

**Assessment of Potential Impacts of
Major Groundwater Contaminants to
Fall Chinook Salmon (*Oncorhynchus
tshawytscha*) in the Hanford Reach,
Columbia River**

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Executive Summary

Past operations of Hanford Site facilities have contaminated the groundwater adjacent to the Hanford Reach of the Columbia River, Washington, with various chemical and radiological constituents. The groundwater is hydraulically connected to the river and contains concentrations of contaminants that sometimes exceed federal and/or state drinking water standards or standards for the protection of aquatic life. For example, concentrations of chromium in shoreline seeps and springs at most 100 Area operable units exceed concentrations found to be toxic to fish. Nitrate and tritium concentrations in shoreline seeps are generally below drinking water standards and concentrations potentially toxic to aquatic life, but nitrate concentrations may be high enough to synergistically interact with and exacerbate chromium toxicity.

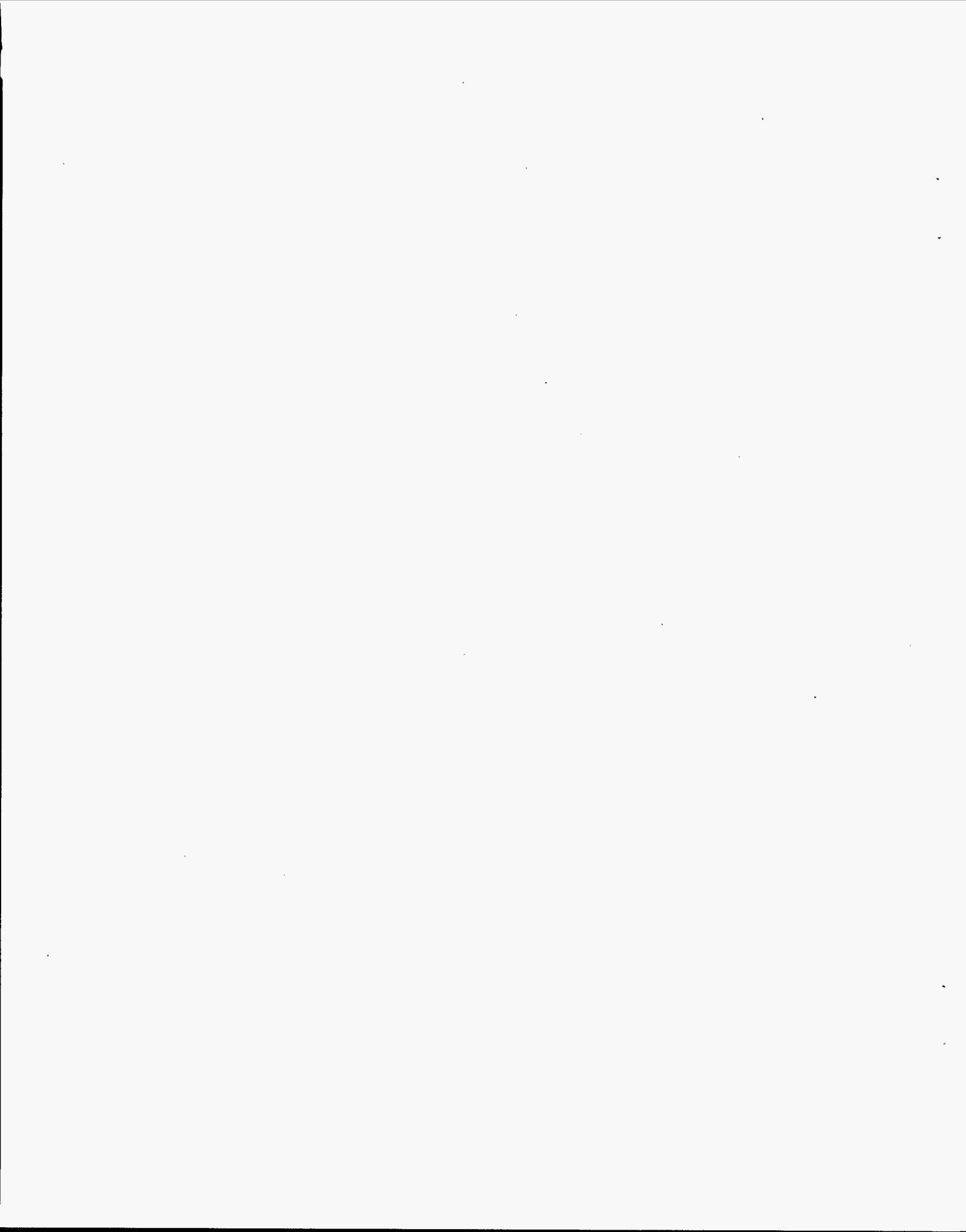
The Hanford Reach also supports the largest run of fall chinook salmon (*Oncorhynchus tshawytscha*) in the Columbia River Basin. Numbers of fall chinook salmon returning to the Hanford Reach have increased relative to other mainstem populations during the last 30 years. Groundwater discharge appears to occur near some salmon spawning areas, but contaminants are generally not detectable in surface water samples. The concentration and potential toxicity of contaminants in the interstitial waters of the substrate where fall chinook salmon embryogenesis occurs are presently unknown. New tools are required to characterize the extent of groundwater contaminant discharge to the Hanford Reach and to resolve uncertainties associated with assessment of potential impacts to fall chinook salmon.

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1.0 Introduction

Past operations of Hanford Site facilities have contaminated the groundwater adjacent to the Columbia River with various chemical and radiological constituents. The groundwater constituents that exceed established or proposed drinking water standards (DWS) (40 CFR 141-143 and Chapter 248-54 Washington Administrative Code) or ambient water quality criteria (EPA 1986) include hexavalent chromium (Cr)^(a), nitrate (NO₃), tritium (³H), strontium-90 (⁹⁰Sr), technetium-99 (⁹⁹Tc), and total uranium (U) (DOE 1993).

Contaminants generated from past reactor operations in the 100 Areas are transported to the Hanford Reach via the groundwater (DOE 1993). Other areas of the Hanford Reach (Figure 1.1) have not been studied as extensively; however, it is apparent that contaminants, especially tritium generated in the 200 Area, have also migrated to the Hanford Reach shoreline and enter the Columbia River near the old Hanford Townsite and downstream to the 300 Area (Woodruff and Hanf 1993). Contaminant concentrations measured in Columbia River water (i.e., mid-channel) are well below drinking water standards or standards for the protection of aquatic life (Woodruff and Hanf 1993; DOE 1993; Dirkes et al. 1993). However, contaminant concentrations in shoreline seeps/springs exceed the DWS in some locations: chromium at 100-K, 100-D, and 100-H; tritium at 100-B, 100-N, and the Old Hanford Townsite; gross beta at 100-N and 100-H; and strontium-90 at 100-N, 100-H, and 100-F (Woodruff and Hanf 1993; Peterson and Johnson 1992).

