PNNL-14008

Evaluation of the Effects of Chromium on Fall Chinook Salmon in the Hanford Reach of the Columbia River: Integration of Recent Toxicity Test Results

D. D. Dauble T. M. Poston
G. W. Patton R. E. Peterson

May 2003

Prepared for the U.S. Department of Energy
under Contract DE-AC06-76RL01830

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Richland, Washington 99352

Executive Summary

This report summarizes results of a series of recent laboratory studies conducted to evaluate the effects of chromium on chinook salmon (*Oncorhynchus tshawytsch*). Individual studies focused on determining the relationship between exposure concentration and toxicological response for a range of life stages, including fertilization, egg through swim-up (early life history), parr health, and avoidance-preference of juveniles. Study designs were representative of possible exposure scenarios in the Hanford Reach of the Columbia River.

Results of fertilization tests showed no reduction of fertilization rates relative to control treatments over a concentration range of 11 to 266 μg/L hexavalent chromium. Although there is some uncertainty associated with estimating the concentration of chromium that eggs and sperm could be exposed to in the Columbia River, results indicate that chromium from Hanford groundwater sources poses no risk to the natural fertilization process of fall chinook salmon.

Early life history studies revealed no effect (neither a statistically significant difference among treatments nor a dose-dependent response) at any chromium concentration for percent viable hatch. Additionally, no effects on growth or survival were observed for chromium up to 266 μg/L. The on-site study showed a systematic increase in chromium level with increasing concentrations of chromium ≥22 μg/L. Although whole-body concentrations of chromium declined after swim-up (when exposures were stopped), tissue concentrations for the 54 μg/L exposure group remained 2x higher than controls. Overall, results revealed that salmon exposed to aqueous chromium to 266 μg/L during the eyed-egg to swim-up portion of their life cycle were not adversely impacted.

Treatment concentrations were increased during parr health studies from 24 to 120 μg/L and 54 to 266 μg/L after 105 days of exposure because of no apparent effects. Gross pathology observations indicated up to 20% of fish had abnormal coloration and markings in kidney. Cellular response data indicated extended duration of exposure and increased concentration of chromium contributed to histological effects. Observations of gills showed variable and non-dose dependent effects for lamellar hypertrophy and apoptosis. Assessments of blood DNA for chromium-exposed fish were contradictory between exposure days 105 and 134, suggesting use of this endpoint to measure toxicological response needs further refinement. Overall, studies showed that changes to fish health end points were associated with changes in growth and survival of chromium exposed fish following extended exposure period of 134 days, but not at 105 days.

Laboratory avoidance/preference tests showed that juvenile chinook salmon can detect and avoid chromium at concentrations ≥54 μg/L under conditions of 80 mg/L hardness. However, chinook salmon neither avoided nor showed preference for aqueous chromium at concentrations to 266 μg/L under conditions of 200 mg/L hardness. Information from these studies may be useful for specific locations where detailed information on contaminant plumes and microhabitat are known, but cannot be extrapolated to other life stages because of inherent differences in habitat and physiology.

The potential for adverse effects of chromium to salmon populations in the Hanford Reach are thought to be low due to the location and extent of chromium plumes relative to known spawning areas, dilution characteristics of the river, and residence timing of juvenile salmon. Data from three laboratory reports and results of an outside peer review process support the conclusion that the current cleanup criteria of 10 μg/L chromium is adequate for protecting fall chinook salmon populations.

Acknowledgments

The authors wish to acknowledge contributions from David R. Geist, Bruce N. Bjornstad, and Kris B. Olsen in the development of background information on the Hanford Site. Michael C. Newman, Lawrence R. Curtis, Aaron P. Roberts, and members of the Hanford Natural Resource Trustee Council provided valuable review comments. We also thank Terrie Bear and Kathy Neiderhiser for typing the report.

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1.0 Introduction

The Hanford Reach of the Columbia River is the last non-impounded portion of the Columbia River above Bonneville Dam in the United States. This section of the river provides essential riverine habitat for many aquatic organisms, including spawning and rearing habitat for fall chinook salmon (*Oncorhynchus tshawystcha*). The importance of the Hanford Reach as a unique aquatic ecosystem was recognized in 2000 by declaring it as a National Monument (65 FR 114). However, past operations related to nuclear materials production at the U.S. Department of Energy (DOE) Hanford Site have resulted in a legacy of environmental concerns for humans and wildlife. Chromium is one contaminant of major concern, and the DOE currently has activities underway to reduce the amounts of hexavalent chromium in groundwater entering the river (Poston et al. 2001).

The potential exposure of fall chinook salmon to hexavalent chromium has been an important topic of interest in the Hanford Reach because of the proximity of salmon habitats to contaminated Hanford groundwater. Past operations at the Hanford Site discharged hexavalent chromium to the river and also entered groundwater, thus contaminating surrounding nearshore areas. Recent studies showed that concentrations of hexavalent chromium in groundwater upwelling into the Columbia River exceeded the Washington State ambient water quality criteria of 10 µg/L (Hope and Peterson 1996a and 1996b; WAC 173-201A-040).

To address these issues, a series of laboratory studies were conducted by the United States Geological Survey (USGS) and Pacific Northwest National Laboratory (PNNL) to evaluate the effects of chromium on early life stages of chinook salmon. Direction for overall study design was provided by the Hanford Natural Resource Trustee Council (Trustees). Individual reports resulting from the laboratory studies included: 1) *The Potential for Chromium to Adversely Affect Chinook Salmon (Oncorhynchus tshawtscha) in the Hanford Reach of the Columbia River, Washington, USA* (Farag et al. 2000); 2) *Chromium Toxicity Test for Fall Chinook Salmon (Oncorhynchus tshawytscha) Using Hanford Site Groundwater: Onsite Early Life Stage Toxicity Evaluation* (Patton et al. 2001); and 3) *Laboratory Evaluation of the Behavioral Avoidance-Preference Response of Chinook Salmon (Oncorhynchus tshawytscha) to chromium in the Hanford Reach of the Columbia River, Washington, USA* (DeLonay et al. 2001).

The objective of this report is to summarize results of these three chromium toxicity reports, identify any uncertainties in study results, and provide a basis for defining future studies to assess the effects of chromium on chinook salmon. An earlier draft of this report was submitted to the Trustees for review in October 2001 and, subsequently, to an external peer review team of recognized scientific experts. The peer review process validated the content and conclusions of the draft. The report was revised based on review comments and reissued as a revised final draft in August 2002. The Trustees made an additional request in December 2002 that the U.S. Geological Survey (USGS) provide comments prior to finalize-tion of the report. The USGS/Columbia Environmental Research Center conducted a brief review and provided a set of comments for the administrative record on January 31, 2003. Their comments were considered in this final version.

To provide a context for study summaries provided in Section 5.0, we briefly describe the physical setting of the Hanford Reach (Section 2.0), what is known concerning chromium source and transport pathways at the Site (Section 3.0), and life history and habitats of fall chinook salmon (Section 4.0). Individual studies (Section 5.0) included a fertilization test, an off-site early life stage test, an on-site early life stage test, a fish health study (parr stage), and an avoidance-preference behavior evaluation. Sections 6.0 and 7.0 provide an assessment of the potential impacts of chromium to fall chinook salmon that reside in the Hanford Reach and overall conclusions to this document, respectively. More detailed information on sources, distribution, and remediation of chromium at the Hanford Site is included in Appendix A. This report is not intended to be a comprehensive evaluation of all possible effects of chromium on aquatic organisms in the Hanford Reach of the Columbia River.

2.0 Description of the Hanford Reach of the Columbia River

2.1 Geographic Setting

The Columbia River is the second largest river in the contiguous United States in terms of total flow and is the dominant water feature on the Hanford Site. The need for large security and safety buffer zones at the Hanford Site precluded the development of dams on this section of river and allowed the Hanford Reach to remain the only unimpounded section of the Columbia River (above Bonneville Dam) in the United States (Figure 2.1).

General flow patterns and current channel configuration is a product of fluvial processes associated with the Columbia River since the end of the last ice age (~12,000 yrs B.P.) when a series of giant ice-age floods last inundated the region (Baker et al. 1991). Since that time, the Columbia River has been more-or-less confined to its present position by extremely coarse flood deposits, which bound the present river channel. The river is generally confined to a narrow channel along the straight, north-south segment north of Richland. Upstream, however, in the vicinity of the "horn," the river has widened considerably during unregulated high seasonal flow. This is indicated by a network of fluvial channels in the vicinity of the 100-D and 100-H Areas that dissect older cataclysmic-flood and fluvial deposits across the horn (Figure 2.2). Today, upstream flow regulation practices prevent the river from rising to flood stage, so that regulated flows are not expected to occupy these abandoned channels.

River substrate in the Hanford Reach is primarily composed of pebble-cobble gravel with fine to coarse sand filling the matrices between gravel clasts. In most cases, this substrate is derived from recent fluvial reworking of the Pleistocene cataclysmic flood deposits (i.e., Hanford formation) or Miocene-Pliocene-age ancestral river deposits of the Ringold Formation. Occasionally, the river may scour into, and flow directly on top of, coarse-grained deposits of either the Hanford formation (e.g., near Locke Island; Figure 2.3) or the partially indurated Ringold Formation (e.g., adjacent to 100-K Area). Locally, finer-grained sand and silt may be deposited in slackwater areas, such as along the inside of meanders, sloughs, or on the lee sides of mid-channel islands or bars.

2.2 River Flow Characteristics

The Columbia River originates in British Columbia, Canada and drains a watershed of about 680,000 km^2. The Columbia River, within the United States, has seven dams above the Hanford Site and four dams below. The Priest Rapids Dam is the nearest upstream dam from the Hanford Site and McNary Dam is the nearest downstream dam. The section of the river known as the Hanford Reach extends approximately 82 km from the base of Priest Rapids Dam to the upstream limit of the McNary Dam impoundment (Lake Wallula) near the southern boundary of the Hanford Site. Flows through the Hanford Reach are controlled primarily by release at Priest Rapids Dam and fluctuate over a large range to accommodate electrical power production and fishery requirements (Figure 2.4). The long term annual average flow is approximately 3,360 m^3/s, with daily flow ranging from 1,000 to 7,000 m^3/s (McGavock et al. 1987). Flows typically peak from April to June during the spring runoff and are lowest during

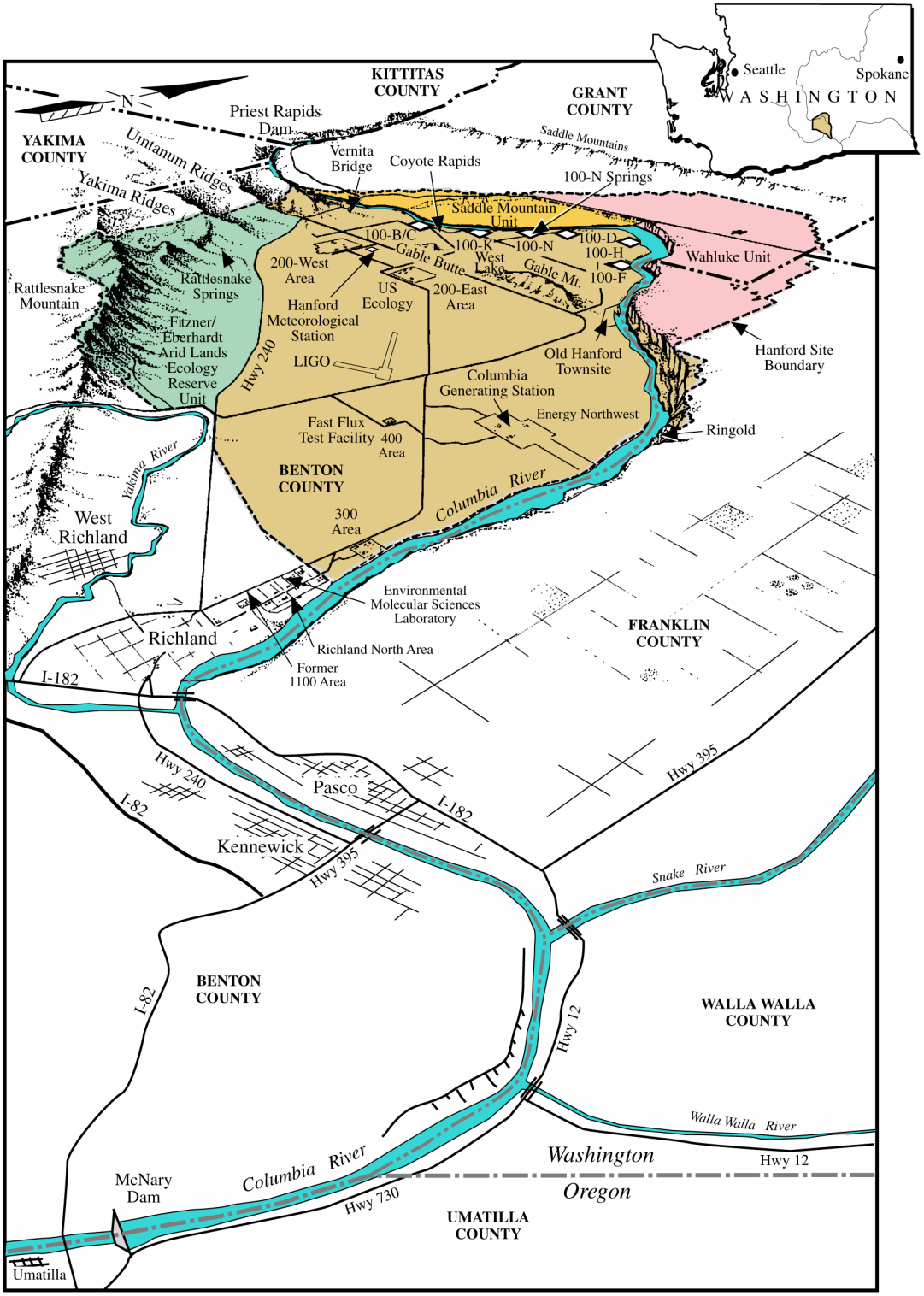

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Figure 2.1. The Hanford Reach and Surrounding Areas

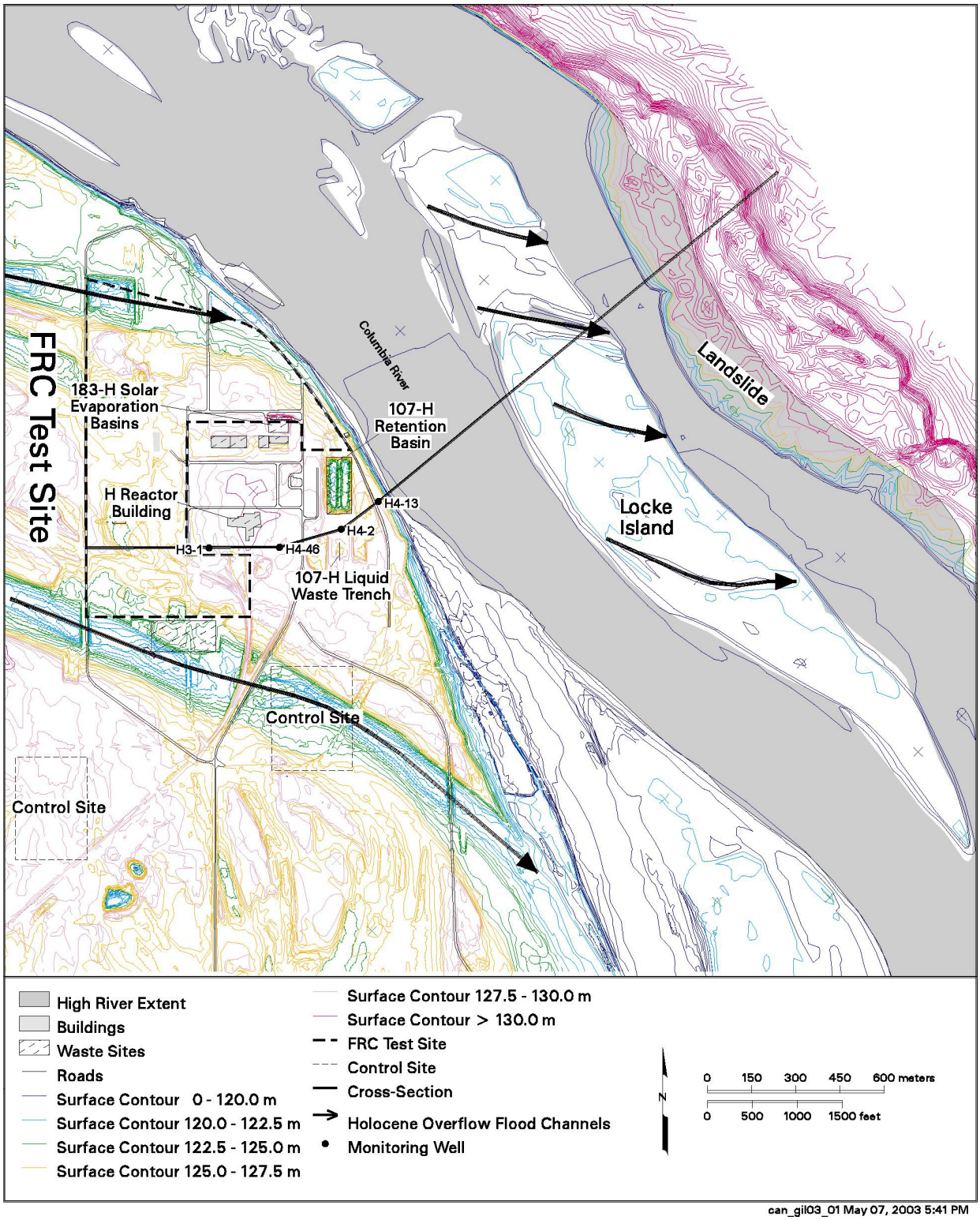

Figure 2.2. Fluvial Channels at the 100-H Area

Figure 2.3. Aerial Photo of the Hanford Reach near Locke Island

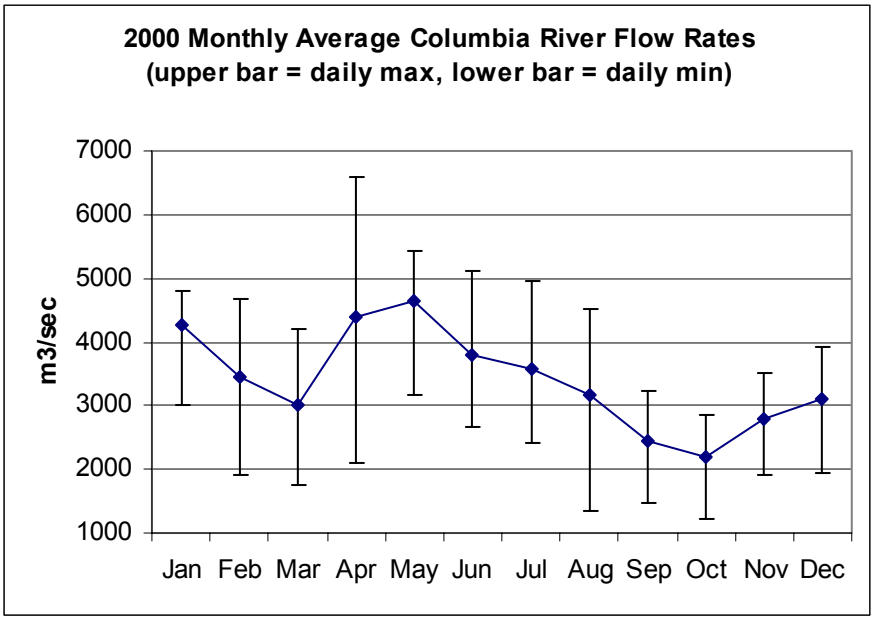

Figure 2.4. Discharge from Priest Rapids Dam Upstream of the Hanford Reach in 2000

September to October. The river stage can also change considerably during a 24-hr period (Figure 2.5) with vertical variations of 1.5 m commonly observed (Dirkes 1993).

