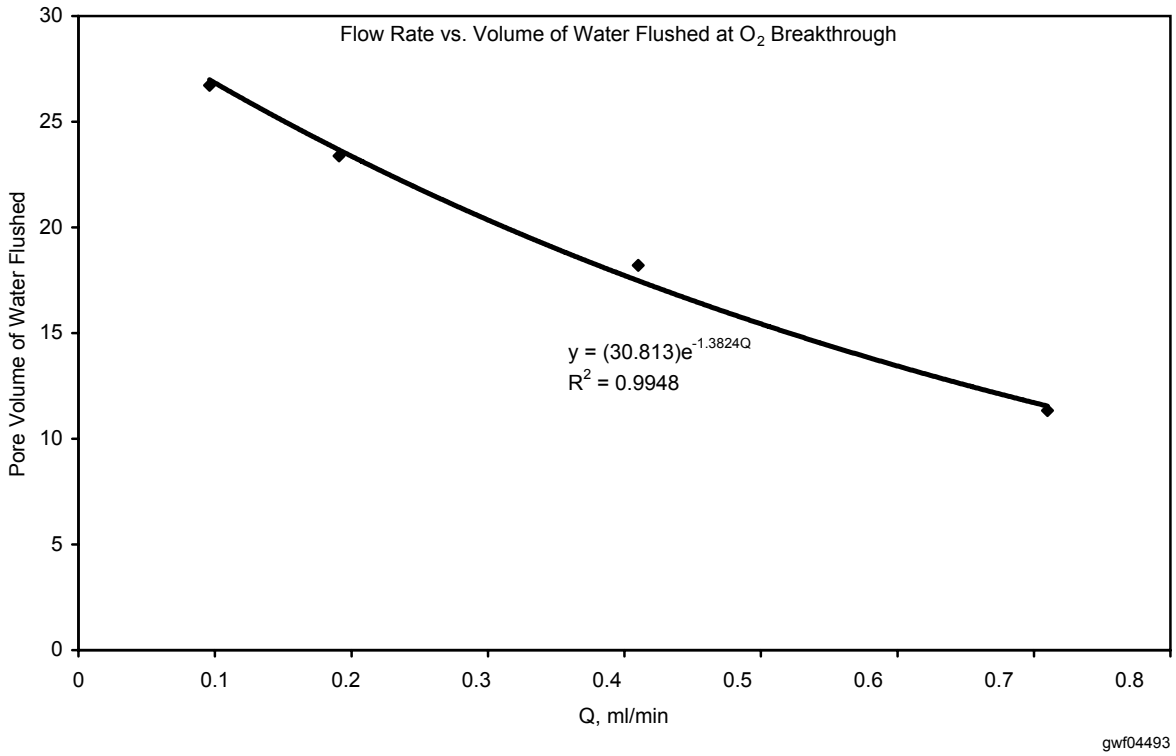


Figure 3.3-3. Water Flow Rate versus Number of Column Pore Volumes Associated with Initial Oxygen Breakthrough