One of the primary pathways for environmental contaminants to enter the Hanford Reach is the infiltration and migration of wastes through soil to the groundwater of the unconfined aquifer. Once in the unconfined aquifer, contaminants may be carried down gradient toward the surface water of the Columbia River. Rates of transport to the river vary, and concentrations of contaminants reaching the river are affected by sorption, degradation, decay, and groundwater flow rates. When the hydraulic head of the groundwater exceeds the elevation of the surface water, discharge from the aquifer to the river occurs. This discharge occurs in the form of shoreline seeps and springs and upwelling of water through the river bottom into the river channel. Although the relative contribution of various groundwater sources has not been quantified for the entire reach, groundwater upwelling is believed to contribute the majority of groundwater discharge to the Columbia River. Shoreline seeps contribute an insignificant amount, occurring usually during periods of low river discharge (Dirkes 1990).

Fluctuations in Columbia River discharge and stage influence the groundwater level and contaminant concentrations in wells located up to 800 m from the river (Luttrell et al. 1992). During periods of high Columbia River discharge, water moves from the river into the riverbank (bank storage) where it overlies and/or mixes with groundwater (Peterson and Johnson 1992). As the river recedes, water flows back into the river channel (bank discharge). Where the bottom of the river channel is above the bottom of the unconfined aquifer, groundwater discharge to the river occurs beneath the river bed, as well as from springs and seeps along the river bank. The concentrations of contaminants in the interstitial areas of the substrate where the groundwater upwells into the river channel are generally higher than concentrations observed in the river.

The interface zone, where groundwater upwells through the river bottom material into the surface water, is important for a number of biological processes. For example, bottom substrate provides habitat for incubation of fall chinook salmon eggs and fry (i.e., embryogenesis), spawning, and early life history rearing for benthic fish species such as sculpins and dace, and production of periphyton and aquatic invertebrates, including sensitive species such as snails and limpets. Thus, understanding the transport of contaminants to the Columbia River, chemical speculation of the contaminants, and concentrations in the subsurface environment of the river channel where upwelling occurs is necessary to make risk assessments to the aquatic ecosystem.

(a) Unless otherwise specified, "chromium" as used throughout this report, denotes the hexavalent (Cr⁺⁶) form.

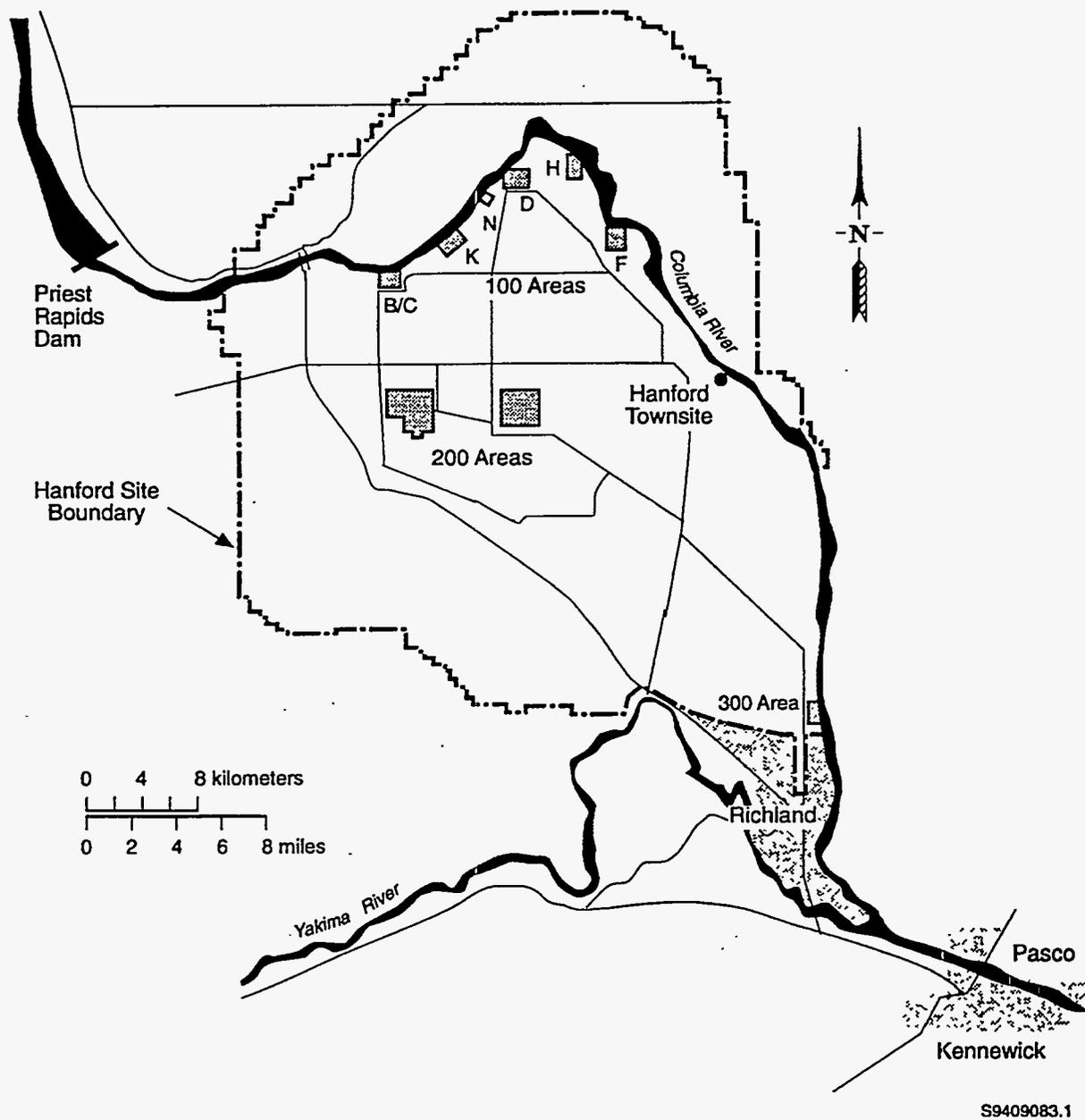


Figure 1.1. Hanford Reach of the Columbia River Near Richland, Washington

This evaluation was initiated by Pacific Northwest Laboratory (PNL)^(a) in response to inquiries from the U.S. Department of Energy (DOE) on the potential for chromium to impact aquatic biota within the Hanford Reach of the Columbia River. Potential impacts to fall chinook salmon (*Oncorhynchus tshawytscha*) were evaluated because the Hanford Reach population is the largest in the Columbia River Basin (Dauble and Watson 1990), and the population is extremely important to sport and commercial fisheries, Columbia River salmon recovery plans, and federal treaties with Canada and Native Americans. Long-term trends in fall chinook salmon abundance in the Hanford Reach provide no evidence that populations have been negatively impacted by site operations (Watson 1970; Dauble and Watson 1990). However, early life stages of fall chinook salmon spend a significant portion of their life cycle in or near the substrate of the river bottom where concentrations of groundwater contaminants would be expected to be the highest. Therefore, the purpose of this report is to provide a preliminary assessment of the potential for selected contaminants to negatively affect the production of fall chinook salmon in the Hanford Reach.