2.3 Groundwater/River Interface

Under current conditions, the bulk of contamination that leaves the Hanford Site does so by way of the groundwater flow system, which discharges through a zone of interaction (ZOI) and across various interfaces with the Columbia River channel. The principal features and terminology associated with the ZOI are illustrated in Figure 2.6 (Peterson and Connelly 2001). The groundwater/surface water interface can be subdivided into two principal zones: the riparian zone, which is characterized by being alternately submerged and exposed in response to the fluctuating river stage, and the riverbed zone that is continuously submerged (hyporheic zone).

Within the riparian zone, discharge from the ZOI frequently appears as riverbank seepage during periods of low river stage. River water infiltrates the banks during periods of high river stage and forms either a layered system or a mixture during interaction with the approaching groundwater. As seepage continues to flow during the period of low river stage, the composition of the seepage may change dramatically from nearly pure river water to primarily groundwater. At the continuously submerged interface, discharge occurs within the riverbed sediments that are constantly submerged, i.e., at elevations below the lowest river stage. This region contains sediment porewater that is influenced by the gradual influx of groundwater that upwells from the underlying aquifer and by the entrainment of river water.

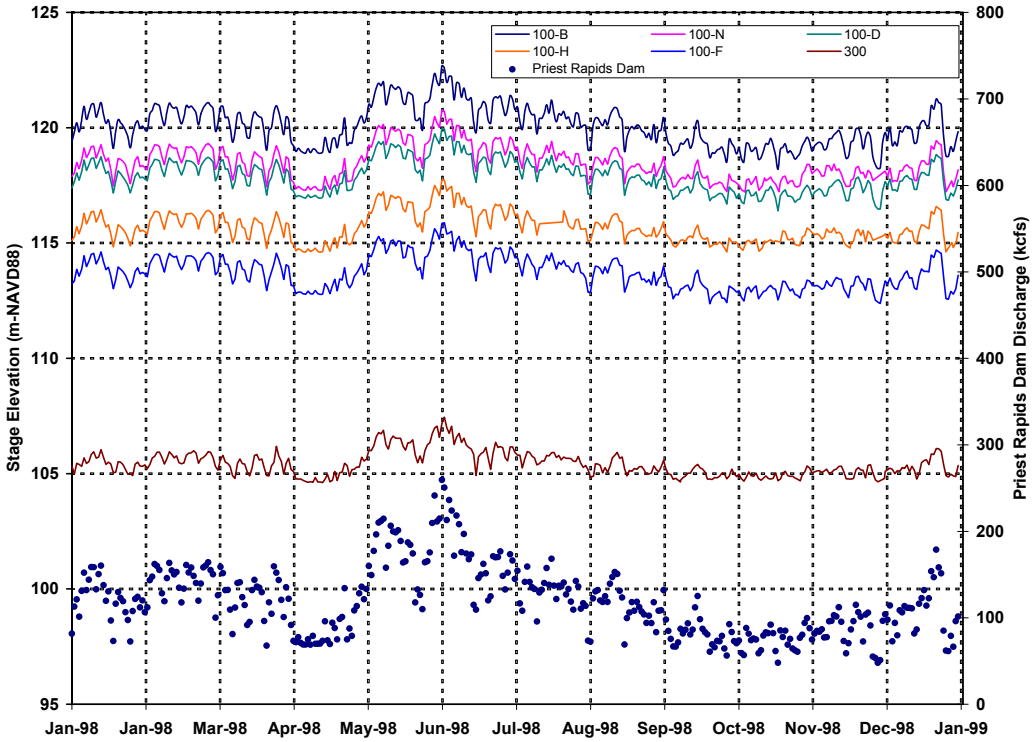

Figure 2.5. River Stage Elevation and Discharge from Priest Rapids Dam at Selected Locations of the Hanford Reach During 1998

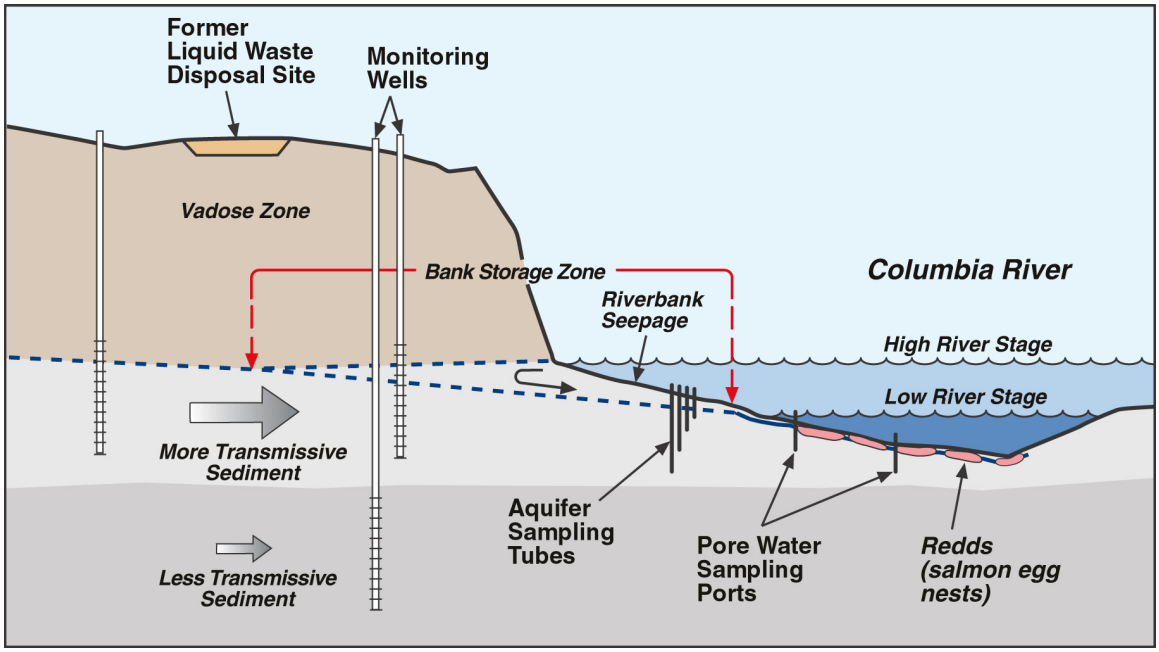

Figure 2.6. Conceptual Model for the Zone of Interaction Between Groundwater and the Columbia River

Physical, chemical, and biological processes occur within the ZOI that potentially alter the characteristics of the approaching groundwater. Current information suggests that physical processes are the dominant influence on contaminant concentrations and fluxes at locations of discharge into the free-flowing stream of the river. Physical processes include (a) layering and mixing of groundwater and river water, which infiltrates the banks and riverbed sediments, and (b) variable hydraulic gradients caused by river stage fluctuations. Chemical processes may change the characteristics of a contaminant in groundwater such that it becomes less mobile (e.g., adsorbs to sediments or precipitates). Biological activity in the zone may sequester contaminants and immobilize them or introduce them to the food chain.

3.0 Chromium Sources and Transport Pathways

This section describes the general chemical characteristics of chromium and current distribution of chromium in groundwater near the Columbia River. Additional information on current conditions is contained in the most recent Groundwater Monitoring Project Annual Report (Hartman et al. 2001) and the Project website at http://hanford-site.pnl.gov/groundwater.

3.1 Geochemistry of Chromium

The focus of this report is on chromium from past operations at the Hanford Site. However, chromium is also a naturally occurring metallic element found in terrestrial and marine environments, including basaltic rock fragments within sedimentary formations that overlie the Columbia River Basalt Group. In uncontaminated groundwater beneath the Hanford Site, the dissolved chromium concentration ranges between 0.5 and 4.4 μg/L (DOE/RL 1997). During 1999, results for dissolved chromium for numerous water samples collected from various depths and locations in the Hanford Reach indicated non-detection or very low concentrations in the range of 0.1 to 0.4 μg/L (Poston et al. 2000).

Chromium commonly occurs in the +3 and +6 oxidation (i.e., valence) states in nature. Trivalent chromium is the less soluble and therefore less mobile form, while hexavalent chromium is fully soluble and, therefore, mobile in groundwater and surface water systems. Trivalent chromium is most frequently associated with solid material (e.g., suspended particulate matter and precipitates) and hexavalent chromium is usually associated with the dissolved phase (Rai et al. 1989). Hexavalent chromium is the more toxic of the two forms and is of most concern with respect to protecting aquatic organisms.

Groundwater, riverbank seepage, and shoreline sediment samples were analyzed in 1993 and 1994 to determine the oxidation state of chromium in various parts of the Hanford environment (Thornton et al. 1995). The principal conclusion was that chromium present in the unconfined aquifer is predominantly hexavalent, and remains so because of the low amounts of organic matter and ferrous iron (Fe^{2+}) available in the aquifer. Thus, the investigators noted only slight potential for reduction of hexavalent chromium to trivalent chromium by ferrous iron and organic matter at the groundwater/ river interface. Consequently, most chromium transported to the river channel via the aquifer occurs as hexavalent chromium.

3.2 Distribution of Chromium Near the River

The distribution of chromium in groundwater beneath the Hanford Site is described in the annual Groundwater Monitoring Project report (Figure 3.1; Hartman et al. 2001). A summary of recent chromium concentrations in wells located near the river is included in Appendix A. The chromium plumes of most concern to aquatic resources are beneath the 100-K, 100-D, and 100-H Areas, because of relative high concentrations and their proximity to riverbed habitat at those areas. Active remediation of those plumes is in progress using pump-and-treat systems and *in situ* redox manipulation (Poston et al. 2001).

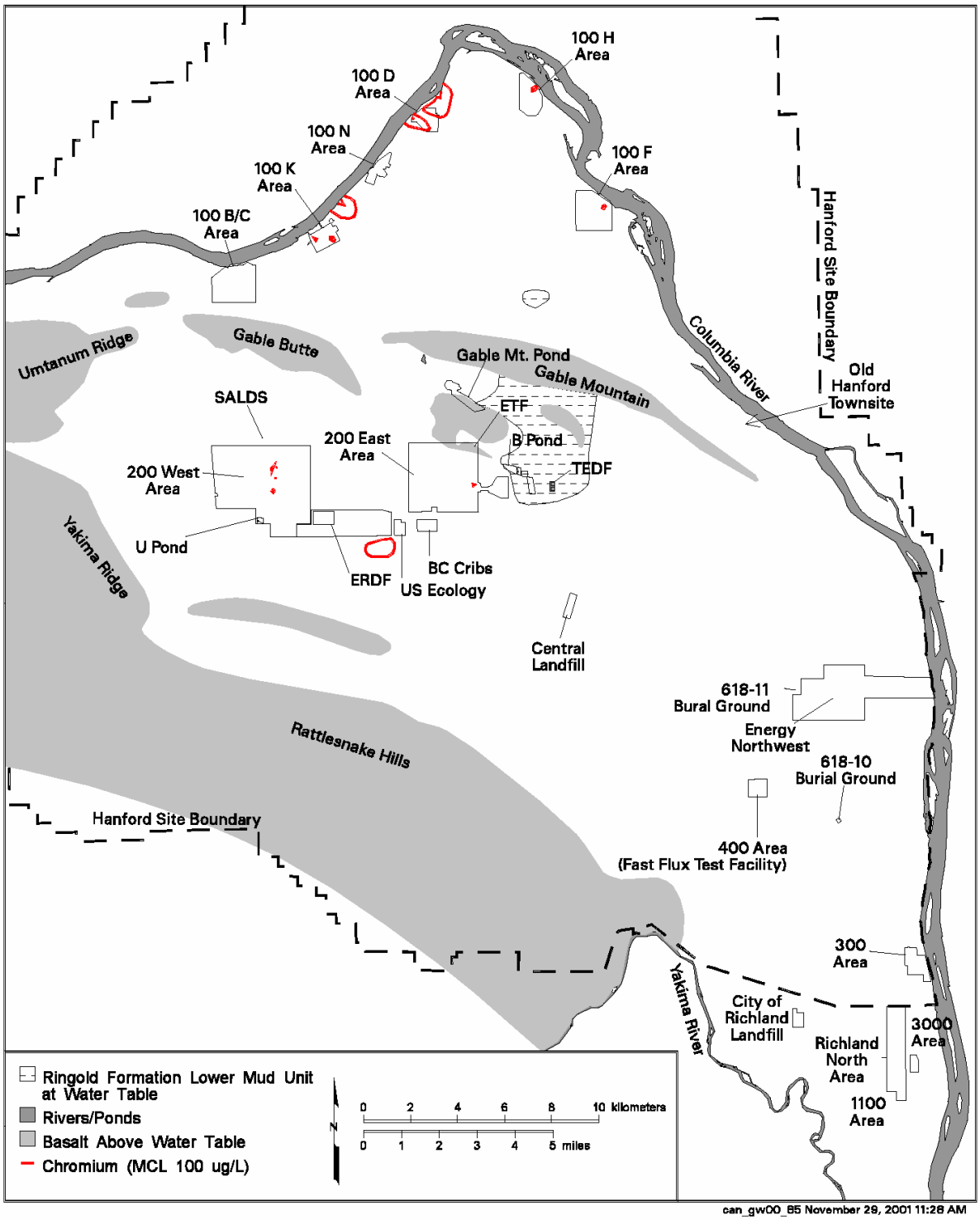

Figure 3.1. Hanford Site Map Showing the Chromium Concentrations (above 100 µg/L) in Groundwater

3.2.1 100-K, 100-D, and 100-H Area Chromium Plumes

The distributions of chromium in groundwater beneath the 100-K, 100-D, and 100-H Areas during fiscal year 2000 (October 1 to September 30) are shown in Figures 3.2, 3.3, and 3.4, respectively. Contour lines represent the approximate boundaries for the extent of contamination. The shape of the plumes reflects (a) the observed concentration of chromium at monitoring wells, (b) information on source facilities/waste sites, and (c) the direction and rate of groundwater movement. Where monitoring well coverage is limited, dashed lines are used to indicate increased uncertainty in the extent of the plume.

The major plume at 100-K (Figure 3.2) is associated with effluent disposal to the 100-K liquid effluent disposal trench during the operating years 1955 to 1971. The mound that formed beneath the trench created a radial flow pattern and caused chromium-contaminated water to be moved as far inland as 1,600 m. The chromium observed in groundwater currently is maintained to some extent by the continued, slow downward migration of moisture from the overlying soil column beneath the trench, and possibly by advection of contaminated water that was forced inland during the years that the mound was present. Pumping wells are currently removing groundwater from the area between the trench and the river, with the treated effluent being injected back into the aquifer at an upgradient location.

Two other smaller chromium plumes are present beneath the 100-K Area. The first is a relatively high concentration "hot spot" located at the southeast end of the 183-KE water treatment plant basins. Well 199-K-36 has shown chromium concentration as high as 2,700 μg/L in the past. The source is contamination in the overlying vadose zone from past spillage and leakage of sodium dichromate from transfer and storage facilities. This plume is apparently small, because no evidence of widespread distribution from this source area is visible in down gradient wells. The second area is centered around the KW reactor building, where three wells have shown consistently high concentrations of chromium for several years. The source for this plume is not well known, although it is most likely associated with sodium dichromate storage and handling facilities, or possibly made-up reactor coolant water.

