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Hanford Reach Project

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Accounting a Few Radionuclides from Hanford Groundwater

- Data Report 3 - Fall 1987

by
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CONTENTS

ACKNOWLEDGEMENT

Crucial data on Columbia River mixing required that river flow be held at 112,000 cfs. That river flow had to be provided from Priest Rapids Dam for durations of about 6 hours. Jim Ledgerwood of the Grant County P.U.D. provided the necessary river control amid an abundance of competing demands. We thank the Grant County P.U.D. dispatchers who delivered those flows.

Flow rate data below Priest Rapids Dam were provided by Phil Boucher of the U.S. Geological Survey (Water Resources, Pasco Field Office). These data permitted an approximate solution to a problem of statistical averaging. We also thank the U.S. Geological Survey (Tacoma Office) for critical review of our groundwater discharge concepts and measurements of 1986. Battelle Pacific Northwest Laboratory (PNL) provided tritium and iodine-129 sampler records for Priest Rapids Dam and Richland. Tim Connor (Hanford Education Action League) provided recently released iodine-129 documents.

NOTICE

GLOSSARY

prefixes used:

milli = 10^{-3}

pico = 10^{-12}

c	instantaneous concentration of a contaminant in water (pCi/L)
$\bar{C}_{down}$	time-averaged downstream sampler concentration of contaminant
cfs	cubic feet per second. 1 cubic foot = 7.48 gallons = 28.32 liters (L)
$\bar{C}_{ground}$	average concentration of undiluted contaminant in groundwater at entry to river
$\bar{C}_{river}$	time-averaged downstream concentration of contaminant, representative of river water
$\bar{C}_{up}$	average upstream sampler concentration of contaminant
Curie (Ci)	1 Curie=37 billion (=3.7×10^{10})nuclear disintegrations per second
HRM	Hanford River Mile (nominal, posted miles, downstream of Vernita Bridge)
L	liter (see also "cfs")
mile	1609 meters
n	number of elements, e.g. drogues
N_N	release of contaminant from N-Springs at N-Reactor (Ci/year)
N_{other}	release or collection of contaminant from unidentified source or sink (Ci/year)
$\bar{Q}_{ground}$	average flow rate (cfs) of undiluted groundwater entering river from HRM 28

$\bar{Q}_{river}$ annual average Columbia River flow rate (cfs) through Hanford Reach
x, y, z coordinates, respectively: downstream, across river to east, upward
α time averaging factor adjusting downstream sampler burden of contaminant
γ lateral mixing constant: 2(lateral diffusion coefficient)/(mean river current speed)
κ average river concentration of contaminant divided by average downstream sampler conc.
σ width (feet) and also standard deviation of concentration distribution

INTRODUCTION and SUMMARY

Hanford Reservation is a nuclear weapons production facility operated by the U.S. Department of Energy (USDOE). Hanford also produces, disposes of, and manages nuclear wastes.

A very small fraction of these wastes enter the Columbia River. USDOE contractors for various Hanford Operations monitor their waste releases, and Battelle Pacific Northwest Laboratories (PNL) conducts extensive research and monitoring programs which seek to assure the public that there are no significant, adverse public health, safety, or environmental impact of these releases. Proposals for continuing N-Reactor operation beyond its design lifetime after an accident at a similar reactor at Chernobyl, USSR, and for building a huge, geologic waste disposal repository have brought the first critical, public attention to Hanford.

The Hanford Reach Project (HRP) was established by SEARCH Technical Services (a private, scientific consulting company) in 1985 to characterize groundwater pathways from Hanford to the Columbia River. HRP is an independent public interest project, funded almost entirely from a consulting business based on fire and accident investigations. The purpose of HRP is to provide a technical basis for reasonable management of Hanford facilities so that present and future contamination of Hanford groundwater does not unduly threaten the viability of the Columbia River.

SEARCH believes that facilities which produce or dispose of wastes at Hanford can only be managed rationally if the waste streams are identified and characterized and if their contaminants are accounted through the Hanford environmental system. Since USDOE does not account its waste disposals through the groundwater/river system at Hanford, HRP has undertaken preliminary accounting of a few Hanford radionuclide flows through the aquifers and then down the river to Richland. The accounting involves measurement of radionuclide concentrations, identification of pathways, measurement of flow rates, and, finally, calculation of total activities. Accounting at several locations through the pathway system introduces checks of measurements and pathway concepts.

Fortunately, almost all of the information required to account some radionuclides through Hanford groundwater and the river is already available. PNL collects many of the radionuclide concentration data required for this accounting as part of the monitoring of the Hanford site and the river upstream and downstream of Hanford. The little remaining field work necessary for a preliminary accounting involves flow measurements of a major groundwater entry at Hanford River Mile (HRM)

28 and measurement of the mixing rates of river-borne contaminants between their entry at HRM 28 and PNL's downstream samplers at HRMs 42.5 and 47.6. That field work and the attendant calculations for the accounting of tritium, iodine-129, and comments on other radionuclides are the subject of this data report. Figure 1, below, shows the some of the important locations at Hanford:

Fig. 1. HANFORD SITE - MEASUREMENT LOCATIONS

PNL has three river *monitoring* stations. The upstream sampler is located at Priest Rapids Dam, which is upstream of all Hanford Operations. Therefore, samples of river water collected at Priest Rapids Dam are free of any contamination from Hanford and provide reference levels of river background.