This report is organized into four additional sections. Section 2.0 identifies the major contaminants of concern in groundwater, shoreline seepage, and river water from the Hanford Site. It also presents the federal drinking water standards for each constituent. Section 3.0 briefly reviews the potential toxicity of the selected contaminants to fish, compares the toxicity of each constituent to the concentrations measured in Hanford Site groundwater, and identifies constituents that may affect fall chinook salmon embryogenesis. Section 4.0 assesses the potential impacts on fall chinook salmon from groundwater contamination.

(a) Pacific Northwest Laboratory is operated by Battelle Memorial Institute for the U.S. Department of Energy under Contract DE-AC06-76RLO 1830.

2.0 Major Contaminants of Concern

The major contaminants found in groundwater at the Hanford Site include chromium (Cr), nitrate (NO₃), tritium (³H), strontium-90 (⁹⁰Sr), technetium-99 (⁹⁹Tc), and total uranium (U) (DOE 1993). We selected chromium, tritium, and nitrate for our assessment of potential impacts to aquatic resources because these constituents are known to negatively affect biological processes (i.e., concentrations exceed drinking water standards (DWS) and/or water quality criteria for the protection of aquatic life; Table 2.1) and a good historical record of these contaminants in the groundwater exists. Thus, spatial and temporal trends in their concentration help define groundwater movement and the extent of groundwater contamination.

This section of the report provides a brief historical account of activities relating to the use of chromium, tritium, and nitrate at Hanford; summarizes the current regulatory standards for these contaminants; discusses information concerning their concentrations in Hanford environs; and relates measured values to their aquatic toxicity.

Chromium

Sodium dichromate was used in reactor operations as an additive to inhibit corrosion in reactor piping, and large quantities of chromium were discharged into trenches, cribs, and the Columbia River. In water samples, hexavalent chromium is most common because it is highly soluble in water, exists in solution as a complex anion, and is not sorbed to any significant degree by clays or hydrous metal oxides. It is, however, sorbed strongly to activated carbon. Because hexavalent chromium is a moderately strong oxidizing agent, it is able to react with reducing materials to form trivalent chromium. Trivalent chromium reacts with aqueous hydroxide ions to form the insoluble chromium hydroxide, and precipitation of this material is common in natural waters. Because of ambient pH and redox potential, hexavalent chromium is the form considered most toxic under conditions found in the Hanford Reach. A measurement of total chromium (hexavalent and trivalent chromium) in a water sample essentially reflects the hexavalent concentration (Peterson and Johnson 1992).

Regulatory Standards and Guidelines

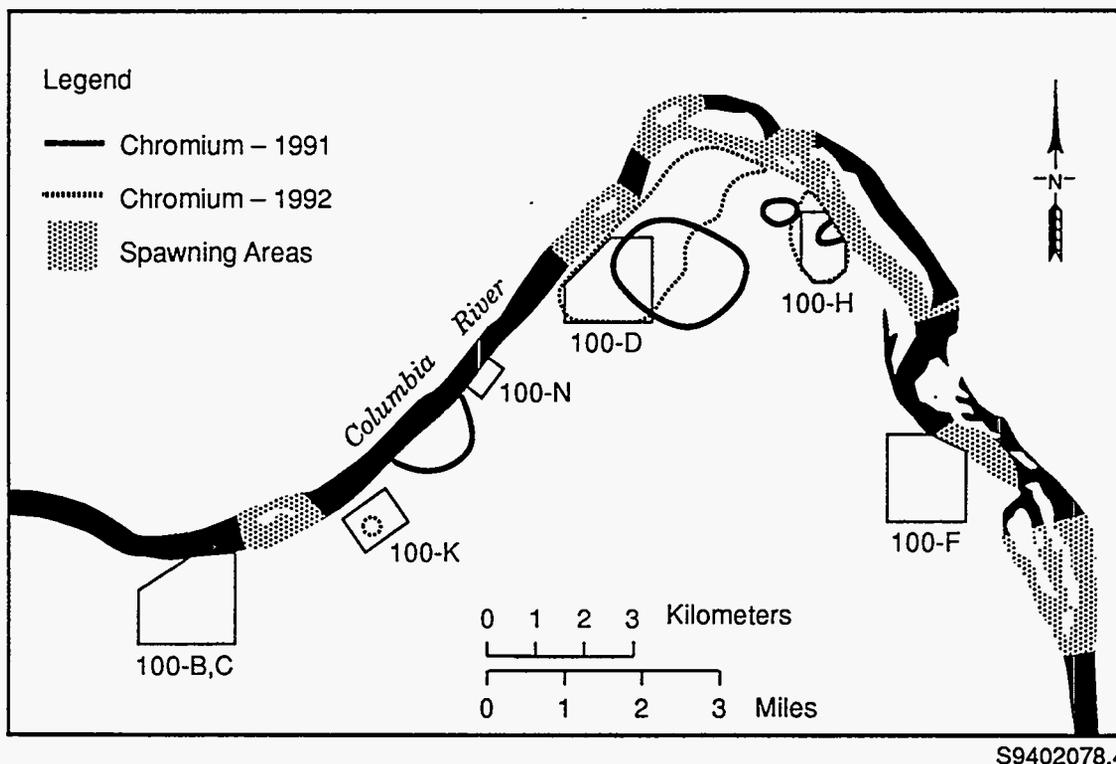
The state of Washington-recommended maximum concentration level for total chromium in drinking water is 0.05 milligrams per liter (mg/L). In 1992, the federal DWS for total chromium was raised from 0.05 mg/L to 0.10 mg/L (40 CFR 147.62, Phase II Rule, effective July 30, 1992). The EPA's water quality criteria for the protection of aquatic organisms is 0.01 mg/L (EPA 1980).