Chromium is distributed in two general regions in the 100-D Area (Figure 3.3). The source for the more northerly plume is believed to be sodium dichromate storage tanks, transfer facilities, and pipelines that were located near the 105-D reactor building. Significant amounts of sodium dichromate are known to have leaked or spilled to the ground during the operating years. This plume has spread fairly widely because of a bend in the configuration of the water table, i.e., flow directions from the 105-D reactor location vary within a corridor of northwest to northeast. This plume is the target of interim remedial action by a pump-and-treat system. The extraction wells are indicated on Figure 3.4 and the extracted groundwater is piped to the 100-H Area for treatment.

The second chromium-contaminated region is frequently referred to as the "100-D hot spot" and is located in the western portion of 100-D Area. This plume was discovered in 1995 during an investigation of porewater in the adjacent river channel gravels (Hope and Peterson 1996a). It was confirmed by samples from aquifer sampling tubes installed at the shoreline and by the installation of groundwater well 199-D4-1 (Rohay et al. 1999). The plume has the highest concentrations of chromium of all the 100 Area plumes, with concentrations of the core of the plume reaching approximately 2,000 μg/L. This plume is the target of interim remedial actions using *in situ* redox manipulation methods. A test barrier was

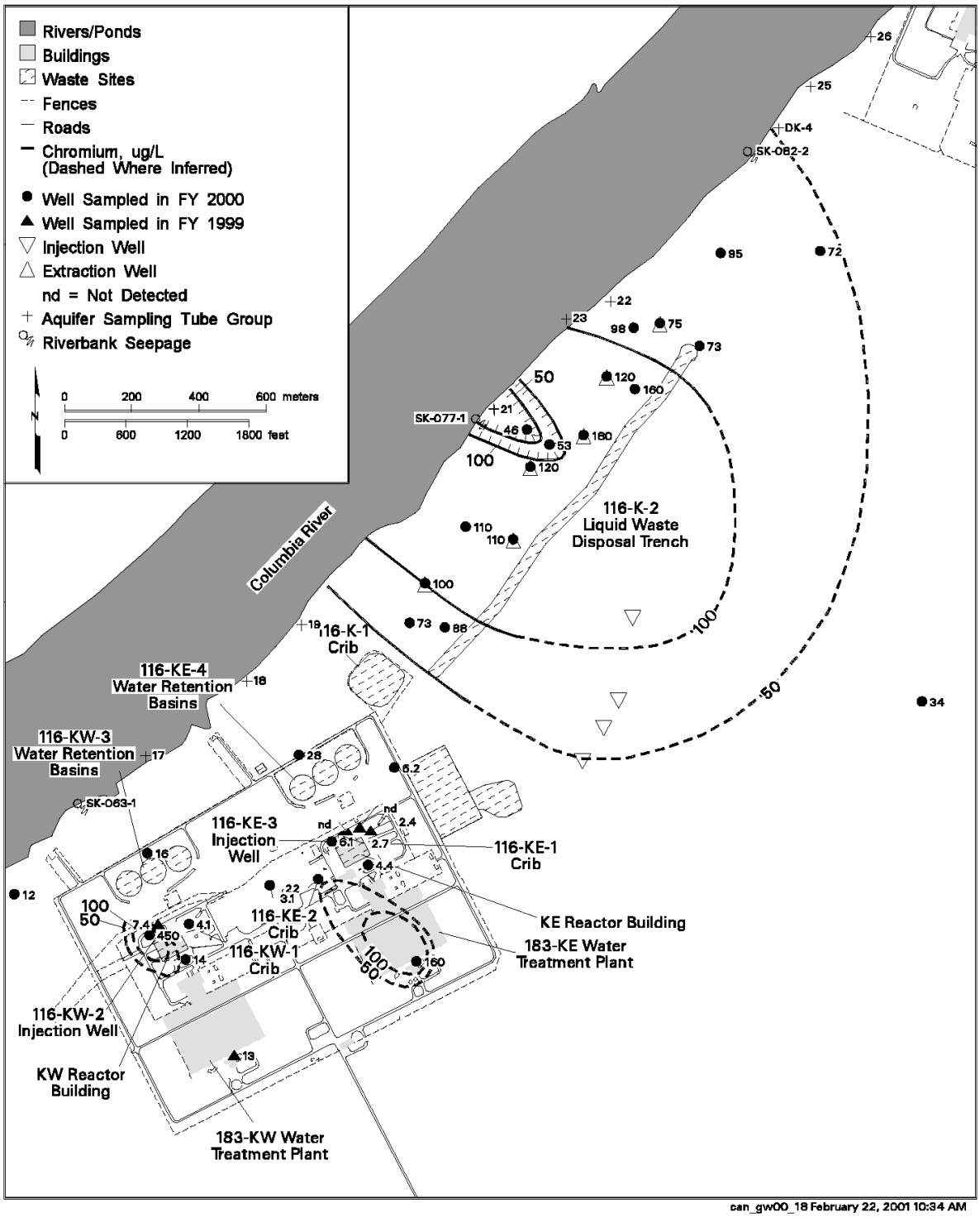

Figure 3.2. Distribution of Chromium in Groundwater Beneath the 100-K Area During Fiscal Year 2000

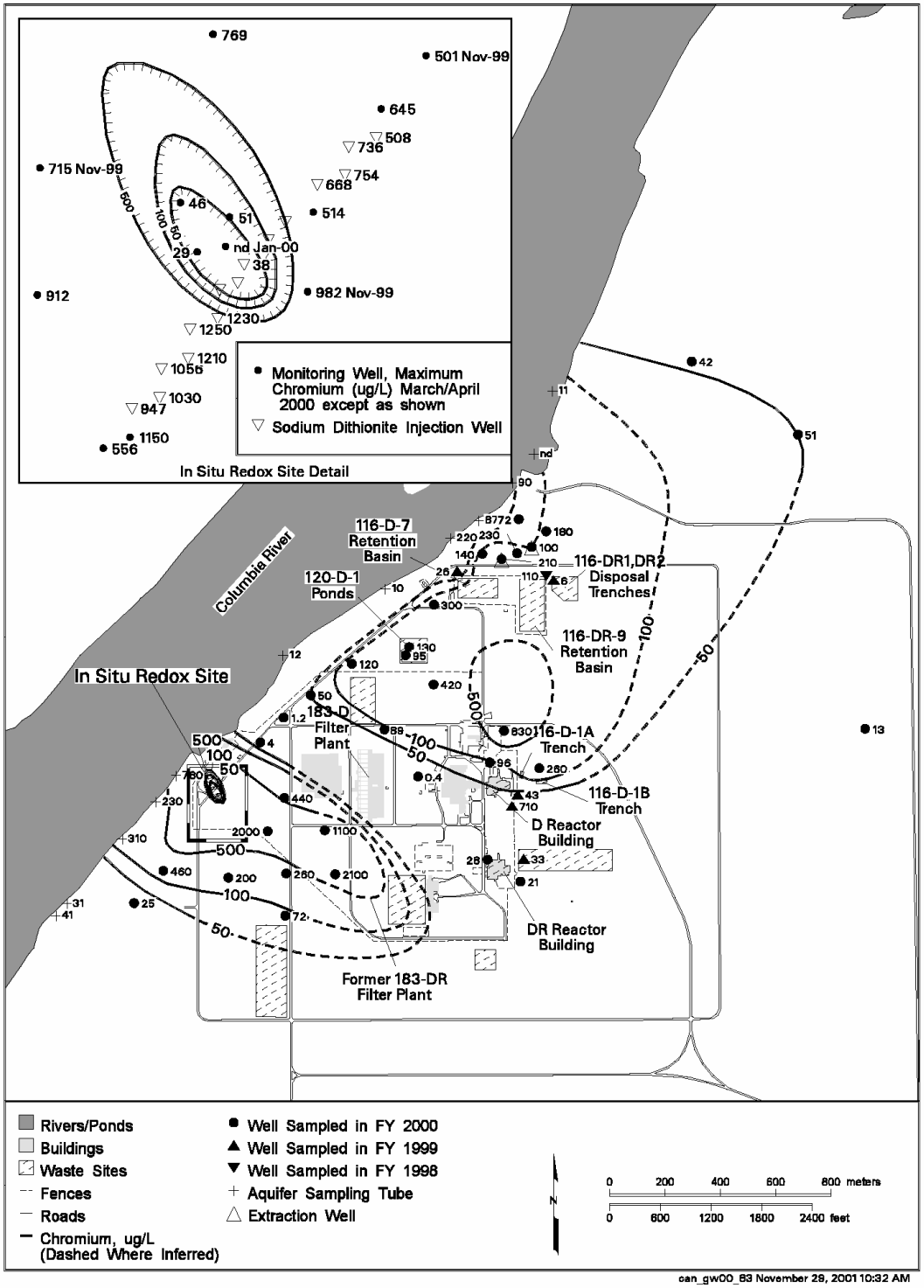

Figure 3.3. Distributions of Chromium in Groundwater Beneath the 100-D Area During Fiscal Year 2000

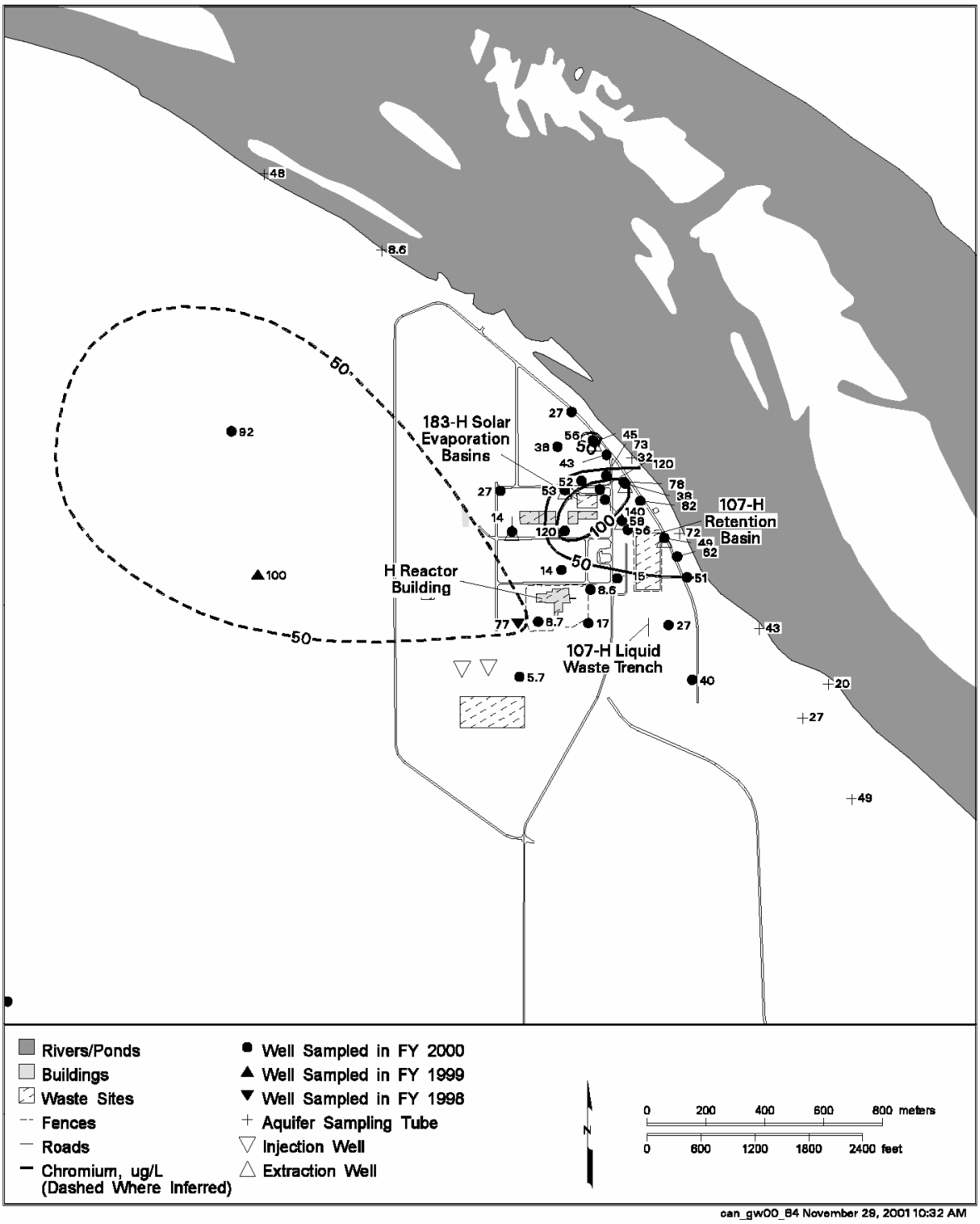

Figure 3.4. Distributions of Chromium in Groundwater Beneath the 100-H Area During Fiscal Year 2000

installed near the river during 1997 and 1998, and full-scale activities began in 2000 (Williams et al. 2000). A line is included on Figure 3.3 to indicate the approximate extent of the barrier created by the reduction of hexavalent chromium to trivalent chromium.

The chromium plume at 100-H Area is shown in Figure 3.4 and is the smallest of the chromium plumes currently being targeted by pump-and-treat systems under interim remedial action. As in the other reactor areas, residual amounts of chromium probably remain in the vadose zone because of the mound that was generated beneath the leaking retention basin during the operating years. Two other sources contributed chromium to the plume observed currently; leakage and spillage from fuel fabrication wastes formerly stored in the 183-H Solar Evaporation Basins, and migration into the 100-H Area of chromium-bearing groundwater, presumably from the 100-D Area. The total size of the chromium plume at 100-H Area appears to be getting smaller because of the interim remedial action using pump-and-treat methods.

Chromium is present in groundwater at the other reactor areas (i.e., 100-B, 100-N, and 100-F Areas), but not at concentrations that have warranted interim remedial actions. It has not been considered a contaminant-of-concern at the 300 Area or along other segments of shoreline where plumes from the 200 Areas have reached the river, such as the Hanford Townsite. A summary of concentrations in near-river wells for these areas during year 2000 is provided in Table A.1.

3.2.2 Chromium in Riverbank Seepage and Nearshore Environments

A comprehensive summary of contaminant data for riverbank seepage and sediment associated with seepage sites was completed in 1992 (Peterson and Johnson 1992), relying primarily on data collected during the previous year under CERCLA limited field investigations (DOE/RL 1994). Hexavalent chromium observed in riverbank seepage samples ranged from undetected to a maximum of ~120 μg/L along the Hanford Reach from the 100-B Area to several miles downstream of the 100-F Area (Figures 3.5 and 3.6). Only two results exceeded the 100 μg/L drinking water standard, although many exceeded the more stringent 11 μg/L standard for protection of freshwater aquatic organisms. The highest concentrations were observed at the 100-D Area along shoreline adjacent to the current interim action via pump-and-treatment. At that time, the chromium "hot spot" in the western part of the 100-D Area had not yet been discovered, and no seepage sites were found along that segment of shoreline during the 1991 fieldwork.

Fine-grained sediment associated with riverbank seepage sites were also collected and analyzed for chromium during the fall 1991 study (DOE/RL 1992). Chromium concentrations ranged from very low to a maximum of ~120 mg/kg, with the highest values occurring at the same 100-D Area location as the peak seepage samples. Most of the chromium values for seepage sediment fall below a reference value for world average shale of 90 mg/kg (Bowen 1966). Many results were near or below the background value for chromium in Hanford Site soils, which is in the range 3 to 33 mg/kg (DOE/RL 1992).