There are two *downstream* sampler stations. The first downstream sampler is located at DOE's fabrication facility at "300" Area which is on the west bank of the river at HRM 42.5. The following radionuclides are sampled at HRM 42.5:

element	isotope
cobalt-60	
niobium	-95
zirconium	-95
ruthenium	-106
iodine-129	
	-131
cesium	-134
	-137

cerium -144
plutonium -238
-239,240

Any pathways of these radionuclides, which enter the river between Priest Rapids Dam and HRM 42.5 might be detected from these monitoring results. Any pathways which enter the river downstream of HRM 42.5 would not be detected, of course. Thus, the data from this first downstream sampler are only partly downstream and provide only weak assurances of the impacts of Hanford Operations on the river.

The second downstream sampler station is at the Richland water pumping station at HRM 47.6. The following radionuclides are routinely sampled at this downstream sampler which is clearly downstream of all Hanford Operations:

element	***isotope***
tritium (hydrogen)	-3
strontium	-89
	-90
uranium	-234
	-235
	-238

All the radionuclides on these two downstream sampler lists are also monitored by PNL at the upstream sampler at Priest Rapids Dam. The amount of each of these particular radionuclides entering the river from Hanford can be estimated by subtracting the upstream concentration from the downstream concentration and multiplying by the river flow rate. This difference may then be attributed to problems or insensitivity of the sampling system, radionuclide-bearing pathways from Hanford Operations, or "other pathways." The possibility of "other pathways" does not have to be considered seriously for these sampled radionuclides because the Hanford Reach is only 45 miles long and there are no other known sources of radionuclides in sufficient amount to affect concentrations in the river.

In order for the downstream-upstream differences to be useful for preliminary accounting, the relations between radionuclide concentrations in PNL's samples and actual concentrations of the same radionuclides in the river must be understood. That is, the sampler data are not representative of the river for two reasons which are described in this report and detailed in the appendices.

To begin the accounting, in 1986 HRP measured a lower bound on the flow from the largest groundwater pathway at its point of entry into the Columbia River at HRM 28. That lower bound was 6.3 cfs (=178 L/sec).

In 1987, the rate that contaminants mix laterally across the river was measured from this point of entry at HRM 28 downstream to HRM 36.8. These results were then extrapolated to PNL's downstream river sampler sites at

HRMs 42.5 and 47.6. This has allowed PNL's sampler data to be related to average concentrations in the river water passing those samplers. The resulting, preliminary accounting of some radionuclides in the groundwater/river system are as follows:

- **Tritium** enters the Hanford Reach almost entirely from two sources: Groundwater at HRM 28 (1600 Ci annually) and primary cooling water from N-Reactor (200 Ci annually).

- **Iodine-129** enters the Hanford Reach from at least two sources: Groundwater at HRM 28 (about $1x10^{-3}$ Ci annually) and an unreported source ($2x10^{-3}$ to $6x10^{-3}$ Ci annually). Correlation of this unreported entry with monthly river flow suggests entry of the unreported iodine-129 from confined aquifer(s) below the river. The original source of this unreported iodine-129 is likely groundwater from near "200 West" Area which has contaminated a high-speed pathway in one of the confined aquifers under Hanford. The characterization of this pathway should be a high priotity effort at Hanford.

- **Technetium-99** in groundwater entering the river at HRM 28 probably elevates the concentration at the downstream sampler by a detectable amount (0.02 pCi/L). Thus, accounting of technetium-99 in the groundwater/river system is probably feasible if this radionuclide is added to the downstream-upstream analyses.

- **Other radionuclides**, including strontium -89 and-90, cobalt-60, cesium-137, and some uranium and plutonium isotopes, remain unaccounted. That accounting should be a high-priority item for science at Hanford.

BACKGROUND

Hanford operations routinely discharge radionuclides in waste water to the soil. These discharges become contaminated groundwater which migrates toward the Columbia River. During migration many of the contaminants are sorbed onto soils and those with short lifetimes decay. Some of the contaminants which remain enter the Columbia River and are carried downstream to Richland and beyond. These radionuclides affect the present and future usability of both the aquifers under Hanford and the Lower Columbia River.

At present, most of the radionuclides which reportedly enter the Columbia River pose little or no particular public health or safety problem. The important unresolved questions concerning Hanford's contaminated groundwaters are -

- proper technical management of the wastes
- management accountability to the public

- the possibility of unreported problems which are serious
- damage to agricultural, environmental, and recreational resources
- future projects
- future impacts

These questions involve dangerous wastes at the production reactors, processing wastes at "200" and "300" Areas, and proposed projects such as a high-level waste repository. Those wastes must be isolated from the accessible environment (air and water) and from the public (direct exposure, water and air consumption, and agriculture) for sufficient time for the radionuclides to decay.

Once a groundwater problem has developed, it may be difficult or impossible to correct. Thus, the groundwater regimes at Hanford must be understood technically well enough to form a basis for reasonable engineering management for the short term and for the next several million years. As a practical matter, this basis does not exist.