Concentrations at Hanford

Chromium has been detected in groundwater monitoring wells adjacent to the Columbia River near operable units 100-B, 100-D, 100-H, and 100-K (Figure 2.1; Dresel et al. 1993a, 1993b; Evans et al. 1990, 1992a, 1992b). In 1991, total chromium concentrations in the groundwater exceeded the former federal DWS (0.05 mg/L) at 100-D, 100-H, and 100-K Areas and the surrounding areas

Table 2.1. Drinking Water Standard and U.S. Environmental Protection Agency Water Quality Criteria to Protect Aquatic Life for Chromium, Nitrate, and Tritium

Constituent	DWS	EPA Water Quality Criteria
Total chromium (Cr)	0.10 mg/L (federal) 0.05 mg/L (state)	0.01 mg/L (hexavalent)
Nitrate (NO ₃ ⁻)	45 mg/L	None exist
Tritium (³ H)	20,000 pCi/L	None exist



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Figure 2.1. Distribution of Chromium in Groundwater Showing Concentrations Greater than the Federal Drinking Water Standard (0.10 mg/L) During 1991 and 1992 in Relation to the Major Fall Chinook Salmon Spawning Areas (Modified from Dresel et al. 1993b; Woodruff and Hanf 1993; and Evans et al. 1992b.)

(Evans et al. 1992b). In 1992, chromium concentrations in the groundwater exceeded the current federal DWS at 100-D, 100-H, 100-K, 100-N, and in a few unfiltered samples in the 300 Area (Dresel et al. 1993b; Woodruff and Hanf 1993). Concentrations of chromium in filtered and unfiltered samples were similar in 1992, indicating that chromium is in solution and expected to be mobile (Dresel et al. 1993b). Some of the highest measured chromium concentrations on the Hanford Site have been found in groundwater monitoring wells within the 100-D Area (e.g., wells 199-D5-12 and 199-D5-15). The concentration of chromium in these wells increased from 0.46 mg/L in March 1990 to over 2 mg/L in 1992 (Dresel et al. 1993b; Evans et al. 1992a). A sizeable chromium groundwater plume is located in the 100-H Area between the 183-H Solar Evaporation Basins and the bank of the Columbia River (Evans et al. 1992b). Samples taken within this plume have ranged in concentration from 0.04 to 0.41 mg/L (Dresel et al. 1993b; Evans et al. 1992b). The maximum concentration of chromium detected in the 100-K Area in 1992 was 1.4 mg/L (Dresel et al. 1993b).

As described previously, river bank discharge (seepage) is the outflow of mixed groundwater and river water from the river bank usually after periods of high river flow. In springs sampled during 1988, chromium was not detected (Dirkes 1990; Table 2.2). In samples collected from shoreline springs during 1991, chromium was detected entering the Columbia River in the 100-B/C, 100-K, 100-D, 100-H, and 100-F Areas (Peterson and Johnson 1992; Table 2.2). Chromium exceeded the federal DWS (0.10 mg/L) in seepage at the 100-D Area and was slightly elevated at 100-B, 100-K, and 100-H Areas (Peterson and Johnson 1992). Except for the springs sampled in the 100-F Area, the concentrations exceeded U.S. Environmental Protection Agency (EPA) water quality criteria for aquatic organisms (0.01 mg/L). However, seepage to the river from surface springs is believed to contribute a small fraction of the total groundwater discharge to the Columbia River

Table 2.2. Concentrations of Total Chromium in Groundwater Monitoring Wells (1988 - 1993) and Shoreline Seeps (1988 and 1991)

Area	Total Chromium Concentrations (mg/L)						Total Chromium Concentrations (mg/L)	
	Groundwater Wells						Seeps/Springs	
	1988(a)	1989(b,c)	1990(c,d)	1991(e)	1992(c,f)	1993(g)	1988(h)	1991(i)
100-B	0.029-0.036	<0.01-0.018	<0.01-0.049	<0.01-0.02	<0.02-0.023	<0.002-0.067	NS	0.01-0.055
100-K	NS	<0.01-0.16	<0.01-0.157	NS	1.4	0.004-0.170	NS	0.01-0.07
100-N	<0.01-0.014	<0.01-0.039	<0.01-0.196	0.021-0.17	ND	NS	<0.01	0.001-0.01
100-D	NS	0.120-0.692	0.12-0.464	NS	0.04-2.0	0.005-1.82	NS	0.07-0.15
100-H	<0.01-0.422	0.012-0.42	<0.01-0.359	0.025-0.32	0.004-0.36	0.004-0.049	<0.01	0.01-0.08
100-F	<0.01-0.017	<0.01-0.013	<0.01-0.014	0.022	0.004-0.303	0.002-0.206	NS	0.001-0.01

(a) Jaquish and Bryce (1989).

(b) Evans et al. (1990).

(c) Filtered sample (i.e., Cr⁺⁶).

(d) Evans et al. (1992a); Dresel et al. (1993a).

(e) Evans et al. (1992b).

(f) Merz and Dresel (1993); Dresel et al. (1993b).

(g) Hanford Environmental Information System.

(h) Dirkes (1990).

(i) Peterson and Johnson (1992).

NS = not sampled.

ND = not detected.

(DOE 1993; Dirkes 1990). Because of high dilution from the Columbia River, samples of river water from the Hanford Reach do not indicate elevated quantities of chromium except for localized areas of high concentration near shoreline seeps (Dirkes 1990).

Based on concentrations of chromium measured in the groundwater and shoreline seeps and springs, hexavalent chromium from the 100-Area activities is believed to be reaching the Hanford Reach (DOE 1993). Cushing (1993) sampled periphyton and caddisfly larvae from the river adjacent to the 100-Area operable units in 1991-1992 and reported that levels for stable chromium were below detection limits. However, chromium was detected in organic material near shoreline seeps during 1993 in the vicinity of the 100-H and 100-D Areas (J. Amonette, PNL, personal communication). The concentration of chromium at the interface zone between the surface and groundwater in the river channel is unknown.

Aquatic Toxicity

Little is known about the effects of chromium on aquatic organisms because chromium toxicity varies widely depending on pH, temperature, valence of chromium ion, water hardness, and presence of other contaminants. Closely related species have different sensitivity to chromium. Trivalent chromium is relatively non-toxic in most freshwater systems because it is highly insoluble, forms stable complexes, and does not readily cross cell membranes. Conversely, hexavalent chromium is readily taken up by living organisms and is highly toxic in aquatic ecosystems. The effects of long-term, continuous exposure of aquatic organisms to chromium is not well understood.

The chronic toxicity level to aquatic life (lowest observed effect level) determined by the EPA (EPA 1986) is 0.01 mg/L for hexavalent chromium. Chronic bioassays performed at Hanford with chromium and sensitive life stages of rainbow trout and fall chinook salmon suggested that concentrations of chromium should not exceed 0.02 mg/L in the Columbia River (Foster et al. 1961, as cited in Becker 1990).