During September 1991 through January 1992, periphyton samples from artificial substrate at the 100-N and 100-H Areas (Cushing 1993) were analyzed for radionuclides, heavy metals (including chromium), and trace elements. Concentrations of chromium in the periphyton ranged from ~16 to ~32 mg/kg (dry weight of sample) - values comparable to background levels for Hanford Site soils. One

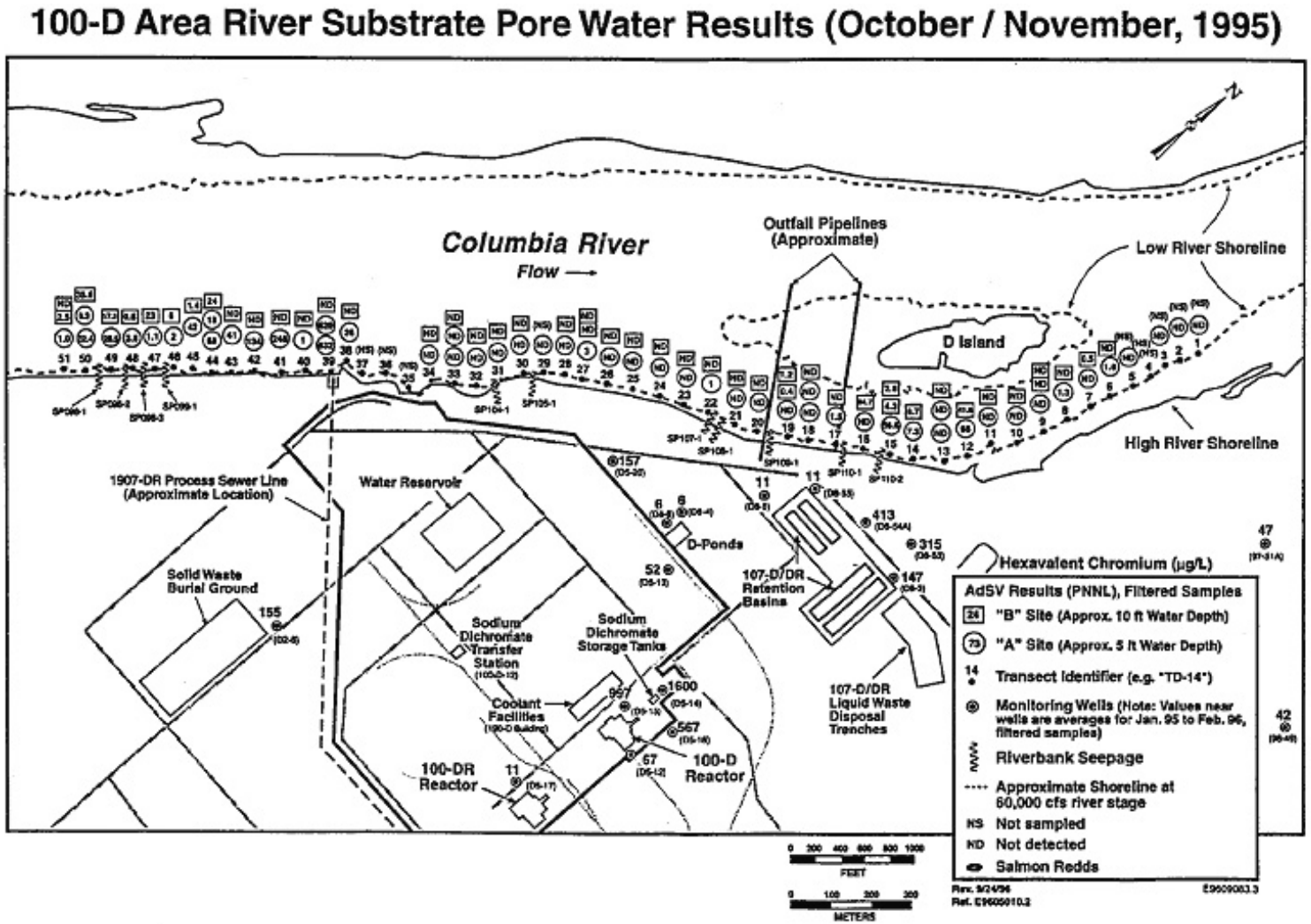

Figure 3.5a. Water Quality Monitoring Results for Seeps, Aquifer Tubes, Substrate Porewater, and Nearshore River Water at the 100-D Area (from Hope and Peterson 1996a)

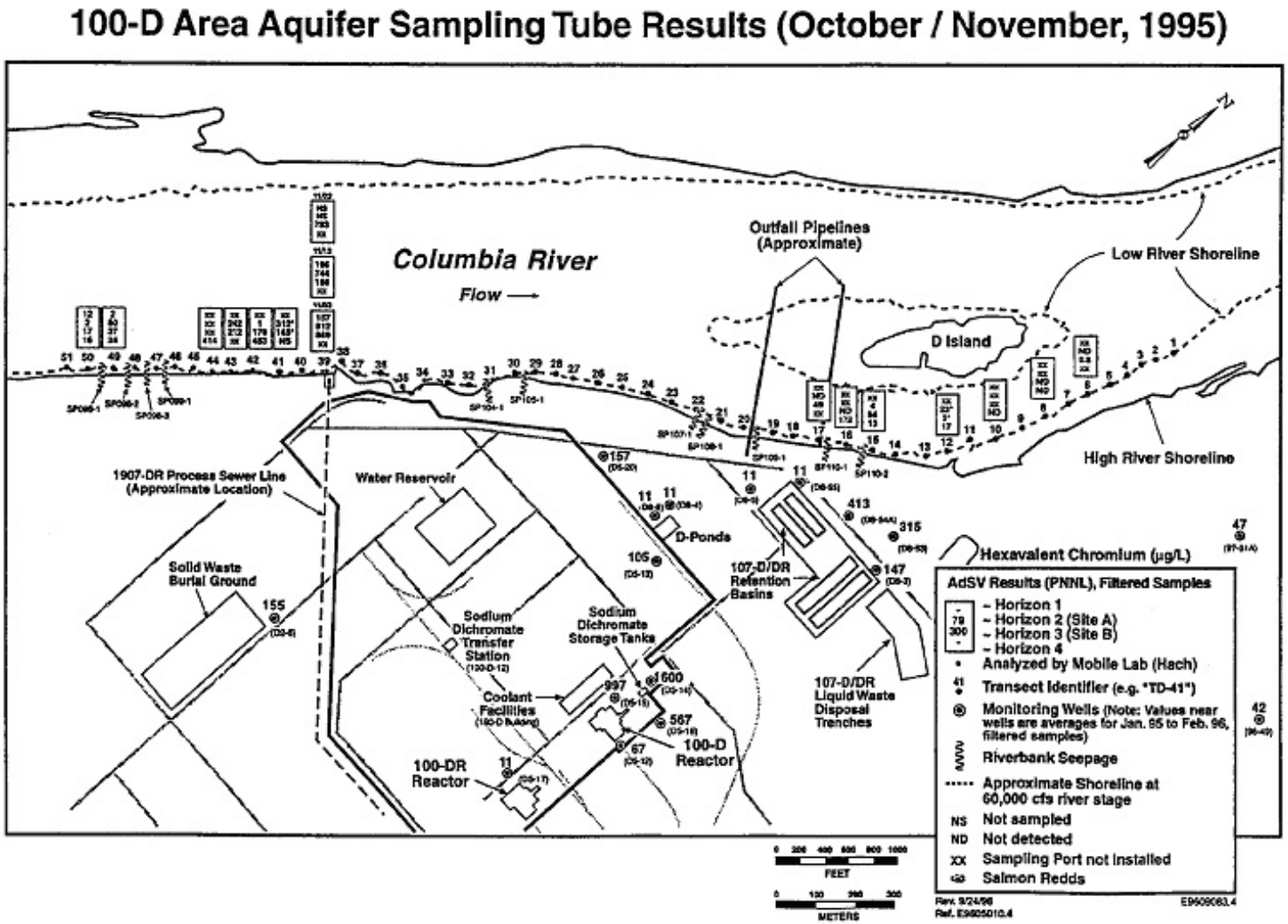

Figure 3.5b. Water Quality Monitoring Results for Seeps, Aquifer Tubes, Substrate Porewater, and Nearshore River Water at the 100-D Area (from Hope and Peterson 1996a)

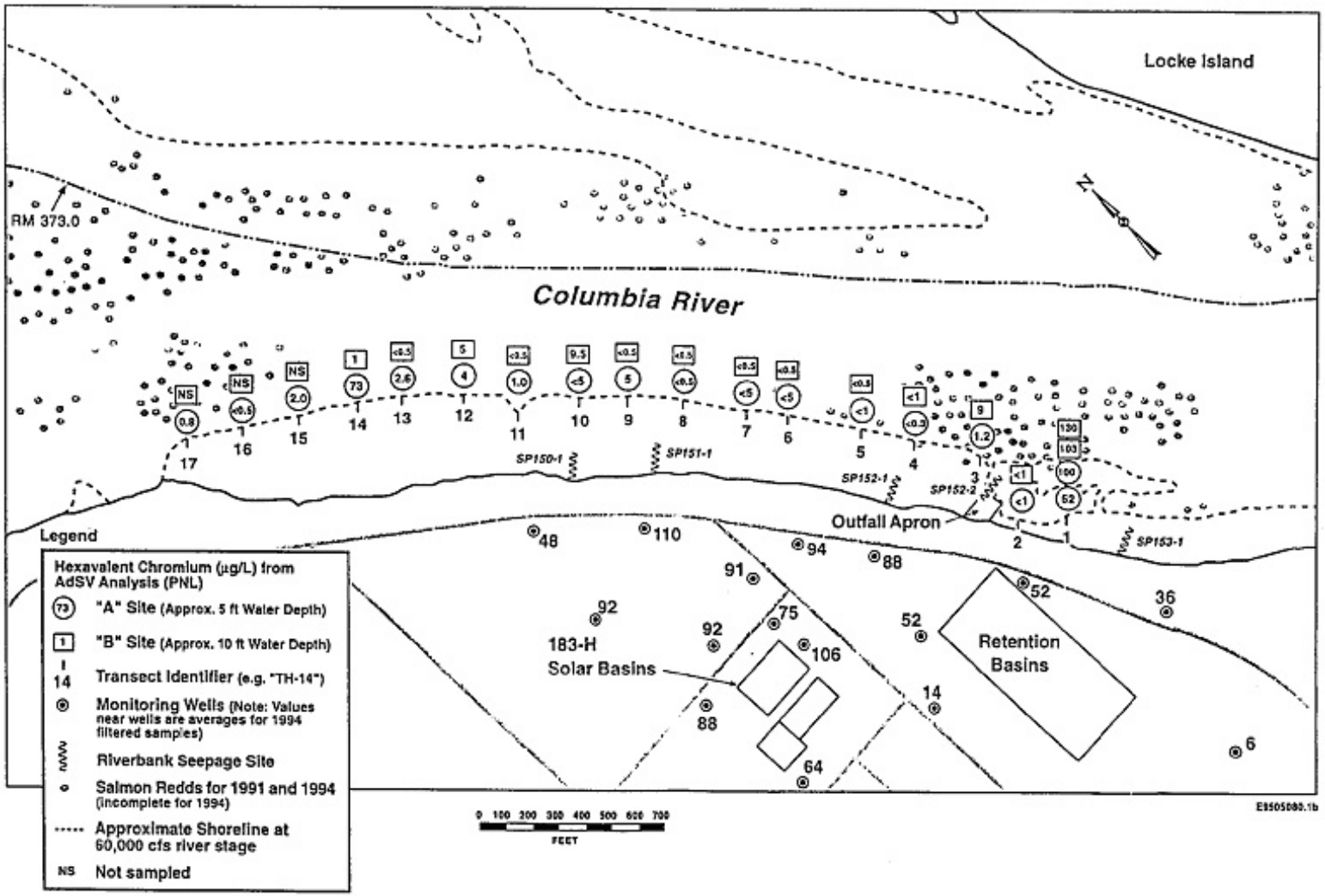

Figure 3.6. Water Quality Monitoring Results for Seeps, Aquifer Tubes, Substrate Porewater, and Nearshore River Water at the 100-H Area (from Hope and Peterson 1996b)

conclusion from the investigation was that there was no significant difference in the concentration of chromium in periphyton between reference samples collected upstream and those from the shoreline adjacent to the 100-N and 100-H Areas.

Periphyton and inorganic fractions were sampled from river cobble exposed along the shoreline at low river stage in September 2000. Chromium varied in concentration from ~90 to ~200 mg/kg for samples collected at the 100-B, 100-D, 100-H, and Hanford Townsite shorelines (Figure 3.7). The highest values occurred from the shore adjacent to the outfall spillway at 100-H Area. This site was frequently bathed in reactor coolant, which contained 700 μg/L chromium, during the operating years of H-Reactor (1949 to 1965). While these values appear to be slightly above expected "background" levels for sediment, no samples were collected at intervening locations, so it is premature to conclude that the elevated values are caused by groundwater plumes. Other heavy metals typical of upstream mining sources (e.g., copper, lead, and zinc) are also elevated in these samples.

3.2.3 Chromium in Riverbed Substrate Porewater

Field investigations to determine the chromium concentrations in substrate porewater from riverbed sediments were undertaken in the spring and fall of 1995 to support groundwater remediation decisions. The areas of interest included the 100-K, 100-D, and 100-H Areas because of the extent of chromium in groundwater at those areas. Special interest was directed at the 100-D and 100-H Areas because of their proximity to salmon spawning habitats. The highest chromium concentrations in substrate porewater were in the range ~100 to ~130 μg/L at a riverbed redd site adjacent to the former 107-D water retention basins and the 100-H outfall structure (see Figure 3.6). These concentrations are generally higher than concentrations observed in groundwater at near-river monitoring wells. This apparent discrepancy may be a consequence of the highest concentrations in the groundwater plume having already passed the near-river wells. Also, the specific conductance of the porewater samples indicated appreciable dilution by river water that had infiltrated the sediment. Chromium concentrations in the undiluted groundwater beneath the site could potentially be as much as 50% higher than the measured values, although the dilution relationship is not well defined.

At the 100-D Area in 1995, a previously undetected chromium plume was detected in the south-western region, with peak substrate porewater concentrations of ~630 μg/L at a riverbed site adjacent to the former 1907-DR process sewer outfall. These high values were confirmed by high values in samples from aquifer sampling tubes at the shoreline. Subsequent installation of monitoring wells in the area defined the extent of the plume (see Figure 3.4) and groundwater remediation efforts were started to prevent the plume from entering the river.

In September 1996, divers installed semi-permanent sampling tubes at sites previously sampled at the 100-K, 100-D, and 100-H Areas (Letter report, M. H. Sturges to A. J. Knepp, "Installation of Riverbed Pore Water Sampling Ports in the Columbia River at 100-D/DR, 100-H, and 100-K Reactor Areas," ERC Correspondence No. 044041, March 11, 1997, S. J. Hope, author). At each site, three tubes were installed to allow porewater sample collection from 0.15, 0.30, and 0.45 m (0.5, 1.0, and 1.5 ft) below the riverbed.

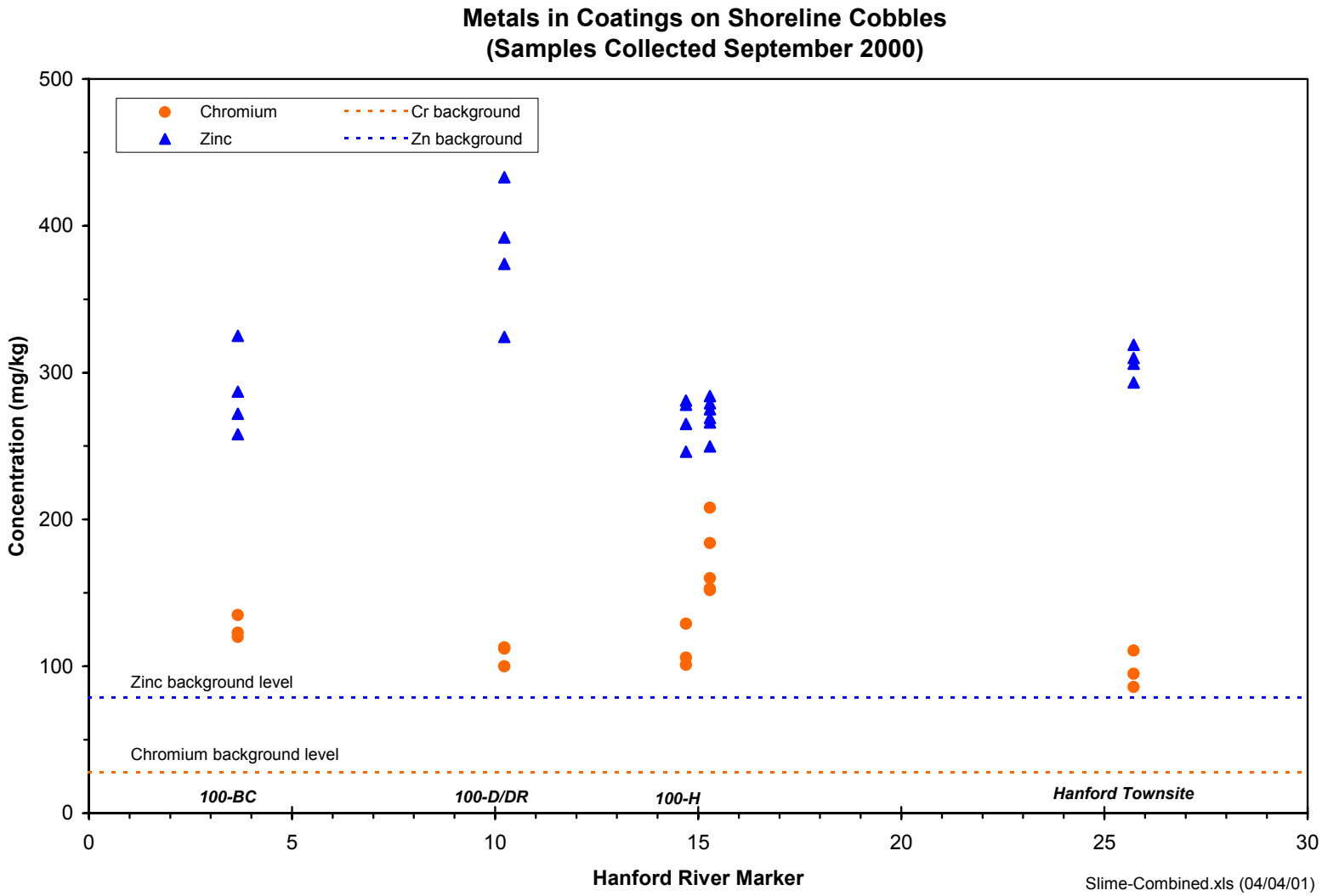

Figure 3.7. Chromium Concentrations in Sediment and Periphyton Samples Collected Along the Hanford Shoreline in September 2000

The observed concentrations were generally lower than those from the previous year's surveys at 100-H and 100-D. The highest value was ~100 μg/L from the same general riverbed area at 100-H where similar values were observed in 1996.