In order to build a basis for management of Hanford's wastes, SEARCH advocates an accounting of the flows of those wastes which are released to the Hanford environment. That accounting may be begun by examining one *easy-to-review pathway* and one *conspicuously-large-and-important pathway*. The easiest pathway to review is N-Springs at HRM 9 at N Reactor which springs are fed from documented, primary cooling water discharges to nearby Trenches 1301-N and 1325-N. N-Springs carry one billion gallons (=3.6x10^9 liters) of water contaminated with strontium-90 and other radionuclides to the river annually [1].

Hanford's conspicuously-large-and-important groundwater pathway connects waste water disposal from the "200 East" Separations Area to shoreline springs at Hanford River Mile (HRM) 28. See Appendix C for discussion of pathway analysis and the location map, Fig. 2, below:

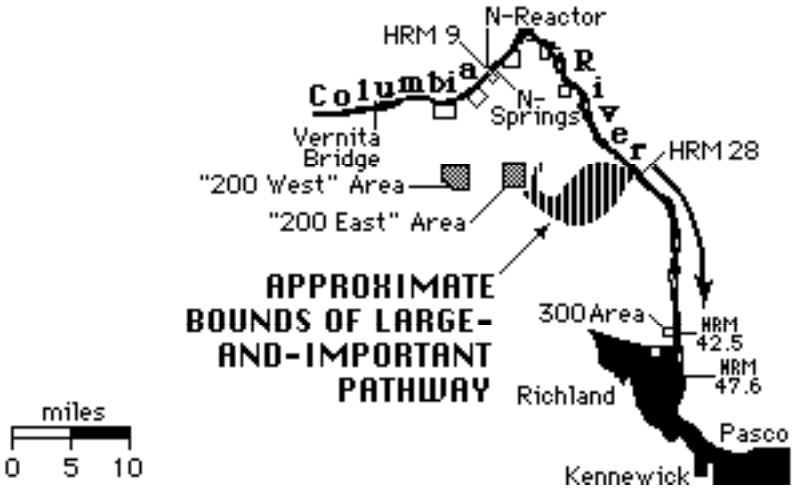


Fig. 2. Location map

This pathway carries two billion gallons (=7x10^9 L/year) of groundwater contaminated with tritium, iodine-129, technetium-99, cesium-137, uranium-234, etc., to the river. This pathway which has no NPDES permit releases twice the volume of N-Springs, for which USDOE does have a permit for non-radiological waste disposal to the river. (The non-radiological constituents of the water entering the river at HRM 28 have not been reported.)

By way of comparison, the Columbia River flow is about 380 trillion gallons (=1.0x10^{14} L) per year. That is, even the largest known groundwater pathway which discharges at HRM 28 is diluted by a factor of at least **14,000** (=1.0x10^{14} / 7x10^9) after entering the river. This large dilution factor ameliorates the effect of contaminated groundwater entry on the quality of Columbia River water .

Figure 3, below, shows results of a study by Brown and Haney which suggest that tritium-contaminated groundwater had already reached the river from the "200 East" Separations Area as early by 1960.

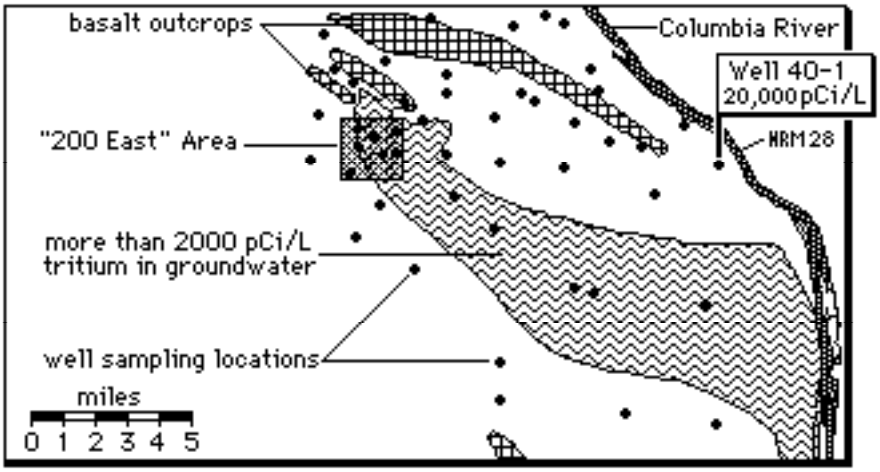


Fig. 3. Brown and Haney's estimated tritium plume in 1960 [2]

Although Brown and Haney missed the major part of this pathway which was subsequently found to enter the river at HRM 28, that pathway was actually observable as a concentration of 20,000 pCi/L tritium appearing at Well 40-1, as shown in the figure, above.

After another 20 years, the broad outline of the pathway to HRM 28 had been identified by a tritium plume ranging from 30,000 to 300,000 pCi/L arriving at the river, shaded in the inset to Fig. 4, below.