In general, concentrations of hexavalent chromium of ≥ 0.01 mg/L adversely impacted sensitive fish species (Eisler 1985; Table 2.3). The chronic lethal value for both rainbow trout (*O. mykiss*) and brook trout (*Salvelinus fontinalis*) reported by EPA (1986) is 0.264 mg/L. Survival of chinook salmon and rainbow trout was adversely affected by 0.08 mg/L of hexavalent chromium in a study by Olson and Foster (1956). Reduced growth, a sub-lethal effect, was a more sensitive indicator than mortality. In this same study, growth appeared to be reduced at the lowest concentration of 0.013 mg/L of hexavalent chromium (Olson and Foster 1956). Growth rates in young rainbow trout and chinook salmon were also reduced in fish exposed to 0.016 to 0.021 mg/L for 14 to 16 weeks (EPA 1980). Growth of chinook salmon was reduced at a measured concentration of 0.016 mg/L (EPA 1986). At a concentration of 0.023 mg/L hexavalent chromium, salinity tolerance and serum osmolality were impaired in migrating coho salmon (*O. kisutch*) exposed for 4 weeks (Sugatt 1980).

Nitrate

Nitric acid was used extensively in many decontamination operations of the reactors. Hence, nitrate was present in many waste streams and is present in groundwater monitoring wells throughout the site (DOE 1993; Dresel et al. 1993b; Evans et al. 1992b). Because nitrate is highly mobile in groundwater, it can be used to define the extent of contamination in Hanford aquifers (Evans et al. 1992b). We discuss nitrate here because it has the potential to react with other chemical contaminants, including chromium, and is a good indicator of groundwater movement.

Regulatory Standards and Guidelines

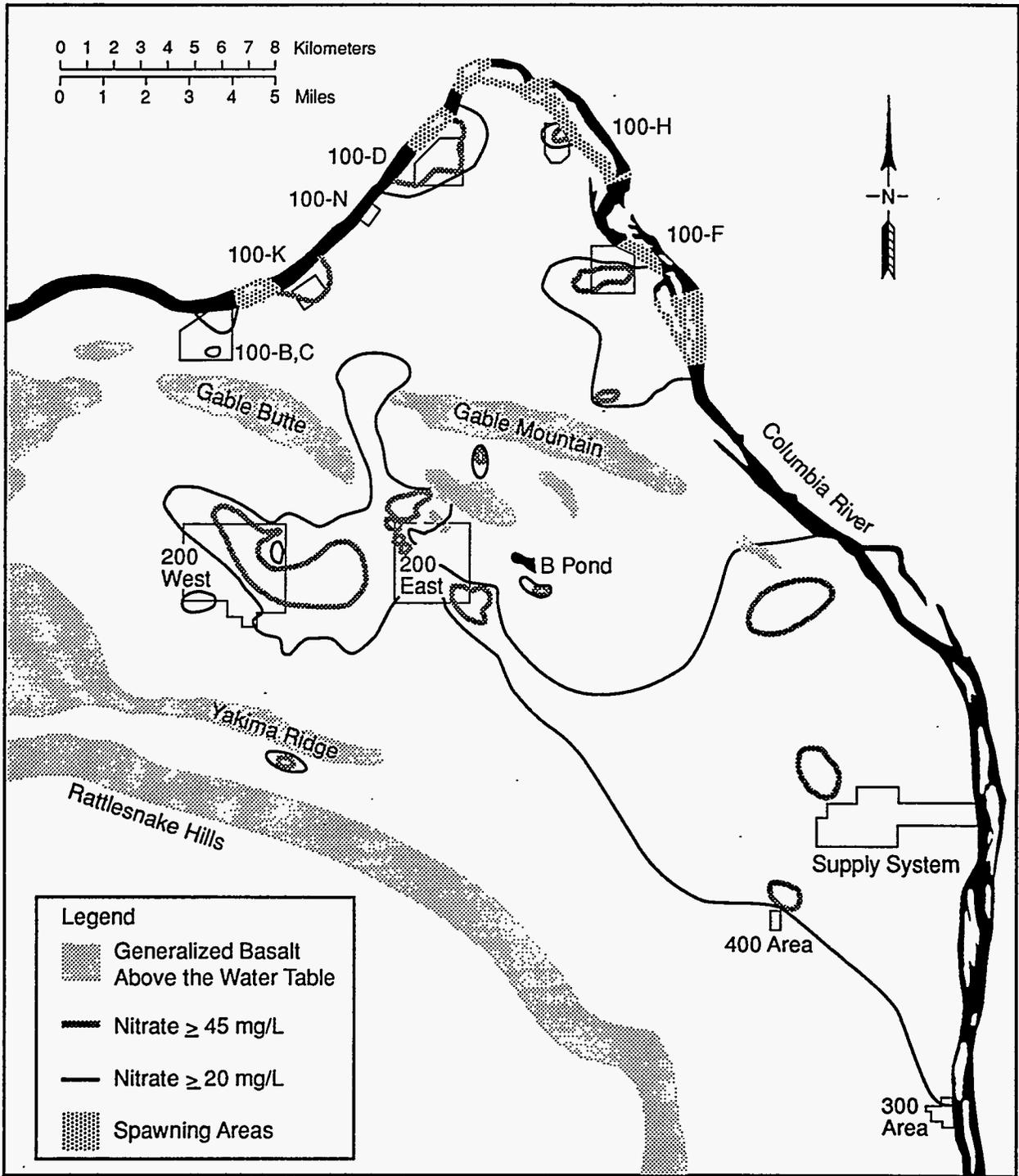
The DWS for nitrate (as NO_3^-) is 45 mg/L; no criteria exist for the protection of aquatic life. Concentrations of nitrate that would exhibit toxic effects on cold-water fish would rarely occur in nature; hence, restrictive criteria are not provided by EPA (EPA 1986).

Concentrations at Hanford

Nitrate plumes are located in groundwater throughout the 100 Areas, with the 100-K, 100-D, 100-H, and 100-F Areas having concentrations of nitrate exceeding the DWS (Figure 2.2). Nitrate plumes in the 300 and 400 Areas and also equal or exceed the DWS. Nitrate concentrations in groundwater plumes in 1992 ranged from 110 mg/L near 100-K to 220 mg/L near 100-H (Dresel et al. 1993b).

Table 2.3. Summary of Effects on Selected Fish Species from Hexavalent Chromium

Species	Parameter	Effective Concentration (mg/L)
Rainbow trout and chinook salmon fingerlings	Survival	0.08
Rainbow trout and chinook salmon fingerlings	Reduced growth rates	0.016-0.021 for 14-16 weeks
Coho juveniles	Impairment of salinity tolerance	0.023 for 4 weeks
Rainbow and brook trout	Survival	0.26



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Figure 2.2. Distribution of Nitrate in Groundwater During 1991 and 1992 in Relation to the Major Fall Chinook Salmon Spawning Areas (Modified from Dresel et al. 1993b; Woodruff and Hanf 1993; and Evans et al. 1992b.)