3.3 Toxicity of Chromium

As noted in Section 3.1, hexavalent chromium is the more toxic of the two forms present in the aquatic environment and is also the principal form present in nearshore environments of the Hanford Reach. Prior to the set of studies described in this document, there was little information on the toxicity of hexavalent chromium to chinook salmon. During early studies to evaluate the impact of Hanford operations on the Columbia River, Olson and Foster (1956) reported increased mortality for juvenile fall chinook salmon exposed to 77 and 180 μg/L chromium at 100 and 55 days post-hatch, respectively. Information on other fish species is limited. However, Eisler (1986) reported a 96 hr LC50 of 200 μg/L for salmon fingerings and 495 μg/L for rainbow trout eggs. The toxicity of chromium to aquatic species can vary by an order of magnitude or more, in some cases, depending on a variety of biological and physical factors. These factors include species differences, age or development state, temperature, pH, salinity, length of exposure, and interaction with other contaminants. Chromium is readily accumulated by aquatic organisms, but the range of bioconcentration factors reported for fish suggests it is not biomagnified in food chains.

4.0 Life History and Habitats of Fall Chinook Salmon

4.1 General Life History and Population Status

Fall chinook salmon are anadromous meaning they migrate to the ocean as juveniles, spend 1 to 5 years in salt water, then migrate upstream as adults to spawn in fresh water. Adult fall chinook salmon enter the mouth of the Columbia River in late summer to begin their upstream migration to spawning areas such as the Hanford Reach of the Columbia River. The loss of mainstem riverine habitat because of hydroelectric development throughout the Columbia River basin led to increased numbers of fall chinook salmon returning to spawn in the Hanford Reach in the early 1960s (following completion of McNary and Priest Rapids dams in 1954 and 1950, respectively).

Approximately 80% of the upriver bright fall chinook salmon migrating over McNary Dam (river kilometer 470) spawn in the Hanford Reach (Dauble and Watson 1997). Adult returns to the Hanford Reach ranged from ~21,000 to ~49,000 fish from 1970 to 1983; numbers peaked at over 90,000 fish in 1987. In recent years, the number of fall chinook salmon spawning in the Hanford Reach have averaged about 50,000 adults, while the average run size to the Snake River has declined to <1,500 adults (Dauble and Geist 2000). Redd counts have shown a general increase over time since surveys were initiated in 1948 (Figure 4.1).

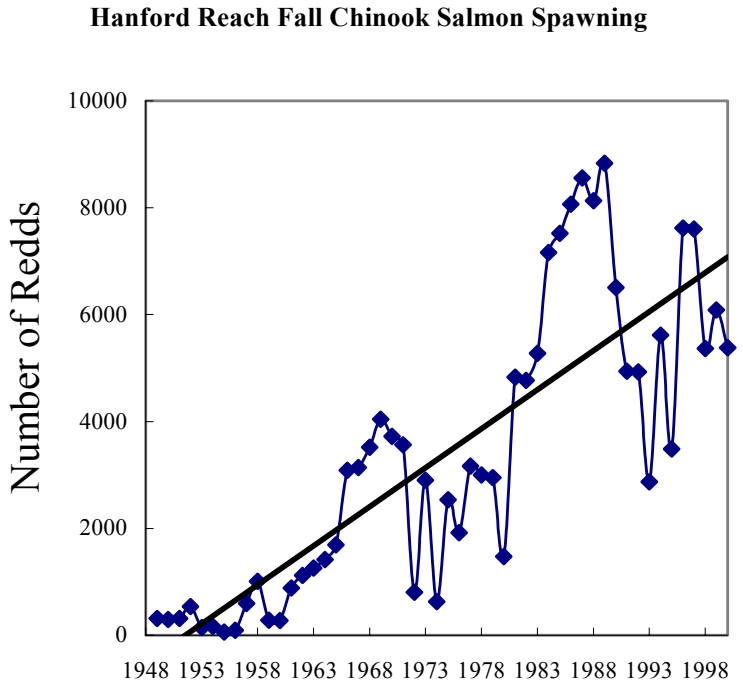

Figure 4.1. Peak Redd Counts for Fall Chinook Salmon in the Hanford Reach, 1949-1998

Dauble and Watson (1997) noted that fall chinook salmon spawned from mid-October to the third week in November in the Hanford Reach. Peak spawning, or the date of the highest redd count, ranged from October 26 to November 26. Mean water temperatures were 15.3°C and 12.5°C at first observed spawning and peak spawning, respectively.

4.2 Spawning Habitats

Fall chinook spawn throughout the Hanford Reach from about river kilometer 560 near Wooded Island to upper Vernita Bar at river kilometer 628 (Figure 4.2). Ten general spawning areas in the Hanford Reach are routinely monitored by aerial surveys (Figure 3.8; Dauble and Watson 1990). The areas with highest redd densities are Vernita Bar (Area 10) and Upper Locke Island (Area 5). These two locations contributed 33% and 25%, respectively, to the total number of redds counted from 1948 through 1992 (Dauble and Watson 1997). Vernita Bar is located just downstream of Priest Rapids Dam at river kilometer 628 and Upper Locke Island is located near the 100-H Area. The trend or relative proportion of redd counts at the 100-H Area follow the counts at the Vernita Bar from 1949 through 1998. For example, counts for the 100-H Area and the Vernita bar (high spawning areas) are compared to the 100-D Area and Ringold/Wooded Island (low density spawning areas).

Most spawning in the Hanford Reach is concentrated in braided sections of the main channel or wider reaches having complex channel formations. For example, 22 of 27 segments with channel bars had spawning activity (Dauble and Geist 2000). These channel bars represent areas where alluvium is

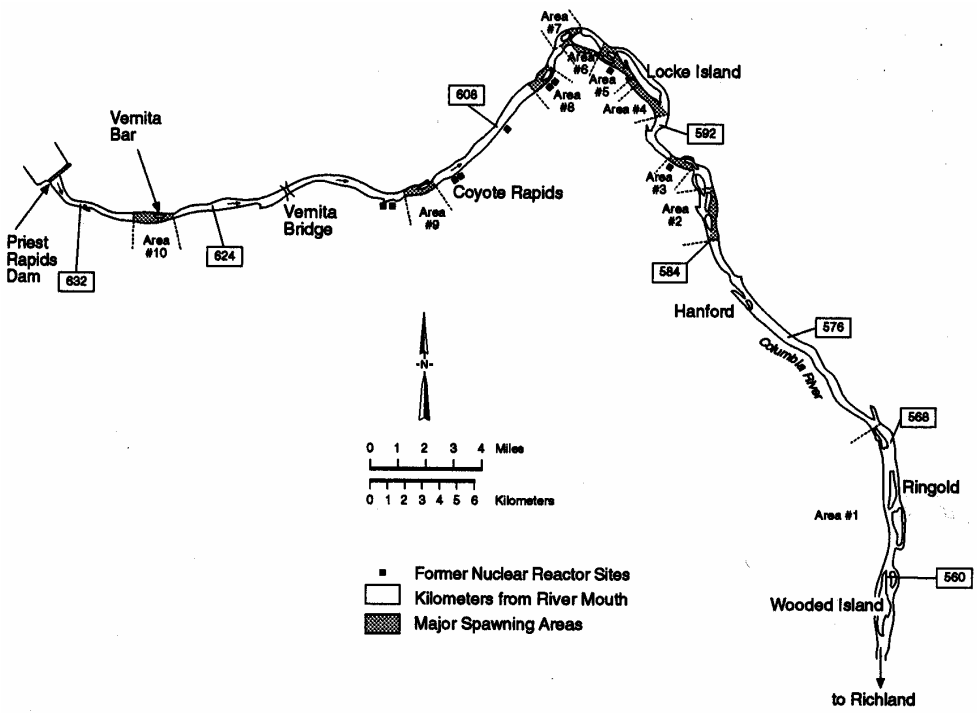

Figure 4.2. Location of Ten Major Fall Chinook Salmon Spawning Areas in the Hanford Reach of the Columbia River (from Dauble and Watson 1997)

deposited due to reduced bed gradient. Alluvium is highly porous, allowing river water to penetrate into the bed material (hyporheic habitat), creating conditions beneficial to both spawning and egg incubation (Figure 4.3).

Within major spawning areas, the distribution of individual redds is sometimes patchy (Figure 4.4). Redds exhibited a high fidelity and occurred in clusters at the spatial scale of a spawning area, but within each cluster they tended to be uniformly spaced (Geist et al. 2000). General characteristics of fall chinook salmon spawning areas in the Hanford Reach include: water depths of ~1 to 3 m, velocities ranging from 60 to 120 cm/sec, and dominant substrates of 7.6 to 15.2 cm. Spawning clusters, or areas of concentrated redds, occurred where the velocity was faster and riverbed flatter than in non-spawning areas. About 80% of the distribution of redd clusters could be explained based on water velocity and lateral slope of the river bottom (Geist et al. 2000).

Complex channel patterns associated with redd clusters create geomorphic bed formations that facilitate interstitial flow of mixed surface and groundwater. The proportion of surface water and groundwater in this mixing (i.e., hyporheic) zone is a function of the geomorphic bed forms, volume of regional groundwater discharge to the river, and the river stage (Geist and Dauble 1998). In fact, upwelling from the hyporheic zone into fall chinook salmon spawning areas was greater than upwelling

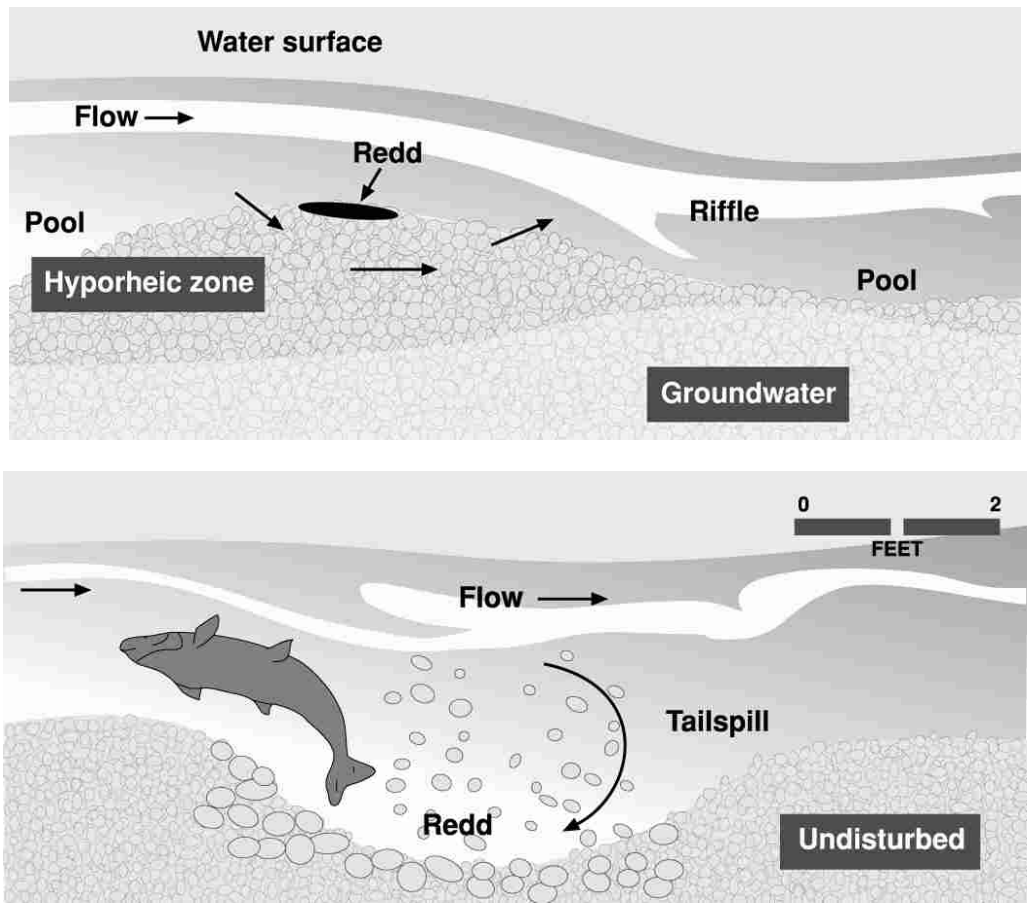

Figure 4.3. Schematic of Flow Patterns in a Spawning Area and Characteristics of a Redd

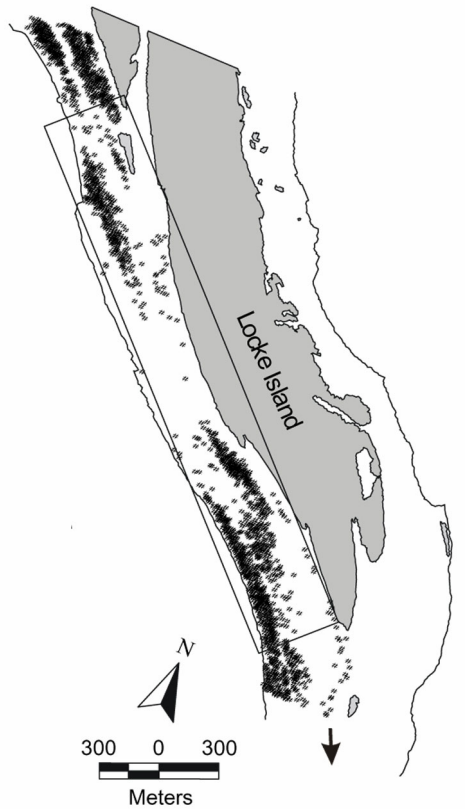

Figure 4.4. Distribution of Fall Chinook Salmon Redds within a Major Spawning Area (from Geist et al. 2000)

into non-spawning areas at two study sites in the Hanford Reach (Geist 2000). Furthermore, the upwelling into spawning areas was composed of a higher proportion of river water than upwelling in non-spawning areas. Hyporheic discharge zones composed of undiluted ground water or areas with little or no upwelling were not used by spawning salmon.

4.3 Rearing, Feeding, and Juvenile Migration

Salmon eggs deposited in Hanford Reach redds develop over the late fall and winter, with alevins or "fry" emerging from March through May (Chapman et al. 1983; Becker 1985). Although redds occur at a range of depths to 8 m, emergent fry move inshore and select backwater sloughs and shoreline embayments, which are slower in velocity and rich in food, for initial foraging and rearing. Dominant food organisms for fall chinook salmon fry during their freshwater rearing period in the Hanford Reach include both the larval and adult forms of midgefly and caddisfly (Becker 1973; Dauble et al. 1980; Rondorf et al. 1990).

Fall chinook salmon are the dominant salmonid found in the Hanford Reach during spring, with as many as 20-30 million fry produced annually from natural production (Washington Department of Fish and Wildlife [WDFW], unpublished data). Up to 8 million juvenile fall chinook salmon are released from the Priest Rapids Hatchery each year. During their period of residence in the Hanford Reach, fry can be

found in relatively high densities along shallow nearshore embayments, taking advantage of large cobble and rooted vegetation as cover. Larger outmigrants (i.e., spring chinook salmon, sockeye salmon, and steelhead smolts) tend to use the Hanford Reach primarily as a migration corridor to the lower Columbia River and spend a greater proportion of their time in offshore, deeper habitats (Dauble et al. 1989). Juvenile fall chinook salmon originating from the Hanford Reach migrate downstream in the first year of life, with most fish leaving the Hanford Reach by the end of July (Becker 1973).

5.0 Summary of Key Results from Trustee Studies

This section of the report summarizes the objectives, key findings, and uncertainties associated with recent toxicity studies conducted to determine the potential for chromium to adversely affect chinook salmon in the Hanford Reach of the Columbia River. To accomplish this activity, we reviewed the three primary technical documents produced under the direction of the Trustees: Farag et al. (2000), Patton et al. (2001), and Deloney et al. (2001). This section of the report is organized according to the type of exposure (e.g., life stage or response) involved in the toxicity test (Table 5.1).

5.1 Fertilization Tests

The objective of these tests was to determine the potential for chromium to adversely affect chinook salmon gametes and the fertilization process. Fertilization tests were conducted by testing both Snake River cutthroat trout (*O. clarki* spp) and chinook salmon. Fertilization of eggs was conducted by mixing sperm and ova in a solution of 1% sodium chloride for 1 min. The exposed and potentially fertilized eggs were then rinsed with "Hanford" experiment water containing chromium and allowed to water harden in simulated Hanford Reach water for 1.5 hr. Simulated Hanford Reach water was laboratory well water from the Jackson National Fish Hatchery adjusted to approximate Hanford Reach hardness. After water hardening, the test embryos were transferred to well water systems at either the Jackson National Fish Hatchery (cutthroat trout) or McNenny Hatchery. The eggs were evaluated for fertilization at 10 days by clearing in acetic acid to facilitate the observation of developing embryos.

5.1.1 Key Findings

Results of the fertilization tests showed no reduction of fertilization rates relative to controls treatments over a concentration range of 11 to 266 µg/L chromium. Control treatment fertilization was ~48% and 67% for cutthroat trout and chinook salmon, respectively.