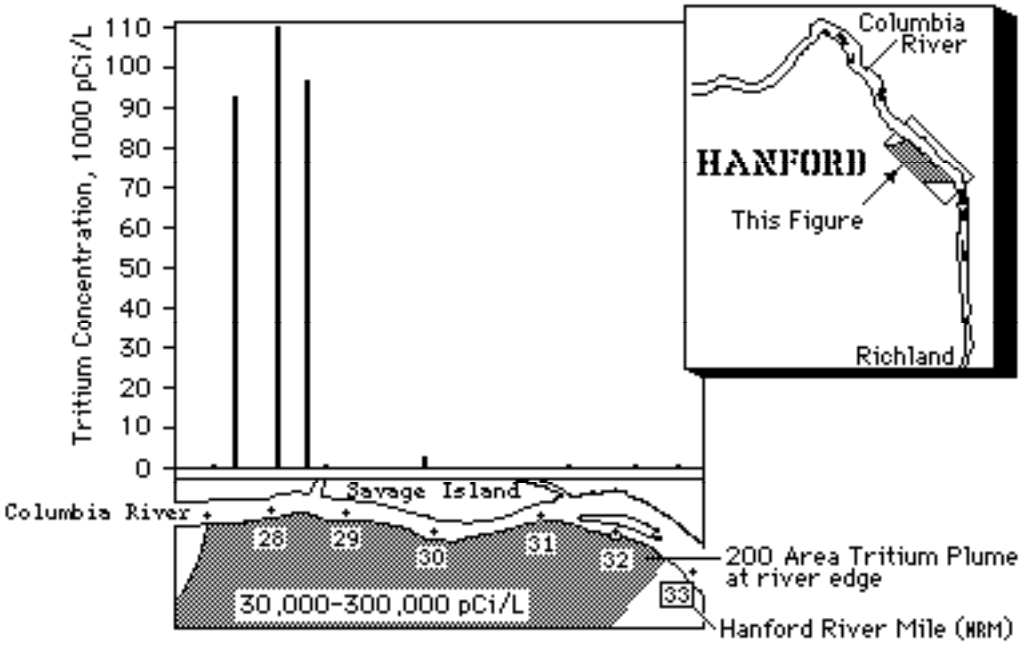


Fig. 4. Localized contaminant discharge to the river at HRM 28 [3]

PNL sampling of shoreline springs in 1983 and 1984 showed that this pathway was much more localized than computer models [4] and well data (shaded area in Fig. 4) had previously suggested. PNL's shoreline springs data are sketched as vertical bars above the HRMs where the springs are located. The high spikes of tritium were all seen in springs very close to HRM 28.

The implied narrower width of the groundwater discharge would have implied a narrower groundwater pathway than had been thought. This, in turn, would have implied a shorter-than-predicted travel time from "200 East" Area facilities to the river. PNL did not pursue these implications.

In order to estimate the actual speed of the groundwater movement and the actual volume and importance of this localized discharge to the groundwater flow system, the ***average*** outflow of groundwater (in cubic feet per second=cfs) had to be determined.

Because groundwater flows under Hanford are little affected by the scant rainfall and the water table has changed little over the last decade [5], the groundwater must flow at its average rate if the river is held at its average level so that transient flows of river water into or out of the river bank have died out. In order to satisfy this logical requirement, the Army Engineer Corps, Bonneville Power Administration, and Grant County P.U.D. cooperated to hold Columbia River level at HRM to a constant, average value for 60 hours, as shown in Fig. 5, below.

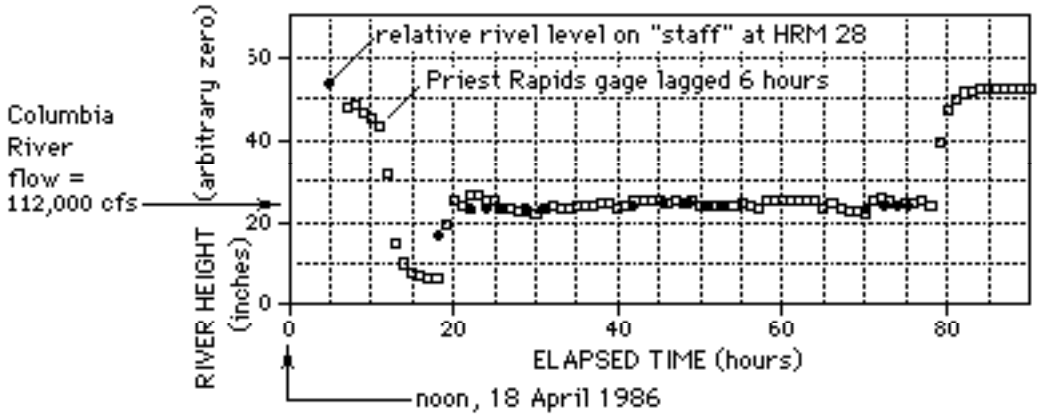


Fig. 5. River levels at HRM 28 for groundwater flow measurement

A longer period of constant river level would have been required to assure that the system had completely come to equilibrium [6]. However, SEARCH's results, Fig. 6, below, suggest a near-equilibrium discharge of at least 6.3 cfs from only 852 feet of shoreline at HRM 28.