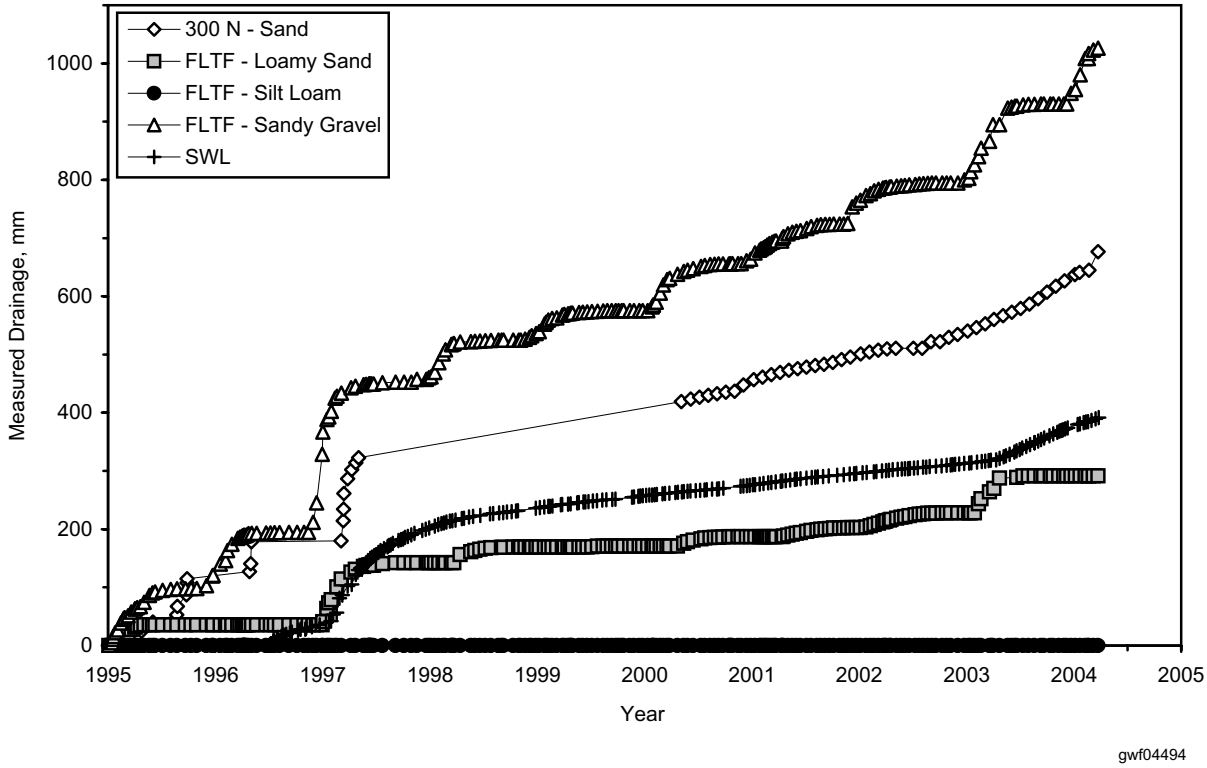
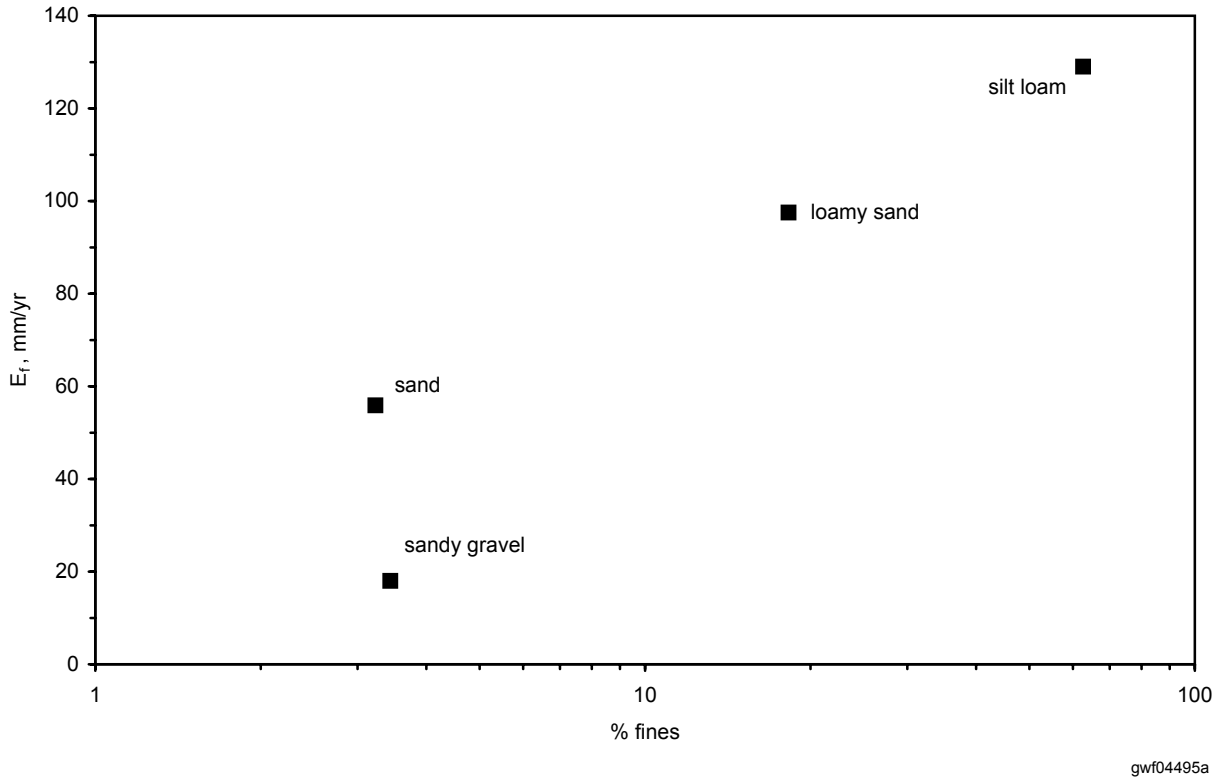