Concentrations of nitrate in groundwater seepage areas are below the DWS, even though nearby groundwater may be drastically higher in concentration (Table 2.4). For example, nitrate measured in shoreline seepage areas near the 100 Areas ranged from 1 to 5 mg/L in 1991 while near-shore wells ranged from 0.2 to 970 mg/L (Peterson and Johnson 1992; Evans et al. 1992b). Biological use of nitrate in the nearshore zone and bank storage of river water (dilution) may reduce levels of nitrate in seepage from those observed in adjacent groundwater samples (Peterson and Johnson 1992). Columbia River concentrations for nitrate do not exceed the DWS.

Aquatic Toxicity

The 96-hour and 7-day LC50 values for chinook salmon and rainbow trout were 1,310 and 1,080 mg/L and 1,360 and 1,060 mg/L nitrate nitrogen (NO₃-N) in fresh water, respectively (Westin 1974). Chronic data, a better indicator of the effects of low-level, long-term exposure, are not available.

Table 2.4. Concentrations of Nitrate in Groundwater Monitoring Wells (1988 - 1993) and Shoreline Seeps (1988 and 1991)

Area	Nitrate Concentrations (mg/L)						Nitrate Concentrations (mg/L) Seeps/Springs	
	1988 ^(a)	1989 ^(b)	1990 ^(c)	1991 ^(d)	1992 ^(e)	1993 ^(f)	1988 ^(g)	1991 ^(h)
100-B	9.5-28.5	12.9-48.4	13.2-33.3	18.0-21.2	18.0-25	0.4-30	6.7	1.5-2.5
100-K	2.7-86.8	3.0-66	4.9-42.3	28.4	110	0.9-120	NS	1-2
100-N	<2.5-67.1	<0.5-93	0.5-24.9	0.9-19.6	2-64	1.0-54	28.6	1-2
100-D	38.7-109	57.0-122	54.8-118	NS	45-88	0.9-170	22	3-4
100-H	3.2-663	4.6-524	3.1-240	0.2-970	110-220	0.4-870	ND	2-5
100-F	<2.5-244	<0.5-151	2.1-134	1.5-75	0.3-93	0.4-120	NS	2-3

- (a) Jaquish and Bryce (1989).
 - (b) Evans et al. (1990).
 - (c) Evans et al. (1992a); Dresel et al. (1993a).
 - (d) Evans et al. (1992b).
 - (e) Merz and Dresel (1993); Dresel et al. (1993b).
 - (f) Hanford Environmental Information System.
 - (g) Dirkes (1990).
 - (h) Peterson and Johnson (1992).
- NS = not sampled.
ND = not detected.

Tritium

Tritium was present in many waste streams and is the most mobile radionuclide in groundwater on the Hanford Site; hence, it reflects the relative extent of contamination in groundwater from site

operations. Tritium, in the form of tritiated water, is a common radiocontaminant arising from nuclear reactions involving fuel and moderating chemicals (Eisenbud 1987).

Regulatory Standards and Guidelines

The federal DWS for tritium is 20,000 picocuries per liter (pCi/L).^(a) Although no specific criteria exist for the protection of aquatic life for tritium, Washington Administrative Code (WAC) 173-201A-040 says "Toxic, radioactive, or deleterious material concentrations shall be below those which have the potential either singularly or cumulatively to adversely affect characteristic water uses, cause acute or chronic conditions to the most sensitive biota dependent upon those waters" (WSR 1992).

Concentrations at Hanford

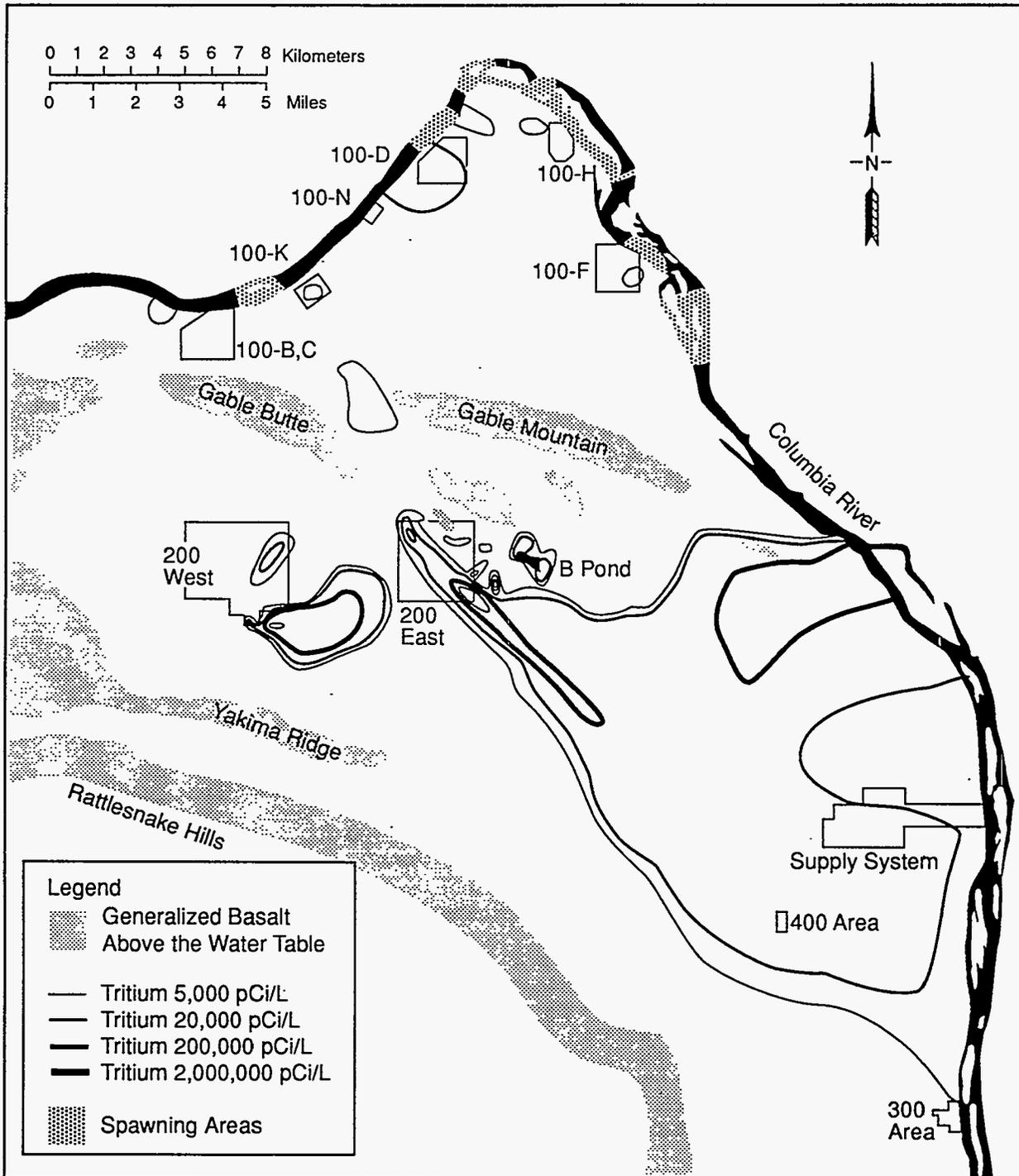
Extensive tritium plumes are located in the 100, 200-East, and 200-West Areas (Figure 2.3). The tritium plume from the 200-East Area affects the groundwater in the 400 Area and has extended to the Columbia River. Tritium plumes within portions of the 100-B, 100-K, 100-N, 100-D, 200-East, 200-West, 400, and 600 Areas have concentrations greater than the DWS (Dresel et al. 1993b). Well 199-K-30 in the 100-K Area contains the highest tritium concentration within the 100 Areas. The maximum tritium concentrations in this well have been 1,220,000, 798,000, and 1,690,000 pCi/L in 1988, 1991, and 1992, respectively (Dresel et al. 1993b; Woodruff and Hanf 1993; Evans et al. 1992a; Jaquish and Bryce 1989). Tritium levels of 78,000 and 39,700 pCi/L were detected in the 100-D Area during 1992 in wells 199-D5-17 and 199-D2-5, respectively (Dresel et al. 1993b; Woodruff and Hanf 1993). Water in the 100-K-East Basins contain tritium at a concentration of 3,700,000 pCi/L (Evans et al. 1992b). The DOE indicates this basin has leaked in the past, and several unidentified tritium sources may be in that area (Evans et al. 1992b). A tritium plume originating in the 200-East Area continues to move to the east-southeast and discharge into the Columbia River from the Hanford Townsite to the 300 Area (Dresel et al. 1993b; Evans et al. 1992b). The maximum concentration of tritium in one well (699-S19-E13) just north of the 300 Area was 11,600 pCi/L in 1992 (Woodruff and Hanf 1993). A maximum groundwater concentration of tritium near the Hanford Townsite was greater than 200,000 pCi/L in 1992 (Luttrell et al. 1992).