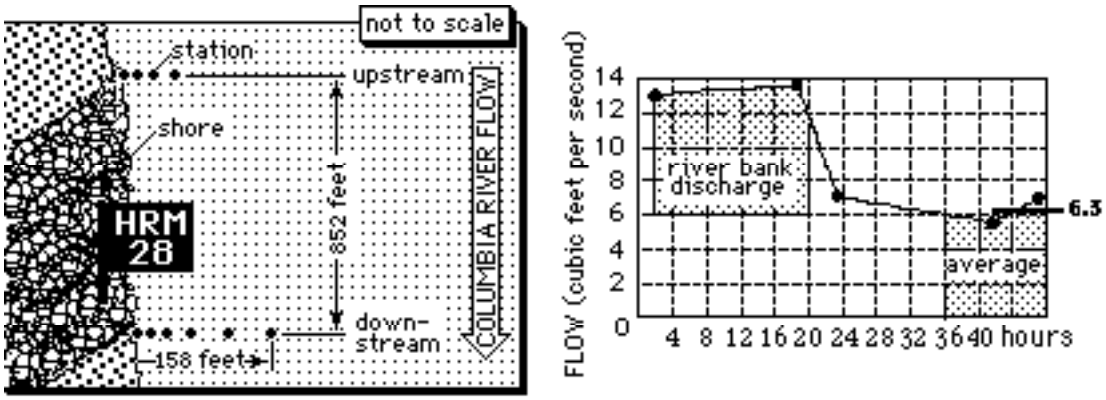


Fig. 6. Arrangement and results of groundwater flow measurement

This flow rate is much greater than Hanford hydrologists had predicted and implies much shorter travel times for contaminated groundwater to migrate from disposal sites near "200 East" Area to the river.

The SEARCH study was designed to provide a *lower bound* measurement of average groundwater discharge in order to resolve the question of the existence of a relatively high speed groundwater pathway from disposal sites to the river. That is, the study was designed to *under*estimate the flow. Unfortunately, such an underestimate is not good estimate of the average groundwater discharge from this pathway: All SEARCH could say with confidence was that the average discharge is probably greater than 6.3 cfs. This implies a travel time of roughly 3 to 5 years along the fastest pathway in which this groundwater travels ... rather than 30 years suggested by PNL's computer model.

This much shorter travel time further implies that the groundwater following the fastest pathway has less time for sorption and radioactive decay to remove contaminants before entering the river than had previously been thought. Indeed, PNL and joint (PNL, Oregon, Washington, Greenpeace) sampling of shoreline springs has already provided evidence of iodine-129, technetium-99, and strontium-90 contamination at HRM 28.

In order to estimate the impact of this discharge, the *true* average discharge was required rather than a lower bound measurement. SEARCH reviewed its study biases in a semiquantitative fashion [7] to estimate an actual average groundwater discharge of 10 cfs along this pathway. By the end of 1986, SEARCH was comparing the tritium burden in this estimated discharge to the elevation of tritium concentrations in river water passing the Hanford Reach, as reported by PNL samplers at Priest Rapids Dam upstream and Richland downstream [8].

ACCOUNTING EQUATION

In order to determine the magnitude of the groundwater discharge at HRM 28 and to begin to account tritium and other radionuclides entering the Hanford Reach, several factors must be considered. The rate of groundwater entry can be estimated crudely the difference in radionuclide concentrations between the downstream samplers at HRMs 42.5 and 47.6 and the upstream sampler at Priest Rapids Dam, Fig. 7.

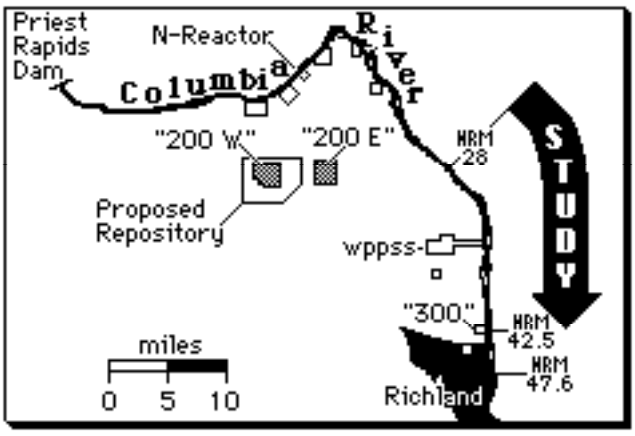


Fig. 7. Study location for river mixing

In particular, the annual average (indicated by an overbar) concentration added ($\bar{C}_{down}$-$\bar{C}_{up}$) to river samples between the upstream ($_{up}$) sampler at Priest Rapids Dam and the downstream ($_{down}$) sampler at HRM 47.6, multiplied by average river flow rate ($\bar{Q}_{river}$) would roughly equal the average amount ($\bar{C}_{ground}\bar{Q}_{ground}$) of tritium in groundwater discharged at the west side of the Columbia River at HRM 28 plus the amount added by N-Reactor (N_N) plus any

other major sources (N_{other}). $\bar{C}_{ground}$ is the annual average concentration of tritium (or other contaminant) in groundwater which is undiluted by river water, and $\bar{Q}_{ground}$ is the average groundwater flow rate. Material which would be lost to sediments and radioactive decay in the river would be included as a negative quantity in the "other sources (N_{other})" term.

Once SEARCH proposed the concept of downstream-upstream accounting, questions of the adequacy of the downstream-upstream sampler data arose: (1) SEARCH noticed that groundwater tended to flow from shoreline springs into the river only when river level was falling or quite low. This implied that the true amount of any contaminant in the river might differ from the average concentration multiplied by the average annual river flow. This question of *averaging* is described later and detailed in Appendix B. (2) PNL noted that the sampler data were not designed for such an accounting and that the relation between sampler data and representative contaminant concentrations in river water was unknown.