Figure 3.3-4. Cumulative Drainage for the 300 North and Field Lysimeter Test Facility Lysimeters from January 1, 1995, to April 2004, and Cumulative Drainage for the Solid Waste Landfill Lysimeter from July 1, 1996, to April 2004



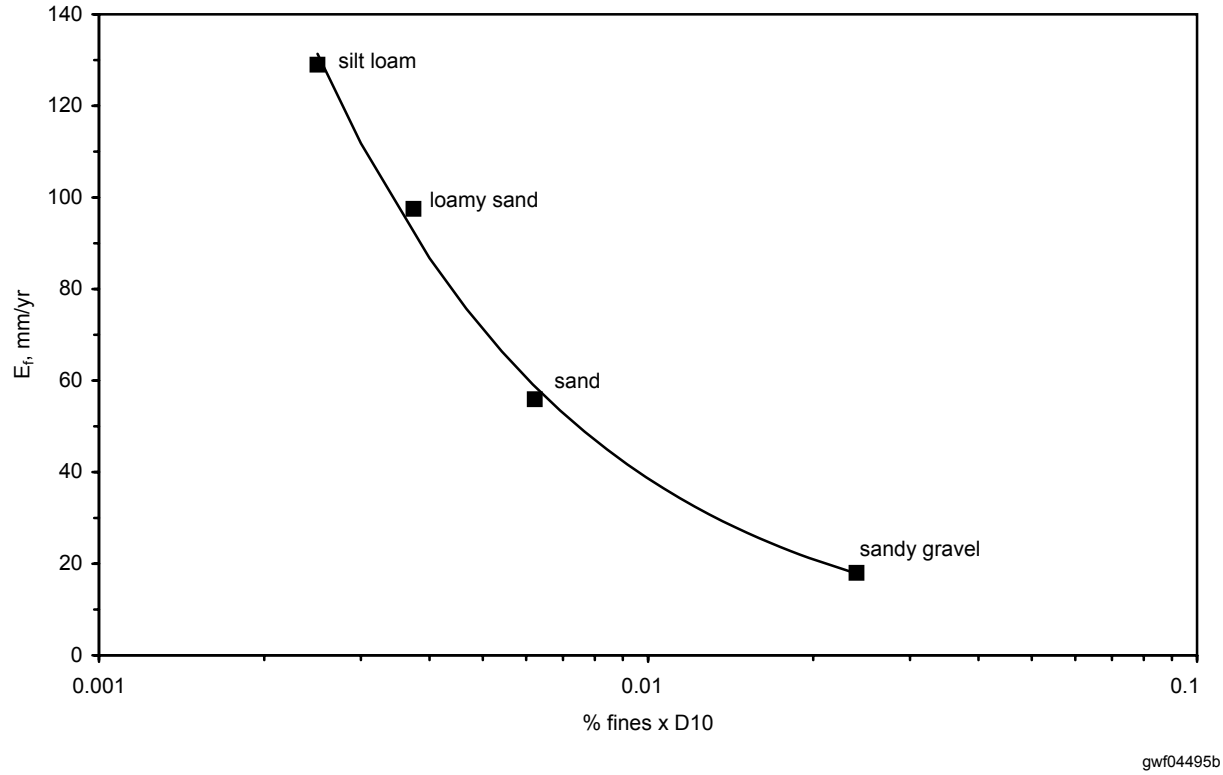