Table 5.1. Summary of Toxicity Studies Evaluated for this Report, Including Exposure Scenarios and Endpoints Tested

Life Stage	Exposure Interval	Water Source	Test Endpoints
Fertilized eggs	1.5 hr	Simulated Columbia River	Fertilization rate
Eye egg to swim-up	83 days	Simulated Columbia River	Hatching success, growth, survival, tissue damage, lipid peroxidation, Cr accumulation
Eye egg to swim-up	83 days	Columbia River	Hatching success, growth, survival, Cr accumulation
Parr (60 days post-hatch)	134 days	Simulated Columbia River	Growth, tissue accumulation, lipid peroxidation, blood DNA, aberrations
Parr	40 min	Simulated Columbia River	Avoidance

5.1.2 Uncertainties Associated with the Fertilization Study

Chromium exposures only occurred from fertilization through water hardening. While additional water exchange is expected to be minimal following the period between water hardening and the start of the eyed-egg stage, this portion of the salmon life cycle was not investigated. The fertilization rates for control populations were lower than optimal; however, the study had sufficient rigor for the intended purpose.

One uncertainty concerning fertilization success is related to estimating the concentration of chromium that eggs and sperm could be exposed to in the Columbia River. Since fertilization occurs in the water column near the upper surface of the redd, the water concentrations of chromium would be diluted depending upon proximity to the groundwater plumes and volume of river discharge.

5.1.3 Conclusions

Results indicate that chromium from Hanford groundwater sources poses no risk to the natural fertilization process of fall chinook salmon.

5.2 Early Life History Studies

This section emphasizes the effects of chromium to hatching success, survival, growth, and survival of early life-stages (i.e., eyed eggs to swim-up stage) of fall chinook salmon. We also discuss results of chromium accumulation in tissues. Two different early life stage (ELS) studies were conducted in the laboratory to evaluate the effects of chromium on early development and survival of fall chinook salmon. The first study was conducted offsite using experimental water simulating conditions thought to be associated with redds in the Hanford Reach and a surrogate population of chinook salmon (Farag et al. 2000). A second study was conducted in Richland, Washington, using Hanford Site groundwater as the chromium source, Columbia River water, and fall chinook salmon stock originating from the Hanford Reach (Patton et al. 2001).

Specific objectives of the off-site ELS study were to: 1) determine the effects of chromium on chinook salmon egg survival, egg hatching and alevin survival, growth and behavioral development, and 2) evaluate the bioconcentration of chromium by chinook salmon and determine effects (of) chromium exposure on DNA strand breakage and lipid peroxidation. Five exposure concentrations (5 to 120 µg/L total chromium) were tested along with a control treatment at approximately 5°C. Exposures were conducted in reconstituted laboratory water adjusted to a hardness of 80 mg/L hardness simulating Columbia River water. The exposures were initiated at the eyed stage of development (day 0) and allowed to continue through the day of median swim up (day 83). At day 32 (median hatch), the newly hatched fry and unhatched embryos were released into the exposure aquaria. Test fish were monitored for an additional 30 days (through day 113) in clean reconstituted laboratory water. Fish were removed at median hatch (day 32), alevin stage (day 70), median swim up (day 83), and at test termination (day 113) for analysis of whole-body chromium residues and lipid peroxidation. Fish were also collected at medium hatch and termination for histological analysis.

The objective of the on-site ELS evaluation was to evaluate the effects of locally-adapted populations of hexavalent chromium from Hanford groundwater on ELS fall Chinook salmon. Exposure conditions included chromium from Hanford groundwater wells near the Columbia River and locally adapted populations of fall chinook salmon. Chromium exposure levels were increased because no significant difference in length or weight was observed between the highest test concentration of 120 μg/L and controls during the off-site test. On-site exposures ranged from 11 μg/L to 266 μg/L chromium. Exposures were from the eyed egg stage (day 0) through median swim up (day 98). At day 70 (median hatch occurred between day 41-47), the newly hatched fry were released from the eggcups into the exposure aquaria. Fish were then held in Columbia River water until the test was terminated on day 132. Whole-body analysis of tissue accumulation was conducted at median hatch, halfway between hatch and swim up (day 70), and at test termination. Fish from on-site tests were not assayed for lipid peroxidase or histology because available results from the off-site tests did not reveal effects due to chromium exposure.

5.2.1 Key Findings

Off-Site Study – Results of the off-site ELS evaluation (Farag et al. 2000) revealed no effect (neither a statistically significant difference between treatments nor a dose dependent response was observed) at any concentration for the following endpoints: percentage hatched, percentage viable hatched, and percentage hatched with deformities. Mean hatching success was ≥93% in all treatment groups. Overall survival at termination was ≥88% and was similar for all treatment groups. There were no statistically significant differences observed between treatments groups for behavior or development.

Other findings include:

- No significant difference in length between treatment and control for fish hatched and reared in water containing chromium up to 120 μg/L
- No gross lesions or necrosis were observed as a result of chromium exposures
- Fish in the two highest exposure concentrations showed a tendency to accumulate slightly higher whole body concentrations of chromium at the termination of the test (day 113) compared to the control and lower exposure concentrations.

On-Site Study – There were no significant differences in survival, growth, or development between treatment and control for fish hatched and reared in water containing chromium up to 266 μg/L. Survival was similar among all exposure groups and exceeded 98% at both swim up and termination (Patton et al. 2001). At swim up and termination, there were no statistically significant differences between survival of the control group and any exposure groups. There were no observable differences in behavior or development between exposure groups. There were also no statistical differences between the days required to reach median swim up for any exposure group compared to the control group. The dates to median swim up ranged from day 95 to day 100, with the overall mean occurring on day 98.

The 54, 120, and 266 μg/L treatment groups had slightly lower average length than the controls at test termination. However, mean fork lengths varied by less than 2% across all treatments and were not

statistically different. The 54, 120, 266 μg/L exposure groups had slightly lower average weights compared to the control group at day 132, but weights were not significantly different among treatment groups.

Analysis of tissue showed a systematic increase in chromium levels with increasing concentrations of aqueous chromium ≥22 μg/L. At hatch, the fish tissue concentrations of chromium were statistically higher than the control group for all exposure groups, except for the 11 μg/L group. All exposure groups were statistically elevated compared to the control at halfway between hatch and swim up, at swim up, and at termination. At test termination, there was a statistically significant decrease in chromium levels in fish tissue for the 266 μg/L and 54 μg/L exposure groups, compared to the tissue levels at swim up when the chromium exposures were ended. This response likely reflects both elimination and growth dilution of chromium due to water in the aquaria being restored to background levels of chromium following swim up (i.e., chromium exposures were stopped) to mimic conditions in the Columbia River following emergence of the juvenile salmon from the gravel substrate.

5.2.2 Uncertainties Associated with Off-Site and On-Site ELS Studies

We assumed two constructs to assess impacts in the context of an ELS toxicity test. First, differences in a response (e.g., death, growth reduction, etc.) should be statistically significant relative to control exposures. Second, the response should demonstrate dose dependence (i.e., as the exposure concentration increases, the magnitude of the response will also increase). It should be noted that the range of exposure concentrations for both tests were not high enough to result in measurable adverse effects of chromium exposures. Since dose dependent effects were not observed for most endpoints, the results could not be extrapolated to traditional toxicological parameter (e.g., lethal concentration resulting in a measured endpoint such as death or reduced growth). Additionally, there were sufficient inconsistencies in the tissue burden results for the off-site ELS studies to suggest inconsistencies with exposure regime, organism response, or tissue analysis.

5.2.3 Conclusions

Overall, the results of both studies reveal that salmon exposed to aqueous chromium to 266 μg/L during the eyed-egg to swim up portion of their life cycle were not adversely impacted. Hatching success, behavior, development, and survival at termination of the test were not affected by the chromium exposures in either evaluation. Subtle effects observed including slight numeric decrease in length and weight of the higher exposure groups at termination; however, these decreases were not statistically different. There was a dose dependent uptake of chromium observed for on-site tests with elevated tissue concentrations for chromium exposures ≥54 μg/L.

5.3 Fish Health Studies

Specific objectives of the fish health tests were to (1) gather data on chromium-induced physiological responses to aid in the interpretation of growth and survival responses documented during the early life stage test, and (2) interpret the physiological effects of chromium on fish and define parameters than can be measured on fish samples collected from the Hanford Reach. These studies were conducted in the

laboratory under conditions designed to simulate Hanford Reach conditions (Farag et al. 2000) and used a "weight-of-evidence" approach to interpret all the physiology data together. Impacts of chromium exposure to juvenile (parr stage or 60 days post-hatch at test initiation) chinook salmon were determined in a flow-through system over 134 days of exposure. Initial exposure concentrations were control, 24 and 54 μg/L. However, test treatments were increased to 120 and 266 μg/L, respectively, at day 105 of exposure because no apparent effect on growth or survival was evident at the initial exposure concentrations.

Exposures to chromium were terminated at day 134. Fish health parameters were measured on fish collected at 105 days and 134 days (i.e., the 24/120 and 54/266 μg/L exposures).

Fish health parameters evaluated included growth; whole body and organ chromium accumulation (histology of spleen, skin, gills, liver, kidney, and intestine); products of lipid peroxidation in selected tissues; chromium accumulation in selected organs; and aberrations in blood DNA. To help assess the information presented in the health effects study, a set of conditions was established and applied sequentially to evaluate each end point in the health effects study.

1. Was there an observed effect (i.e., a difference in the exposed fish compared to the control treatments)? In this caveat, one or both of the exposure levels could be different than the control, however, one could be less than the control response and the other could be greater.

2. If there was an observed effect, was the type of response in the exposed fish treatments in agreement (i.e., both treatments showed the same response)?

3. If the two treatments agreed in type of response, was the observed response dose dependent (i.e., the response for the higher exposure concentration was greater than the response for the lower exposure concentration, both exposure responses were greater than control response)?

4. Where the observations of the initial phase (day 105) of the test consistent with observations in the second phase of the test (day 134)?

5. Did the observed effects appear to correlate with tissue burdens of chromium?

5.3.1 Key Findings

In this section, we summarize results relative to measured endpoints.

Growth – At day 105, there was no observable effect on survival or growth (length and weight) between the chromium-exposed fish (i.e., 24 and 54 μg/L) and the controls. At this time, the two treatment concentrations were increased to 120 and 266 μg/L chromium for an additional 29 days. Following the extended exposure period, fish weights were reduced in the 24/120 treatment and survival was reduced in the 54/266 exposure treatment.

Accumulation of Chromium – There was no statistically significant accumulation of chromium at 105 days for the 24 μg/L exposure; however, mean chromium concentrations were numerically higher

than control fish by factors ranging from 1.7 to 4.4 in gill, kidney, liver, and whole body samples. Chromium levels in gill, kidney, and whole body measurements ranged from 3.5 to 5.8 higher in the 54 μg/L treatment relative to controls at day 105.

Measured concentrations of chromium in control fish were lower at day 134 than they were at day 105 ranging from factors of 0.5 (pyloric caeca) to 7.5 (whole body). In general, tissue burdens increased to levels roughly ten times that of the control treatment in both exposures (24/120 and 54/266). This suggested that accumulation of chromium may have peaked and that mechanisms responsible for the accumulation of chromium were saturated at exposures of 120 μg/L and greater for gill, kidney, and whole body measurements. In contrast, liver and pyloric caeca measurements did not show an accumulation plateau.

Mean concentrations (about 30 μg/g) of chromium observed in the gill and kidney of fish from the 24/120 exposure were not associated with a significant decrease in weight. Additionally, the decrease in survival of chromium-exposed fish at 54/266 treatment corresponded with elevated chromium in gill, liver, and pyloric caeca. Maximum tissue concentrations were 35 μg/g and 30 μg/g for gill and kidney tissue, respectively.

Histology – Gross pathology observations of whole organs indicated some chromium-exposed fish had abnormal coloration and markings in kidney, while no such change was noted for controls. With consideration given to duration of exposure and concentrations of chromium, the response occurred at low frequency in fish at 54 μg/L (2% of test fish after 105 days exposure), in addition to the 24/120 μg/L (9% of test fish), and 54/266 μg/L (19% of test fish) treatments. The discoloration was observed in fish where concentrations of chromium in kidneys ranged from 25.6 to 30.2 μg/g in a dose-dependent trend.

There was some indication of a decrease in hematopoietic cells at day 134 for both exposure treatments, but the effect was not dose dependent. Cell death, fibrosis, and dilation of the lumen was also observed at 134 days in the chromium exposed fish, but only dilation of the lumen appeared to be dose dependent. The effects at day 134 were associated with chromium concentrations in kidney tissue of 30 μg/g. None of these effects were noted at day 105, where kidney concentrations were 25 μg/g chromium, suggesting extended duration of exposure and increased concentration of chromium contributed to histological effects.

Observations of the gill indicated that lamellar hypertrophy and apoptosis (cell death) of chloride cells occurred with variable and non-dose dependent relationships. There were indications of increased chloride cell death at day 134 for both chromium exposure groups, but the response was non-dose dependent at day 105 and appeared to be infrequent (raw data was not available for review). At day 105, there was no discernable pattern of lamellar hypertrophy of apoptosis. Gill chromium concentrations show a clear dose dependent increase at day 105 and day 134. The frequency of both endpoints in gills was low and the concentrations of chromium in gills where these effects occurred were about 30 μg/g in the 24/120 μg/g treatments at day 134.

No adverse effects were observed in spleen histology evaluations at day 105 for either exposure group. There was some indication of mild blood vessel congestion in spleen at day 134 in the 24/120

exposures, but there was no dose-dependency. There was an apparent and slight impact on red blood cells as indicated by the dose dependent increase in eosinophillic spherules at day 134. Chromium concentrations in spleen tissue were not reported.

There was indication of an accumulation of lipid (lipidosis) in livers of exposed fish at day 105, but observations were variable at day 134 and indicated a decrease relative to the control treatment. Tissue concentrations of chromium were non-dose dependent at day 105, but were dose dependent at day 134. No clear pattern of effects relative to chromium tissue concentrations was apparent for glycogen depletion in livers.

Lipid Peroxidation – Elevated lipid peroxidation was noted in the kidneys of fish following exposures to 24/120 and 54/266. Investigators did not collect enough tissue to define these differences statistically (only one measurement per treatment was taken) and no measurements were taken for the 24 and 54 μg/L treatments.

Blood DNA – The results of the DNA assessments in red blood cells were difficult to interpret. Results were expressed as the difference in the coefficient of variation between the fish blood samples and a reference blood sample (CV DIF). The control treatments indicated a sharp drop in CV DIF from day 105 to day 134. The basis for this change in the controls was not clear, (i.e., was it due to some developmental process or natural variation).

Test results at day 105 were dose-dependent. A statistically significant decrease in the mean CV DIF values was observed at each increasing exposure concentration. The interpretation of this data was that the decrease in CV DIF values (up to 36%) corresponded with an increase in variation of DNA content in red blood cells. The increased variation may be indicative of a greater number of DNA fragments in the populations of red blood cells resulting from the preceding 105 days of exposure. However, results at day 134 contradicted the initial set of data. Not only did the control response drop from 1.73 to 0.411 over 29 days, but the mean CV DIF values significantly increased by up to 23% with increasing exposure, indicating a decrease in DNA fragmentation in red blood cells.

5.3.2 Uncertainties Associated with Fish Health Studies

There were several uncertainties associated with the fish health studies, in particular, the blood DNA studies.

1. Could the difference in response be related to hematopoiesis (i.e., were the fish producing more red blood cells at one part of the test and than at the other)? What was the average age of the red blood cells in relation to the CV DIF values?

2. Although adaptation to chromium exposure may be a factor in the results of blood DNA studies, what is the reason for the shift in response for control treatments?

3. Why was no significant DNA strand breakage in liver and gill tissue observed with gel electrophoresis at either day 105 or day 134?

4. What stands out from the fish health study is that statistical differences were observed for both treatments relative to the control response. However, it is not clear what the biological significance is relative to exposure conditions in the Columbia River because the test interval extended >30 days past the expected residence time of an alevin in a redd.

5.3.3 Conclusions

A dose-dependent response for selected fish health endpoints was corroborated with tissue concentrations. Although chinook salmon parr accumulated chromium, no effect on growth and survival was noted at day 105 for 24 μg/L and 54 μg/L exposures. However, both growth and survival were significantly reduced for fish when exposures were extended 29 additional days at 24/120 μg/L and 54/266 μg/L. Gross pathology observations of kidney tissue suggested possible neurosis of tissue and associated discoloration at higher doses (266 μg/g) and over extended (>105 days) exposures. However, observations of gill, spleen, and liver had no clear response pattern. Assessments of blood DNA for chromium-exposed fish were contradictory between day 105 and 134 suggesting that use of this endpoint as a measure of toxicological response needs further refinement.

Using the "weight-of-evidence" approach, fish health studies showed that DNA damage, lipid peroxidation, and necrosis of the kidney cells occurred simultaneously as a result of chromium exposure. These malfunctions were associated with changes in growth and survival for chromium-exposed fish following the extended exposure period of 134 days.

5.4 Off-Site Avoidance-Preference Behavior Studies

The objective of the behavioral studies was to determine if juvenile chinook salmon would exhibit an avoidance or preference response to chromium under a test scenario representative of conditions present in the Hanford Reach of the Columbia River. To accomplish this, juvenile chinook salmon were exposed to aqueous chromium using two different water sources: (1) 80 mg/L hardness as $CaCO_3$ (i.e., representative of average surface water conditions), and (2) 200 mg/L hardness as $CaCO_3$ (i.e., representative of a undiluted groundwater source). Avoidance-preference experiments were conducted using a counter-current apparatus similar to that used by Sprague (1968). In brief, the experimental chamber produced a steep, central gradient between the control and test treatment. Behavior trials consisted of a 40-min acclimation period followed by a 40-min test period during which only the final 20-min interval was selected as the observation period. Behavioral response data was recorded as the proportion of time spent in the test solution versus time spent in the control solutions. The frequency of gradient crossing by fish (i.e., number of trips) and the residence time per gradient crossing (trip time) was also recorded.