The U.S. Geological Survey concurred that the river would not completely mix tritium across its width from the west shoreline discharge at HRM 28 before the tritium was sampled by PNL at the west shore at HRM 47.6. The Geological Survey estimated of the relation between sampler concentrations and representative river water concentrations [9] based on general models [10] and data collected upstream and downstream of the Hanford Reach of the Columbia River. The Geological Survey concluded that SEARCH had overestimated (at 2500±400 Ci/year) the groundwater contribution to the river by a factor of about three.

To the extent that tritium (or other contaminant) is incompletely mixed into the river water between HRM 28 and HRM 47.6, the downstream sampler concentration $\bar{C}_{down}$ must be adjusted. This adjustment is accomplished by the introduction of a factor κ which is defined as the ratio of concentration ($\bar{C}_{river}$-$\bar{C}_{up}$) added to representative river ($_{river}$) water passing down the Hanford Reach to the concentration ($\bar{C}_{down}$-$\bar{C}_{up}$) difference between the downstream and upstream samplers:

$$\kappa=(\bar{C}_{river}-\bar{C}_{up}) / (\bar{C}_{down}-\bar{C}_{up})$$

The value of κ must be determined experimentally, as described in the next section and in Appendix A.

Similarly, the product of the annual difference in sampler concentration ($\bar{C}_{down}$-$\bar{C}_{up}$) with the average river flow $\bar{Q}_{river}$ is not exactly the actual, annual addition of tritium between the samplers. The actual addition of tritium is the average of the product $\overline{(C_{down}-C_{up})Q_{river}}$ instead of the product of the averages. This effect can be accommodated by a time averaging factor α defined as:

$$\alpha = \overline{(C_{down}-C_{up})Q_{river}} / [(\bar{C}_{down}-\bar{C}_{up})\bar{Q}_{river}]$$

This time averaging factor α must also be determined from measurements and observations.

The Accounting Equation for each radionuclide entering the Hanford Reach of the Columbia River may then be written as follows:

$$\alpha\kappa(\bar{c}_{down}-\bar{c}_{up})\bar{Q}_{river} = \bar{c}_{ground}\bar{Q}_{ground} + N_N + N_{other} \qquad (1)$$

If Eq. (1) is used to account tritium, either SEARCH's flow rate estimate of $\bar{Q}_{ground}$ = 10 cfs can be checked or an unknown source of tritium (N_{other}) to Hanford Reach can be identified. In the event that the measurements of tritium in the river and reported releases from N-Reactor satisfy Eq. (1) with $\bar{Q}_{ground}$ not much greater than 10 cfs, the accounting of tritium entering the Hanford Reach may be considered to be adequately accounted with no other *significant* sources of tritium present. That is,

$$N_{other} = 0$$

As will be seen shortly, the various river data, in addition to the reported tritium releases from N-Reactor, yield a groundwater inflow rate at HRM 28 of $\bar{Q}_{ground}$=8 cfs, calculated from Eq. (1).

MIXING

In order to account the water which enters the west bank of the river at HRM 28 and continues downstream to Richland, the rate of lateral (cross-river) mixing must be known. Since tritium-contaminated groundwater enters near the west shoreline at HRM 28 and the downstream sampler at HRM 47.6 is also near the west shoreline, incomplete mixing of tritium across the river causes the tritium concentrations $\bar{c}_{down}$ in downstream samples to overestimate representative tritium concentrations $\bar{c}_{river}$ in river water at Richland. In order to use Eq. (1), the ratio $\kappa=(\bar{c}_{river}-\bar{c}_{up}) / (\bar{c}_{down}-\bar{c}_{up})$ must be determined.

This concentration mixing ratio κ could be measured directly by bringing the whole groundwater/river system into near-equilibrium at a suitable river flow rate and then measuring the cross-river profile of tritium concentrations at HRM 47.6. Response of nitrate (which presumably similar to response of tritium) to changing river level has been measured [11], Fig. 8, below.

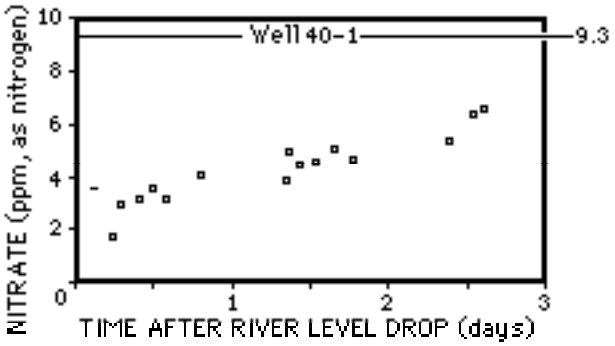


Fig. 8. Nitrate concentration response time for spring at HRM 28

These data suggest that near-equilibrium in tritium concentration would be obtained after roughly 5 days of constant river level. Because of competing requirements for regulated river flows, such a long duration of flat water is difficult to obtain.

Perhaps the most common method of measuring lateral mixing rates would entail discharging dye at a constant rate into the river at HRM 28. Dye concentration would then be measured across the river at HRM 47.6. Such a dye study would still entail degradation of river water quality for a period of a day or two.

Because of these problems with the usual methods of measuring lateral mixing, SEARCH opted for a *drogue* study. Five drogues are sketched in the river, offshore of an observer in Fig. 9, on the next page.