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Figure 3.3-5. Relationship Between the Evaporation Factor, E_f, and (a) Percent Fines; (b) Percent Fines x D10 (r^2 = 0.997). The solid line represents the developed E_f function.

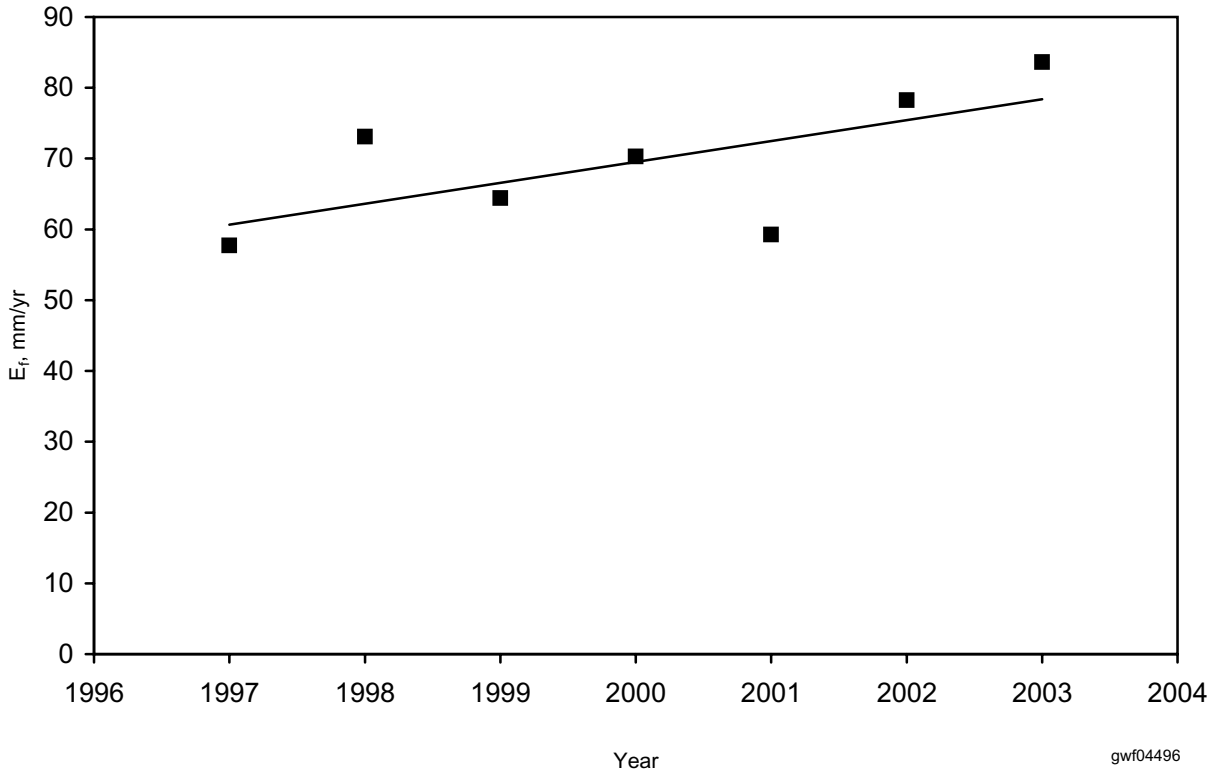

Figure 3.3-6. Solid Waste Landfill E_f Over Time as Calculated from Solid Waste Landfill