Shoreline seepage and spring concentrations in the 100 Areas are generally representative of nearby monitoring well concentrations (Table 2.5). Samples of shoreline springs taken in 1988 near the 100-N shoreline ranged from 74,000 to 111,000 pCi/L (Dirkes 1990) while concentrations in groundwater wells reached 459,000 pCi/L (Jaquish and Bryce 1989). Shoreline seepage areas sampled in 1991 showed tritium concentrations that exceed 20,000 pCi/L at 100-B/C and 100-N Areas, and detectable levels at 100-K, 100-D, and 100-H Areas (Peterson and Johnson 1992).

Localized concentrations of tritium in the river as a result of high concentrations in shoreline seeps were noted by Dirkes (1990). Maximum concentrations of tritium in springs and nearshore waters during 1988 at the 100-N Area were 111,000 and 76,400 pCi/L, respectively (Dirkes 1990). Samples of the 100-N spring and adjacent river water in 1991 showed concentrations of 15,900 and 300 pCi/L, respectively (Peterson and Johnson 1992). River stage and discharge have been shown to significantly affect concentrations of tritium measured in shoreline seeps (Luttrell et al. 1992; Peterson and Johnson 1992). Therefore, it is not clear if this represents a long-term decline in concentration, or if it is due to variation in samples resulting from location, bank storage, and river discharge.

Outside of areas near the 100-N Area and Hanford Townsite spring discharges, the river concentrations of tritium are below the DWS (Dirkes 1990). The total tritium mass discharge to the Columbia River was calculated from groundwater modeling to be from approximately 1,100 to 1,400 Ci/yr (Luttrell et al. 1992). The tritium concentration in the river contributed from groundwater discharge was calculated to be approximately 10 pCi/L (Luttrell et al. 1992).

(a) Concentration assumed to yield an annual dose of 4 mrem/yr.



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Figure 2.3. Distribution of Tritium in Groundwater During 1991 and 1992 in Relation to the Major Fall Chinook Salmon Spawning Areas (Modified from Dresel et al. 1993b; Woodruff and Hanf 1993; and Evans et al. 1992b.)

Table 2.5. Concentrations of Tritium in Groundwater Monitoring Wells (1988 - 1993) and Shoreline Seeps (1988 and 1991)

Area	Tritium Concentrations (pCi/L) Groundwater Wells						Tritium Concentrations (pCi/L) Seeps/Springs	
	1988 ^(a)	1989 ^(b)	1990 ^(c)	1991 ^(d)	1992 ^(e)	1993 ^(f)	1988 ^(g)	1991 ^(h)
100-B	1,470 - 49,700	1,980 - 42,900	2,090 - 4,570	2,280 - 3,530	1,990 - 24,000	1,600 - 17,000	1,100	1,500 - 21,000
100-K	471 - 1,220,000	491 - 882,000	536 - 823,000	655 - 798,000	1,690,000	LDL - 3,320,000	NS	400 - 10,000
100-N	LDL - 459,000	LDL - 218,000	LDL - 260,000	357 - 107,000	LDL - 94,100	LDL - 78,500	74,000 - 111,000	3,000 - 30,000
100-D	3,990 - 33,500	3,690 - 53,300	3,180 - 32,300	NS	3,610 - 78,000	LDL - 73,000	21.5	1,200 - 3,000
100-H	489 - 5,550	429 - 5,280	4,490	LDL - 4,790	LDL - 5,080	LDL - 9,300	545	400 - 4,000
100-F	LDL - 13,500	LDL - 9,550	LDL - 9,520	LDL - 9,980	LDL - 4,040	LDL - 180,000	NS	200 - 400

(a) Jaquish and Bryce (1989).

(b) Evans et al. (1990).

(c) Evans et al. (1992a); Dresel et al. (1993a).

(d) Evans et al. (1992b).

(e) Merz and Dresel (1993); Dresel et al. (1993b).

(f) Hanford Environmental Information System.

(g) Dirkes (1990).

(h) Peterson and Johnson (1992).

LDL = less than detection level (300 - 500 pCi/L).

NS = not sampled.

Aquatic Toxicity

Very little information is available on the toxicity of tritium to salmonids. The number of rainbow trout eggs hatched was not reduced, and no delay in hatching was observed in eggs exposed to tritium irradiation (1×10^7 to 1×10^{10} pCi/L of tritium) during embryogenesis (Strand et al. 1972; as reported in Becker 1990). Exposure to 1.0×10^8 and 1.0×10^9 pCi/L tritium produced a few more abnormal embryos, but had no effect on growth or survival of rainbow trout juveniles (Strand et al. 1972 as reported in Becker 1990). Strand (1975) found that the primary immune response of rainbow trout was significantly suppressed following exposure to 1.0×10^9 and 1.0×10^{10} pCi/L tritium as tritiated water. Juvenile and adult rainbow trout that survived exposure to tritiated water were more susceptible to *Flexibacter columnaris* than nonirradiated fish (Strand et al. 1973). It should be noted that these biological effect values were much greater than the maximum values found in both the groundwater well and nearshore springs at Hanford.