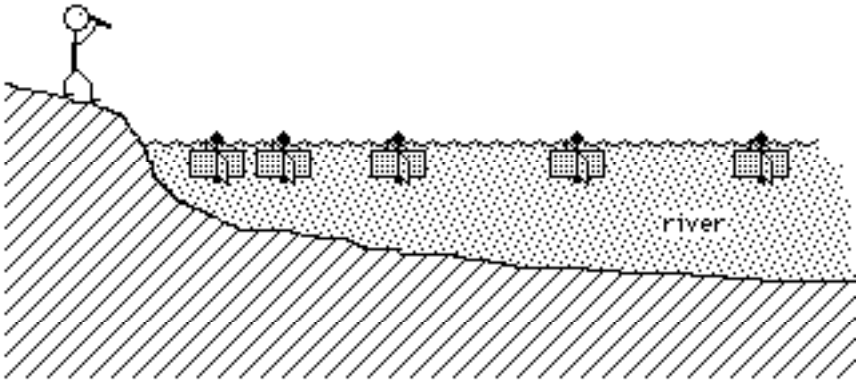


Fig. 9. Drogues

A drogue has a large surface area which follows water surrounding it, much as dye tags a parcel of water. A drogue study can be conducted piecemeal, following an array of drogues down river a couple of miles at a time, thus minimizing requirements for protracted river control. Drogues do not deteriorate

water quality. Furthermore, drogues can be numbered so that the identity of tagged parcels of water remains known. This provides some additional information. Disadvantages include a small number of data points, unwanted sensitivity to some water motions, and the necessity of jointing parts of the study and the necessity of extrapolations.

The drogue study, conducted in July and August 1987, is detailed in Appendix A. The resulting concentration mixing ratios between the groundwater discharge at HRM 28 and downstream samplers located at HRMs 42.5 and 47.6 are, respectively:

$$\kappa = \bar{c}_{river}/\bar{c}_{down} = 0.52,\ 0.57 \qquad (2)$$

These values of κ are applied to Eq. (1) for the accounting of tritium and other radionuclides.

TRITIUM ACCOUNTING

With a concentration mixing rate κ=0.57 for the reach between HRMs 28 and 47.6, Eq. (1) can be solved once the time averaging factor α

$$\alpha = \overline{(c_{down} - c_{up})Q_{river}} \,/\, [(\bar{c}_{down}-\bar{c}_{up})\bar{Q}_{river}]$$

is evaluated. This factor α adjusts for the observation that groundwater enters the river mostly when river level is low. Thus, the river flow rate Q_r when most of the tritium enters the river is less than the annual average flow rate $\bar{Q}_r$. The average product $\overline{(c_{down} - c_{up})Q_{river}}$ of instantaneous concentration difference (c_{down}-c_{up}) and river flow Q_{river} differs from the product of the averages: ($\bar{c}_{down}$-$\bar{c}_{up}$)$\bar{Q}_{river}$. The value of α depends on the response time of the groundwater discharge pathway entering the river and on the river flow spectrum (that is, how much river level changes over various time scales).

The time averaging factor for the shoreline springs at HRM 28 is estimated to be α=0.70 in Appendix B. Thus, the product of two coefficients on the right side of Eq. (1) for tritium sampled at HRM 47.6 is

$$\alpha\kappa = (0.70)(0.57) = 0.40$$

The contribution N_N of tritium from N-Reactor for Eq. (1) is the reported release. These N_N data, river flow, and "other releases" reported were obtained from the respective PNL annual surveillance and monitoring reports; see Table 1, below:

Table 1. ANNUAL RIVER FLOW AND TRITIUM DIFFERENCE

year:	1981	1982	1983	1984	1985	1986
$\bar{c}_{down}$-$\bar{c}_{up}$ (x10^{-12} Ci/L)	30	60	30	40	40	50

$\bar{Q}_{river}$ (x10^{14} L/year)	1.17	1.21	1.19	1.00	0.97	0.96
N_N (Ci/year)	82	360	180	140	270	220
Other releases (Ci/year)	0	0	0	0	0	0

For these six years, the average of $(\bar{c}_{down}-\bar{c}_{up})\bar{Q}_{river}$ for tritium is 4500 Ci/year. The right side of Eq. (1), then, must equal 0.40x4500 Ci/year =

1800 Ci/year ... added tritium passing HRM 47.6.

Of this added tritium passing HRM 47.6, an average of 209 Ci/year originates as N-Reactor. Setting the tritium concentration $\bar{c}_g$ in undiluted groundwater entering the river at HRM 28 at the value in Well 40-1 (2.30x10^{-7}Ci/L), the mean volume of groundwater entering at HRM 28 may be treated as the unknown and calculated from Eq. (1):

$$\bar{Q}_{ground} = (1800 - 209 \text{ Ci/year})/2.30\text{x}10^{-7}\text{Ci/L} = 6.92\text{x}10^{9} \text{ L/year} = 7.7 \text{ cfs}$$

This estimate of the mean flow $\bar{Q}_{ground}$=7.7 cfs (=218 L/sec) of groundwater from HRM 28, which is based on undiluted groundwater tritium concentration $\bar{c}_{ground}$ and river concentrations $(\bar{c}_{down}-\bar{c}_{up})$, is in good agreement with the direct, lower bound measurement of 6.3 cfs obtained at HRM 28 and also with SEARCH's subsequent estimate of $\bar{Q}_{ground}$=10 cfs. These three estimates of groundwater entry from HRM 28 satisfy:

$$\bar{Q}_{ground} = 8 \pm 2 \text{ cfs} \quad . \qquad (3)$$

This result accounts for all the tritium entering the Columbia River along the Hanford Reach to an estimated accuracy of 25 percent. Of the accounted tritium, about 90 percent enters with groundwater at HRM 28 and about 10 percent is released from N-Reactor at N-Springs.