5.4.1 Key Findings

Results of avoidance-preference trials, under conditions of 80 mg/L hardness, indicated that juvenile chinook salmon are capable of detecting and avoiding chromium concentrations ≥54 μg/L. However, chinook salmon neither avoided nor showed preference for aqueous chromium at concentrations ranging from 0 to 266 μg/L under conditions of 200 mg/L hardness.

5.4.2 Uncertainties Associated with the Behavioral Studies

The primary uncertainty related to the results is whether the laboratory studies realistically represent an exposure scenario expected to occur in the environment. For example, was the 20-min test period sufficiently long to measure for habituation of response? Would other factors influence fish behavioral response in the river? If we accept the premise that the test design was comprehensive, the issue becomes whether detection and avoidance of a potentially negative condition is okay (reduces the risk of adverse exposure) or not (excludes the fish from preferred habitat). It is not known if available spawning or rearing habitat is a limiting factor for production of fall chinook salmon in the Hanford Reach.

5.4.3 Summary

Information from these studies may be useful for specific locations where detailed information on contaminant plumes and microhabitat (such as river flow, geomorphology, hyporheic flow, and habitat use) are known. Avoidance-preference tests with juvenile salmon cannot be extrapolated to other life stages (e.g., adults) because of inherent differences in habitat selection, physiological condition, and ecological requirements.

6.0 Assessment of Risk to Fall Chinook Salmon in the Hanford Reach

We start our assessment by summarizing what is known about distribution and concentration of chromium in Hanford groundwater. Next, we describe pathways and environmental factors influencing movement of contaminated groundwater to the Columbia River. Finally, we discuss the potential for chromium from Hanford groundwater sources to adversely impact chinook salmon based on their life history and habitat use. Table 6.1 provides a summary of factors known to influence the exposure risk of chinook salmon to chromium.

6.1 Distribution and Concentration of Chromium in Hanford Groundwater

The distribution of chromium in groundwater plumes near the Hanford Reach shoreline is limited to a stretch from about river kilometer 600 to river kilometer 612. Within this distance, known chromium plumes are localized at three former reactor areas: 100-D, 100-H, and 100-K. To provide a perspective on spatial extent, we estimated the total amount of shoreline encroached on by plumes is ~3.4 km and ~1.5 km based on the ≥50 μg/L and ≥100 μg/L gradients, respectively (using the 2000 data shown in Figures 3.2-3.4). This distance can be compared to a total shoreline distance for the Hanford Reach (including islands) estimated at 355 km (Battelle and USGS 1999). The flux or movement rate of these specific plumes into the river is unknown. However, total groundwater movement into the Hanford Reach is estimated at 0.77 m^3/sec relative to an average discharge of 3,400 m^3/sec for surface water (Poston et al. 2000) or a dilution factor of ~4.4 x 10^3.

Table 6.1. Summary of Environmental Factors Influencing the Relative Risk of Fall Chinook Salmon Populations to Chromium from Past Hanford Operations

Primary controlling factors:
- distribution and extent of contaminant plume(s)
- relative concentration of contaminant plume(s)
- proximity of salmon spawning/rearing areas to contaminant plume(s)

Other environmental factors that reduce or mitigate exposure:
- dilution by groundwater, including hyporheic flows, prior to organism exposure
- dilution by the Columbia River
- adsorption of chromium to fine sediments (reduces bioavailability to aquatic organisms)

Biological factors that influence exposure:
- development stage of juvenile salmon (sensitivity and uptake rates)
- residence time of salmon in affected environment (exposure duration)
- degradation of hexavalent chromium by microorganisms
- hyporheic discharge zones of undiluted groundwater or areas with little upwelling are not used by spawning salmon

Concentrations of chromium in porewater (i.e., in nearshore areas where chromium was measured in riverbed substrate using piezometers, ranged to 630 μg/L). The frequency of high values (i.e., >100 μg/L) was low with most measurements showing non-detection of chromium. Collectively, these studies show that the upper limits of source terms for chromium (i.e., rate and spatial and temporal pattern of release (after Suter 1993) are fairly well defined.

6.2 Factors Influencing Concentration and Movement of Chromium in the River

Actual exposure of fall chinook salmon to chromium from Hanford Site groundwater is also influenced by processes occurring at the groundwater/river interface. During the 1990's, estimates for concentrations of chromium in the nearshore river environment have relied on observations from near-river monitoring wells. Predictions of effect and risk assessments that rely on such data are conservative because contaminants are diluted in the zone of groundwater/river interaction and in pore fluids of riverbed sediments. For example, there is evidence that some dilution of contamination occurs before groundwater passes into the free-flowing stream of the river (e.g., Peterson and Johnson 1992). Based on historical information, the record-of-decision for the interim remedial actions addressing chromium contamination in groundwater adopted a dilution factor of 50% for setting target compliance concentrations (EPA 1996). That is, the target is to reduce chromium concentrations in groundwater near the river to 22 μg/L or lower; 22 μg/L is twice the federal standard of 10 μg/L for protecting freshwater aquatic organisms.

A more recent analysis of monitoring data was completed under the Science and Technology effort of the groundwater/vadose zone project and confirms the generalization that riverbank seepage is typically composed of equal parts groundwater and river water (Peterson and Connelly 2001). However, the variability in the degree of mixing is very high, as a consequence of the numerous factors associated with the interaction between groundwater and river water that infiltrates the banks during periods of high river stage. If riverbank seepage is monitored continuously during low river stage, its composition may vary from primarily river water to nearly pure groundwater, depending on location. As described in that report, the principal variables are:

- Volume flux of approaching groundwater.
- Volume flux of infiltrating river water.
- Hydraulic properties and preferential pathways in the zone of interaction.
- Amplitude and duration of river stage cycle.
- Position of habitat of interest relative to the shoreline.

Reduction in contaminant concentrations occurs not only as a consequence of river water infiltrating the banks, but also because of river water that becomes entrained in the riverbed gravels. This entrainment process has been recognized as an important characteristic of salmon spawning habitat (Geist

2000). The relative contributions of (a) entrained river water flowing parallel to the channel, and (b) bank storage water returning perpendicular to the channel, on contaminant concentrations in riverbed porewater, has not been determined.

In addition to the physical process of mixing and contaminant dilution, biological and chemical processes operating in the zone of interaction can reduce the concentration and amount of chromium entering the river. Reduction of hexavalent chromium to trivalent chromium at the interface has been studied and results suggest that most, if not all, hexavalent chromium approaching the river in groundwater enters the free-flowing stream as hexavalent chromium (Thornton et al. 1995). Hexavalent chromium could enter the food chain via microorganisms, but biological uptake has not been demonstrated to be a significant process with regard to mass transport of chromium. Research activities are underway to investigate the role of microorganisms on contaminant behavior in the zone of interaction (Fredrickson and Geist 2001).

6.3 Relationship of Exposure Pathways to Life History, Behavior, and Habitat Use of Fall Chinook Salmon

One important part of any exposure assessment is to develop estimates of the duration of exposure. Knowledge of this variable is critical when attempting to link the results of laboratory studies to real-world effects, particularly when causal inferences are based on concentration-response relationships (after Rand and Petrocelli 1985). Exposure scenarios for fall chinook salmon in the Hanford Reach vary by life stage because of differences in their habitat use and residence time. Fall chinook salmon are anadromous which means they enter the Hanford Reach in late summer, spawn in the fall, then die shortly afterward. Because adults hold in off-shore areas, primarily in deepwater habitats, there is no evidence indicating they would be exposed to chromium from groundwater plumes.

In contrast, eggs incubate in the gravel-cobble substrate of the river bottom (redds) and hatch there in late winter (~January). The young alevins develop and subsist on their yolk until they emerge from redds in early March (Becker 1973, 1985). It is during this development stage that exposure to groundwater contaminants is of principal concern (Geist et al. 1994). After emerging from redds, subyearling fall chinook salmon (or “fry”) rear in shallow nearshore areas for 2 to 3 months before migrating to the Pacific Ocean (Dauble et al. 1989; Becker 1973). This behavior limits potential exposure to chromium from Hanford groundwater to the relatively short freshwater residence period.

Upwelling characteristics may provide cues that adult fall chinook salmon use to locate spawning habitat. This assumption is based on studies of other salmonids that use hyporheic and/or groundwater discharge to locate spawning areas (Witzel and MacCrimmon 1983; Curry and Noakes 1995). The implications with respect to contaminant exposure is that where salmon are presently spawning, there may be a greater likelihood that groundwater is more diluted in the interstitial zone by surface water than in areas that do not support spawning. This observation provides a plausible argument for hydrological mitigation of exposure in areas where salmon spawning habitat and groundwater contamination overlaps. For example, avoidance of chromium by adult salmon could result in reduced availability of spawning

habitat or poor reproductive success if spawning habitat was limited and fish were forced to spawn in high density areas. However, there is no evidence that spawning habitat is limited at current escapement levels (Dauble and Watson 1997).

Although the frequency of occurrence for porewater samples >11 μg/L chromium was 19% of the total at the 100-D/DR shoreline (Hope and Peterson 1996a), only one site occurred in the vicinity of a known spawning area (Dauble et al. 1997; Geist et al. 1997). Similarly, only 1 of 17 porewater sites near the 100-H Area, where more extensive spawning occurs, contained relatively high concentrations of chromium (Hope and Peterson 1996b). The amount of overlap of the chromium plume with known spawning areas of fall chinook salmon indicates the risk of exposure is low. More comprehensive substrate mapping of areas adjacent to contaminant plumes would be necessary to assess habitat suitability.

A recent study by Geist (2000) showed that fall chinook salmon spawned predominantly in areas of the Hanford Reach where hyporheic water discharged into the river channel. Hyporheic discharge zones composed of undiluted groundwater or areas with little or no upwelling were not used by spawning salmon (Geist 2000). Chromium that migrates to the river from groundwater is diluted extensively as it mixes with river water. Indeed, hexavalent chromium was not detected in the river column when samples were taken 2.5 cm above the substrate of porewater where chromium was present (Hope and Peterson 1996a). These data indicate that juvenile fall chinook salmon are unlikely to be exposed to elevated concentrations of chromium after they emerge from the redds and begin to rear along the shoreline.

Avoidance behavior could be considered either beneficial to survival, i.e. fish detect and move away from contaminant concentrations that are harmful; or injurious, if habitat is limited and fish are forced to use suboptimum conditions. For laboratory tests conducted at 80 mg/L hardness, chromium avoidance occurred far below concentrations reported to be acutely lethal for salmonids. For example, the literature reviewed in Deloney et al.(2001) suggested acutely toxic thresholds (i.e., 4-day exposures resulting in mortality to 50% of the test population) did not occur until chromium concentrations were in the range of 500-1,000 times greater than those resulting in avoidance. Chromium concentrations avoided by juvenile chinook salmon were similar to values shown to result in significant tissue accumulation (Patton et al. 2001), reduced growth (Olson and Foster 1956), or physiological impairment (Farag et al. 2000) of ELSs during extended exposures (i.e., greater than 60-120 days). Collectively, these studies indicate avoidance behavior could reduce exposure of juvenile salmon to harmful levels of chromium. There is no evidence suggesting that rearing habitat is limiting to production of fall chinook salmon.

When tested at 200 mg/L hardness, juvenile chinook salmon did not exhibit avoidance behavior at ≤266 μg/L chromium. The implications of these results to conditions in the Hanford Reach are unclear because river water hardness typically ranges from 60 to 80 mg/L. Deloney et al. (2001) also speculated that discharge of contaminated groundwater through redds could potentially elicit an avoidance response of developing alevins that could either alter the timing and maturity of juvenile salmon at emergence or disrupt the imprinting process associated with homing. We found no literature to support or dispute this conjecture.

7.0 Conclusions

Recent laboratory studies provided new information on exposure duration and toxicological response, tissue disposition and uptake rate, and behavioral response of juvenile chinook salmon to hexavalent chromium. Results showed that developing embryos would not be harmed by chromium concentrations expected to occur in substrate used for spawning. The life stage with greatest risk to chromium exposures would be alevins that remain in the gravel past the swim-up stage, since this is a life stage where rapid uptake occurs. Parr-size chinook salmon had elevated tissue burdens following extended exposure to chromium at ≥124 μg/L, and consequently, showed evidence of adverse physiological effects. However, these exposure durations (i.e., 109 days or more) are not realistic, i.e., the time spent by a juvenile chinook salmon at any one location in the Hanford Reach would limit exposure to a period of hours or days, not weeks. Additionally, groundwater seeps are localized relative to total shoreline used for rearing. There is no risk to adult salmon because they hold in deep water areas during their pre-spawning period, and chromium concentrations in the water column are not elevated above background. Potential effects of chromium exposure during the fertilization process are not a concern based on results of laboratory tests.

If we accept the premise that juvenile fall chinook salmon emerge from gravel substrate at median swimup and migrate to nearshore areas to rear soon after, the potential for adverse effects of chromium to salmon populations in the Hanford Reach is low due to the following risk reduction factors:

- known chromium plumes intersect only at the margins of a relatively small percentage (less than 3%) of known spawning areas based on lined distance of shoreline

- groundwater plumes are rapidly diluted by river water (within both hyporheic and surface water habitats), and chromium is not detectable in river water

- residence time of juvenile salmon in nearshore groundwater seep environments is short due to river level fluctuations and life history requirements for migrating downstream in early summer.

Data from the three reports reviewed for this evaluation, site-characterization data for groundwater, life history characteristics of salmon, and results of an outside peer review process support the conclusion that current cleanup criteria of 10 μg/L chromium is adequate for protecting fall chinook salmon populations.

8.0 References

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Appendix A

Summary of Recent Chromium Data for Groundwater Near the Columbia River

Table A.1. Summary of Recent Chromium Data for Groundwater Near the Columbia River

Shoreline Segment	Well Name	Constituent	Method	Filt?	Average (µg/L)	Minimum (µg/L)	Maximum (µg/L)	No. Results	No. Detects
100-B Area	199-B2-13	Total chromium (filt.)	ICP metals	Y	23	23	23	1	1
100-B Area	199-B3-1	Total chromium (filt.)	ICP metals	Y	20	20	20	1	1
100-B Area	199-B3-46	Total chromium (filt.)	ICP metals	Y	10	10	10	1	1
100-B Area	199-B3-47	Hexavalent chromium	Hach spectro	Y	70	70	70	1	1
100-B Area	199-B3-47	Total chromium (filt.)	ICP metals	Y	77	77	77	1	1
100-B Area	699-72-92	Total chromium (filt.)	ICP metals	Y	5	5	5	1	1
100-K Area	199-K-18	Hexavalent chromium	Hach spectro	Y	80	60	90	22	22
100-K Area	199-K-18	Total chromium (filt.)	ICP metals	Y	70	58	79	4	4
100-K Area	199-K-19	Hexavalent chromium	Hach spectro	Y	90	80	100	5	5
100-K Area	199-K-19	Total chromium (filt.)	ICP metals	Y	91	87	98	3	3
100-K Area	199-K-20	Hexavalent chromium	Hach spectro	Y	110	90	120	17	17
100-K Area	199-K-20	Total chromium (filt.)	ICP metals	Y	104	101	108	4	4
100-K Area	199-K-21	Hexavalent chromium	Hach spectro	Y	30	10	50	2	2
100-K Area	199-K-21	Total chromium (filt.)	ICP metals	Y	53	53	53	2	1
100-K Area	199-K-22	Hexavalent chromium	Hach spectro	Y	160	150	170	4	4
100-K Area	199-K-22	Total chromium (filt.)	ICP metals	Y	153	139	166	2	2
100-K Area	199-K-31	Total chromium (filt.)	ICP metals	Y	13	12	15	2	2
100-K Area	199-K-32A	Total chromium (filt.)	ICP metals	Y	25	23	28	2	2
100-K Area	199-K-33	Total chromium (filt.)	ICP metals	Y	16	16	17	4	4
100-K Area	199-K-37	Hexavalent chromium	Hach spectro	Y	70	70	80	4	4
100-K Area	199-K-37	Total chromium (filt.)	ICP metals	Y	67	55	74	3	3
100-K Area	199-K-114A	Hexavalent chromium	Hach spectro	Y	100	50	130	16	16
100-K Area	199-K-117A	Hexavalent chromium	Hach spectro	Y	40	10	100	51	51
100-K Area	199-K-117A	Total chromium (filt.)	ICP metals	Y	34	31	37	2	2
100-K Area	199-K-126	Hexavalent chromium	Hach spectro	Y	80	50	120	11	11
100-N Area	199-N-2	Total chromium (filt.)	ICP metals	Y	7	6	7	4	3
100-N Area	199-N-3	Total chromium (filt.)	ICP metals	Y	5	3	8	4	3
100-N Area	199-N-14	Total chromium (filt.)	ICP metals	Y	5	5	5	4	2
100-N Area	199-N-18	Total chromium (filt.)	ICP metals	Y				2	0
100-N Area	199-N-19	Total chromium (filt.)	ICP metals	Y				1	0
100-N Area	199-N-21	Total chromium (filt.)	ICP metals	Y	3	3	3	1	1
100-N Area	199-N-26	Total chromium (filt.)	ICP metals	Y				1	0
100-N Area	199-N-47	Total chromium (filt.)	ICP metals	Y				1	0
100-N Area	199-N-54	Total chromium (filt.)	ICP metals	Y	4	4	4	1	1
100-N Area	199-N-67	Total chromium (filt.)	ICP metals	Y	7	7	7	2	1
100-N Area	199-N-75	Total chromium (filt.)	ICP metals	Y	3	3	3	1	1
100-N Area	199-N-76	Total chromium (filt.)	ICP metals	Y	2	2	2	2	1
100-N Area	199-N-92A	Total chromium (filt.)	ICP metals	Y	5	4	5	2	2
100-N Area	199-N-96A	Total chromium (filt.)	ICP metals	Y	5	5	5	4	2
100-N Area	199-N-99A	Total chromium (filt.)	ICP metals	Y	4	4	4	2	1
100-N Area	199-N-105A	Total chromium (filt.)	ICP metals	Y				1	0
100-D Area	199-D3-2	Hexavalent chromium	Hach spectro	Y	20	20	30	3	3
100-D Area	199-D3-2	Total chromium (filt.)	ICP metals	Y	26	22	30	2	2
100-D Area	199-D4-1	Total chromium (filt.)	ICP metals	Y	15	15	15	2	1
100-D Area	199-D4-13	Hexavalent chromium	Hach spectro	Y	590	530	640	4	4
100-D Area	199-D4-13	Total chromium (filt.)	ICP metals	Y	539	495	583	2	2
100-D Area	199-D4-14	Hexavalent chromium	Hach spectro	Y	580	490	670	2	2
100-D Area	199-D4-14	Total chromium (filt.)	ICP metals	Y	597	434	759	2	2