3.0 Assessment of Potential Impacts of Groundwater Contaminants to Fall Chinook Salmon

Groundwater plumes containing chromium, nitrate, and tritium enter the Hanford Reach near several major fall chinook salmon spawning areas. However, the extent that these plumes interact with spawning sites is unknown. Further, distribution of salmon redds within each of the major spawning sites is patchy (i.e., the redds are not shore-to-shore or continuous within the river channel). Movement of groundwater into the Columbia River may influence the choice of spawning habitat by fall chinook salmon (Dauble and Hamisfar 1993). The preference of chinook salmon to spawn in locations with high intergravel flow may explain their tendency to aggregate in particular locations, while ignoring other areas that are superficially similar (Dauble and Hamisfar 1993; Dauble and Watson 1990; Vronskiy 1972).

Life history characteristics of the fall chinook salmon, together with the uncertainties associated with groundwater flow, are factors that must be considered in assessing potential risk to the population. For example, embryogenesis occurs in the substrate and, upon hatching, young fish (alevins) remain in the interstitial spaces of the substrate until March to May when they emerge. Embryogenesis and early life stages are typically the most sensitive life stage of fish to toxicants (McKim 1977). The period of embryogenesis and emergence for fall chinook salmon occurs during a seasonal low flow period in the Hanford Reach. The relative contribution of groundwater from bank storage discharge is increased during periods of low river flow (Luttrell et al. 1992). Consequently, the relative contribution of groundwater constituents to the river would be highest during the 3- to 4-month period that sensitive life stages of fall chinook salmon are developing, hatching, and rearing in the Hanford Reach.

Large cyclic variation in concentrations of contaminants in wells near the Columbia River indicates that the groundwater below the Hanford Site is hydraulically connected to the Columbia River (Luttrell et al. 1992). Concentrations in nearshore groundwater monitoring wells of chromium, nitrates, and tritium generally exceed the DWS and the standards recommended for the protection of aquatic life. However, with the exception of some nearshore seep areas, concentrations of chromium, tritium, and nitrate in the Columbia River are generally below the DWS and the criteria recommended for protecting aquatic life.

Concentrations of hexavalent chromium detected in groundwater monitoring wells are significantly above the EPA criteria level for the protection of aquatic life (i.e., >0.01 mg/L) in all the 100 Areas (Dresel et al. 1993b; Evans et al. 1992b; Peterson and Johnson 1992). Chromium concentrations (reported as total chromium) in shoreline seepage and spring areas sampled in 1991 exceeded the chronic toxicity level in all 100 Areas except 100-F (Peterson and Johnson 1992). These concentrations were higher than those reported for samples collected during 1988 (Dirkes 1990). Chromium also appears to be present in organic matter within shoreline seeps near the 100-H and 100-D Areas.

Our review suggests that concentrations of chromium entering the river via shoreline seepage are at levels high enough to cause chronic sub-lethal effects, such as reduced growth and physiological impairment to fall chinook salmon embryos and newly emergent fry. This may be true for any organism that spends a portion of its life cycle within the subsurface environment of the river bottom. However, shoreline seepage is highly variable and subject to radical changes in volume depending on river stage and discharge. Hence, seepage measurements do not provide adequate representation of the contaminant concentrations that enter the Columbia River via the groundwater (Peterson and Johnson 1992). At best, these data should be considered a general indicator of potential impacts to fall chinook salmon embryogenesis.

Although nitrate concentrations are low relative to toxic endpoints, they may influence chromium speciation and toxicity. Concentrations of nitrate in shoreline seeps are below the DWS, but may synergistically contribute to the toxicity of chromium. Complexation of chromium with

other unidentified inorganic or organic ligands in groundwater may also influence toxicity. Tritium is not expected to affect fall chinook salmon because of the large difference between the observed radiotoxicity and maximum seep water concentrations. Although concentrations of tritium within some plumes appear to be much higher than the DWS, levels are much lower than the radio-toxicity values reported above. Unlike nitrate, tritium has no chemical influence on toxicity or chemical speciation of chromium.

Contaminant concentrations observed to cause biological effects to fish are typically determined using laboratory toxicity tests. A direct comparison of these results to conditions existing in the Hanford Reach is complicated because the toxicity of any contaminant varies depending on the species and life stage of the organism, the physical characteristics of the organism's environment (e.g., pH, temperature, hardness, dissolved oxygen, and conductivity), synergistic or antagonistic effects, and the duration of exposure. It is difficult to simulate *in situ* exposures because these factors are variable over time. Thus, it is difficult to predict the actual impacts of contaminants present in the groundwater and subsurface water of the Hanford Reach on fish.

Cushing (1993) evaluated the potential impacts of groundwater discharge to periphyton and insect larvae and found no difference in biomass between areas upstream and downstream of known waste disposal sites. However, these studies were limited to aquatic organisms that inhabit the surface of the river substrate. No studies have been conducted to characterize conditions that fall chinook embryos or other aquatic organisms would experience within the substrate. Thus, we do not have adequate data to fully evaluate the potential impacts of chromium from groundwater sources to fall chinook salmon in the Hanford Reach.

4.0 Conclusions

Chromium is the contaminant of primary concern for aquatic populations. Information needs required for assessing risk to fall chinook salmon include methods to determine valence state of chromium and other parameters that would affect chromium toxicity such as pH, dissolved oxygen, temperature, and concentrations of other chemical constituents. Although nitrate concentrations may be low relative to toxic endpoints, potential synergistic effects with chromium are unknown. Tritium concentrations are useful in studying groundwater movement and to evaluate dilution of groundwater with river water, but are not a toxicological concern.

Data do not exist that can be used to characterize the contaminant concentrations within the interstitial waters of the river substrate near fall chinook spawning sites. Thus, it is not possible at this time to determine the potential exposure and toxicity of groundwater contaminants to fall chinook salmon. Based on long-term monitoring studies, no evidence exists of population-level impacts to fall chinook salmon from past Hanford operations. However, some uncertainty is associated with both the magnitude and spatial/temporal aspects of groundwater discharge to the Hanford Reach to evaluate potential risk to salmon during embryogenesis. Data gathered by monitoring riverbank seepage provide important information on contaminant plume characteristics, but these data do not provide quantitative information on contaminant flux or groundwater volume entering the river system (Peterson and Johnson 1992). Other investigative methods are required to produce data that can be used to model the total flow of contaminated groundwater into the Hanford Reach of the Columbia River and to assess the potential risk to fall chinook salmon. This approach would include techniques that characterize and quantify groundwater movement and contaminant discharge in areas adjacent to salmon spawning sites.

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