Shoreline Segment	Well Name	Constituent	Method	Filt?	Average (µg/L)	Minimum (µg/L)	Maximum (µg/L)	No. Results	No. Detects
100-D Area	199-D4-19	Hexavalent chromium	Hach spectro	Y	460	440	480	3	3
100-D Area	199-D4-19	Total chromium (filt.)	ICP metals	Y	456	442	480	3	3
100-D Area	199-D5-20	Hexavalent chromium	Hach spectro	Y	140	90	180	4	4
100-D Area	199-D5-20	Total chromium (filt.)	ICP metals	Y	158	56	259	2	2
100-D Area	199-D5-36	Hexavalent chromium	Hach spectro	Y	10	10	10	3	2
100-D Area	199-D5-36	Total chromium (filt.)	ICP metals	Y	1	1	1	2	1
100-D Area	199-D5-37	Hexavalent chromium	Hach spectro	Y	50	40	70	3	3
100-D Area	199-D5-37	Total chromium (filt.)	ICP metals	Y	76	37	115	2	2
100-D Area	199-D5-44	Hexavalent chromium	Hach spectro	Y				4	0
100-D Area	199-D5-44	Total chromium (filt.)	ICP metals	Y	1	1	1	2	1
100-D Area	199-D8-4	Total chromium (filt.)	ICP metals	Y	136	130	141	2	2
100-D Area	199-D8-5	Total chromium (filt.)	ICP metals	Y	309	294	320	3	3
100-D Area	199-D8-6	Total chromium (filt.)	ICP metals	Y	95	95	95	1	1
100-D Area	199-D8-55	Total chromium (filt.)	ICP metals	Y	62	62	62	1	1
100-D Area	199-D8-68	Hexavalent chromium	Hach spectro	Y	180	20	390	17	17
100-D Area	199-D8-68	Total chromium (filt.)	ICP metals	Y	349	349	349	2	2
100-D Area	199-D8-69	Hexavalent chromium	Hach spectro	Y	80	30	160	16	16
100-D Area	699-97-51A	Total chromium (filt.)	ICP metals	Y	43	42	44	2	2
100-H Area	199-H4-3	Hexavalent chromium	Hach spectro	Y	140	110	150	5	5
100-H Area	199-H4-3	Total chromium (filt.)	ICP metals	Y	127	110	142	3	3
100-H Area	199-H4-4	Hexavalent chromium	Hach spectro	Y	80	10	170	13	13
100-H Area	199-H4-4	Total chromium (filt.)	ICP metals	Y	114	12	171	4	4
100-H Area	199-H4-5	Hexavalent chromium	Hach spectro	Y	70	60	80	13	13
100-H Area	199-H4-5	Total chromium (filt.)	ICP metals	Y	81	78	85	3	3
100-H Area	199-H4-8	Hexavalent chromium	Hach spectro	Y	50	40	60	4	4
100-H Area	199-H4-8	Total chromium (filt.)	ICP metals	Y	38	38	38	1	1
100-H Area	199-H4-9	Total chromium (filt.)	ICP metals	Y	111	107	115	2	2
100-H Area	199-H4-10	Hexavalent chromium	Hach spectro	Y	30	20	30	3	3
100-H Area	199-H4-10	Total chromium (filt.)	ICP metals	Y	23	19	27	2	2
100-H Area	199-H4-12A	Total chromium (filt.)	ICP metals	Y	54	45	64	2	2
100-H Area	199-H4-13	Hexavalent chromium	Hach spectro	Y	40	30	50	4	4
100-H Area	199-H4-13	Total chromium (filt.)	ICP metals	Y	44	25	54	3	3
100-H Area	199-H4-15A	Total chromium (filt.)	ICP metals	Y	51	46	56	2	2
100-H Area	199-H4-17	Hexavalent chromium	Hach spectro	Y	30	30	40	4	4
100-H Area	199-H4-17	Total chromium (filt.)	ICP metals	Y	29	28	29	2	2
100-H Area	199-H4-18	Hexavalent chromium	Hach spectro	Y	50	30	90	3	3
100-H Area	199-H4-18	Total chromium (filt.)	ICP metals	Y	46	25	84	3	3
100-H Area	199-H4-45	Hexavalent chromium	Hach spectro	Y	30	20	30	4	4
100-H Area	199-H4-45	Total chromium (filt.)	ICP metals	Y	25	21	30	2	2
100-H Area	199-H4-63	Hexavalent chromium	Hach spectro	Y	60	40	90	14	14
100-H Area	199-H4-63	Total chromium (filt.)	ICP metals	Y	73	57	81	3	3
100-H Area	199-H4-64	Hexavalent chromium	Hach spectro	Y	40	10	60	15	15
100-H Area	199-H4-64	Total chromium (filt.)	ICP metals	Y	62	57	65	3	3
100-H Area	199-H6-1	Total chromium (filt.)	ICP metals	Y	39	38	40	2	2
100-H Area	699-89-35	Total chromium (filt.)	ICP metals	Y	7	7	7	1	1
100-F Area	199-F1-2	Total chromium (filt.)	ICP metals	Y	8	8	8	2	1
100-F Area	199-F5-1	Total chromium (filt.)	ICP metals	Y	5	5	5	2	1
100-F Area	199-F5-3	Total chromium (filt.)	ICP metals	Y	5	5	5	1	1
100-F Area	199-F5-6	Total chromium (filt.)	ICP metals	Y	36	27	45	2	2
100-F Area	199-F5-42	Total chromium (filt.)	ICP metals	Y	8	8	8	2	1

Shoreline Segment	Well Name	Constituent	Method	Filt?	Average (µg/L)	Minimum (µg/L)	Maximum (µg/L)	No. Results	No. Detects
100-F Area	199-F5-43A	Total chromium (filt.)	ICP metals	Y	6	6	6	2	1
100-F Area	199-F5-44	Total chromium (filt.)	ICP metals	Y	14	10	18	2	2
100-F Area	199-F6-1	Total chromium (filt.)	ICP metals	Y	5	5	5	2	1
100-F Area	699-66-23	Total chromium (filt.)	ICP metals	Y				2	0
Hanford Townsite	699-41-1A	Total chromium (filt.)	ICP metals	Y				1	0
Hanford Townsite	699-42-E9B	Total chromium (filt.)	ICP metals	Y				1	0
Hanford Townsite	699-46-4	Total chromium (filt.)	ICP metals	Y	4	4	4	2	1
Energy Northwest	699-10-E12	Total chromium (filt.)	ICP metals	Y	4	4	4	4	2
Energy Northwest	699-20-E12O	Total chromium (filt.)	ICP metals	Y	3	3	3	2	1
300 Area North	699-S3-E12	Total chromium (filt.)	ICP metals	Y				1	0
300 Area North	699-S11-E12AP	Total chromium (filt.)	ICP metals	Y				1	0
300 Area North	699-S19-E13	Total chromium (filt.)	ICP metals	Y	4	4	4	2	1
100-B Area	199-B2-13	Specific conductance	Field Instrument	N	325	325	325	1	1
100-B Area	199-B3-1	Specific conductance	Field Instrument	N	394	394	394	1	1
100-B Area	199-B3-46	Specific conductance	Field Instrument	N	417	417	417	1	1
100-B Area	199-B3-47	Specific conductance	Field Instrument	N	416	416	416	1	1
100-B Area	699-72-92	Specific conductance	Field Instrument	N	391	391	391	1	1
100-K Area	199-K-18	Specific conductance	Field Instrument	N	537	515	568	14	14
100-K Area	199-K-19	Specific conductance	Field Instrument	N	421	413	429	2	2
100-K Area	199-K-20	Specific conductance	Field Instrument	N	369	336	406	13	12
100-K Area	199-K-21	Specific conductance	Field Instrument	N	434	408	455	5	5
100-K Area	199-K-22	Specific conductance	Field Instrument	N	371	354	400	5	5
100-K Area	199-K-31	Specific conductance	Field Instrument	N	343	339	346	2	2
100-K Area	199-K-32A	Specific conductance	Field Instrument	N	327	308	356	5	5
100-K Area	199-K-33	Specific conductance	Field Instrument	N	509	467	563	4	4
100-K Area	199-K-37	Specific conductance	Field Instrument	N	344	342	346	2	2
100-K Area	199-K-114A	Specific conductance	Field Instrument	N	306	186	418	15	15
100-K Area	199-K-117A	Specific conductance	Field Instrument	N	220	166	269	41	41
100-K Area	199-K-126	Specific conductance	Field Instrument	N	403	279	528	12	12
100-N Area	199-N-2	Specific conductance	Field Instrument	N	472	430	532	15	15
100-N Area	199-N-3	Specific conductance	Field Instrument	N	1297	1080	1470	15	15
100-N Area	199-N-14	Specific conductance	Field Instrument	N	274	247	327	3	3
100-N Area	199-N-17	Specific conductance	Field Instrument	N	1091	1091	1091	1	1
100-N Area	199-N-18	Specific conductance	Field Instrument	N	1251	1160	1358	3	3
100-N Area	199-N-19	Specific conductance	Field Instrument	N	947	947	947	1	1
100-N Area	199-N-21	Specific conductance	Field Instrument	N	1037	1037	1037	1	1
100-N Area	199-N-26	Specific conductance	Field Instrument	N	749	749	749	1	1
100-N Area	199-N-46	Specific conductance	Field Instrument	N	499	421	557	4	4
100-N Area	199-N-47	Specific conductance	Field Instrument	N	994	994	994	1	1
100-N Area	199-N-51	Specific conductance	Field Instrument	N	271	266	276	2	2
100-N Area	199-N-54	Specific conductance	Field Instrument	N	935	935	935	1	1
100-N Area	199-N-67	Specific conductance	Field Instrument	N	562	528	596	2	2
100-N Area	199-N-75	Specific conductance	Field Instrument	N	292	292	292	1	1
100-N Area	199-N-76	Specific conductance	Field Instrument	N	299	287	310	2	2
100-N Area	199-N-92A	Specific conductance	Field Instrument	N	183	183	183	2	2
100-N Area	199-N-96A	Specific conductance	Field Instrument	N	549	542	556	2	2
100-N Area	199-N-99A	Specific conductance	Field Instrument	N	202	198	206	2	2
100-N Area	199-N-105A	Specific conductance	Field Instrument	N	376	360	384	7	7
100-D Area	199-D3-2	Specific conductance	Field Instrument	N	481	418	512	4	4
100-D Area	199-D4-1	Specific conductance	Field Instrument	N	1035	896	1173	2	2

Shoreline Segment	Well Name	Constituent	Method	Filt?	Average (µg/L)	Minimum (µg/L)	Maximum (µg/L)	No. Results	No. Detects
100-D Area	199-D4-13	Specific conductance	Field Instrument	N	677	620	821	5	5
100-D Area	199-D4-14	Specific conductance	Field Instrument	N	431	385	451	4	4
100-D Area	199-D4-19	Specific conductance	Field Instrument	N	495	483	500	5	5
100-D Area	199-D5-20	Specific conductance	Field Instrument	N	445	431	473	5	5
100-D Area	199-D5-36	Specific conductance	Field Instrument	N	227	222	237	5	5
100-D Area	199-D5-37	Specific conductance	Field Instrument	N	336	327	353	5	5
100-D Area	199-D5-44	Specific conductance	Field Instrument	N	198	115	224	5	5
100-D Area	199-D8-4	Specific conductance	Field Instrument	N	615	598	632	2	2
100-D Area	199-D8-5	Specific conductance	Field Instrument	N	472	472	472	1	1
100-D Area	199-D8-6	Specific conductance	Field Instrument	N	809	809	809	1	1
100-D Area	199-D8-55	Specific conductance	Field Instrument	N	240	240	240	1	1
100-D Area	199-D8-68	Specific conductance	Field Instrument	N	363	171	655	13	13
100-D Area	199-D8-69	Specific conductance	Field Instrument	N	331	211	492	11	11
100-D Area	699-97-51A	Specific conductance	Field Instrument	N	380	380	380	1	1
100-H Area	199-H4-3	Specific conductance	Field Instrument	N	1147	703	1498	3	3
100-H Area	199-H4-4	Specific conductance	Field Instrument	N	573	200	936	13	13
100-H Area	199-H4-5	Specific conductance	Field Instrument	N	538	520	572	13	13
100-H Area	199-H4-8	Specific conductance	Field Instrument	N	539	506	558	3	3
100-H Area	199-H4-9	Specific conductance	Field Instrument	N	788	716	859	2	2
100-H Area	199-H4-10	Specific conductance	Field Instrument	N	330	304	344	3	3
100-H Area	199-H4-11	Specific conductance	Field Instrument	N	484	484	484	1	1
100-H Area	199-H4-12A	Specific conductance	Field Instrument	N	470	376	535	3	3
100-H Area	199-H4-13	Specific conductance	Field Instrument	N	501	491	507	3	3
100-H Area	199-H4-15A	Specific conductance	Field Instrument	N	419	395	438	3	3
100-H Area	199-H4-17	Specific conductance	Field Instrument	N	515	499	528	3	3
100-H Area	199-H4-18	Specific conductance	Field Instrument	N	517	496	536	3	3
100-H Area	199-H4-45	Specific conductance	Field Instrument	N	533	525	540	3	3
100-H Area	199-H4-63	Specific conductance	Field Instrument	N	425	332	540	11	11
100-H Area	199-H4-64	Specific conductance	Field Instrument	N	413	259	474	12	12
100-H Area	199-H6-1	Specific conductance	Field Instrument	N	588	572	603	2	2
100-H Area	699-89-35	Specific conductance	Field Instrument	N	416	416	416	1	1
100-F Area	199-F1-2	Specific conductance	Field Instrument	N	510	504	515	2	2
100-F Area	199-F5-1	Specific conductance	Field Instrument	N	258	206	298	3	3
100-F Area	199-F5-3	Specific conductance	Field Instrument	N	377	361	393	2	2
100-F Area	199-F5-6	Specific conductance	Field Instrument	N	355	317	393	2	2
100-F Area	199-F5-42	Specific conductance	Field Instrument	N	170	156	183	2	2
100-F Area	199-F5-43A	Specific conductance	Field Instrument	N	178	158	197	2	2
100-F Area	199-F5-44	Specific conductance	Field Instrument	N	217	177	257	2	2
100-F Area	199-F6-1	Specific conductance	Field Instrument	N	191	190	191	2	2
100-F Area	699-66-23	Specific conductance	Field Instrument	N	581	581	581	1	1
Hanford Townsite	699-40-1	Specific conductance	Field Instrument	N	412	412	412	1	1
Hanford Townsite	699-41-1A	Specific conductance	Field Instrument	N	409	409	409	1	1
Hanford Townsite	699-42-12A	Specific conductance	Field Instrument	N	386	386	386	1	1
Hanford Townsite	699-42-E9B	Specific conductance	Field Instrument	N	424	424	424	1	1
Hanford Townsite	699-46-4	Specific conductance	Field Instrument	N	387	386	387	2	2
Energy Northwest	699-10-E12	Specific conductance	Field Instrument	N	608	602	613	4	4
Energy Northwest	699-20-E12O	Specific conductance	Field Instrument	N	321	321	321	1	1
300 Area North	699-S3-E12	Specific conductance	Field Instrument	N	359	359	359	1	1
300 Area North	699-S11-E12AP	Specific conductance	Field Instrument	N	365	365	365	1	1
300 Area North	699-S19-E13	Specific conductance	Field Instrument	N	474	456	492	4	4

Shoreline Segment	Well Name	Constituent	Method	Filt?	Average (µg/L)	Minimum (µg/L)	Maximum (µg/L)	No. Results	No. Detects
300 Area	399-1-10A	Specific conductance	Field Instrument	N	458	399	481	9	9
300 Area	399-1-10B	Specific conductance	Field Instrument	N	311	298	320	9	9
300 Area	399-1-16A	Specific conductance	Field Instrument	N	433	380	464	9	9
300 Area	399-1-16B	Specific conductance	Field Instrument	N	329	315	336	9	9
300 Area	399-1-18A	Specific conductance	Field Instrument	N	469	445	476	9	9
300 Area	399-1-18B	Specific conductance	Field Instrument	N	368	347	375	9	9

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