The major implication of this accounting of tritium through the Hanford Reach is that no additional sources or sinks of tritium are required to satisfy the Accounting Equation (1). This implies, for example, that the broad tritium plume entering the river near HRM 36, suggested by Brown and Haney (Fig. 3), is unlikely to be a significant pathway.

IODINE-129 ACCOUNTING

Equation (3) allows the amount of any radionuclide of known concentration in groundwater at HRM 28 to be subtracted from downstream sampler data per Eq. (1). This subtraction is a first step toward the accounting of several radionuclides through the groundwater/river system at Hanford.

Iodine-129, with a halflife of 16,000,000 years, is the only contaminant for which this subtraction presently allows specific conclusions. Iodine-129 is also the only radionuclide which PNL routinely reports at much higher concentrations $\bar{c}_{down}$ downstream of Hanford than upstream $\bar{c}_{up}$. Typically, downstream concentrations are close to 9 times as large as upstream concentrations.

Since 1981, PNL has attributed the increased downstream concentrations of iodine-129 to

> seepage of ground water from the unconfined aquifer underlying the Site into which process cooling water and low-level liquid wastes have been discharged at the 200 Areas [12].

This suggests that much of the increase in iodine-129 burden of river water passing down the Hanford Reach of the Columbia River is attributable to the HRM 28 springs which have been shown to account the tritium burden. Reference data from the PNL annual reports are summarized in Table 2, below, which provides part of the basis for subtraction of the impact of the HRM 28 shoreline springs.

Table 2. Iodine-129 difference in river, well concentrations, river flow, and reported releases

year:	1981	1982	1983	1984	1985	1986
$\bar{c}_{down}$-$\bar{c}_{up}$ ($\times 10^{-17}$ Ci/L)	3.9	6.3	5.1	6.2	7.9	9.1
C_{ground}: Well 41-1 ($\times 10^{-12}$ Ci/L)	-	-	-	-	0.25	0.21
$\bar{Q}_{river}$ ($\times 10^{14}$ L/year)	1.17	1.21	1.19	1.00	0.97	0.96
Other releases (Ci/year)	0	0	0	0	0	0

These few concentration c_{ground} data for Well 41-1 near HRM 28 can be supplemented by measurements from adjacent Well 40-1 in 1979, which had an average iodine-129 concentration of 0.21 pCi/L and an average tritium concentration of 185,000 pCi/L [12]. Thus, it appears reasonable to set the undiluted groundwater concentration of iodine-129 at 0.21 pCi/L for the 1981-1986 period.

Unlike tritium which is not removed from groundwater by chemical or biological processes because it is part of the groundwater, iodine is subject to removal [14]. Indeed, enough iodine-129 may be removed from the groundwater between the nearshore wells and the shoreline springs to affect the accounting. Such a removal of iodine would diminish both the iodine discharged to the river at HRM 28 and the amount of iodine sampled downstream from the discharge at HRM 28. Table 3, on the next page, shows that the iodine/tritium ratio decreases closer to the river.

Table 3. Iodine-129 and tritium approaching the shoreline

Well or Spring	iodine-129 (pCi/L)	tritium (pCi/L)	**iodine/tritium**	Ref.
Well 42-12 (two miles west)	0.68	330,000	**20**x10^{-7}	[13]
Well 41-1	0.21	230,000	**9**x10^{-7}	[15]
Shoreline Spring 28-2	0.062	110,000	**5**x10^{-7}	[16]

These few data hint that roughly 5/9 of the iodine at nearshore Well 41-1 arrives at the bank of the Columbia River at HRM 28. It seems reasonable to suppose that between 5/9 of the iodine-129 at Well 41-1 and all of the iodine at Well 41-1 enter the river. This provides a range of effective iodine-129 concentrations for the accounting:

$$[(5/9)0.21=] \quad 0.12 < \bar{c}_{ground} < 0.21 \text{ pCi/L}$$

With the mean groundwater flow rate given by Eq. (3) as $\bar{Q}_{ground}$=8±2 cfs (=7.1±1.8 x10^9 L/year), the range of annual influx of iodine-129 from shoreline springs at HRM 28 is expected to lie between the lower estimate of concentration times the lower estimate of groundwater flow and the product of the higher estimate of each:

$$0.6\text{x}10^{-3} < \bar{c}_{ground}\bar{Q}_{ground} < 1.9\text{x}10^{-3} \text{ Ci/year} \ .$$

Inasmuch as there are no reported releases of iodine-129 from any Hanford Operation (such as N-Reactor) to the river, this discharge of iodine-129 from groundwater at HRM 28 is the total accountable addition of iodine-129 to the Hanford Reach.

This release adds an average apparent burden $\bar{c}_{ground}\bar{Q}_{ground}/\alpha\kappa$ to the downstream sampler according to Eq. (1). Equation (2) gives the concentration mixing factor between HRMs 28 and 42.5 (the iodine sampler location) as κ=0.52. Assuming that the iodine-129 from HRM 28 enters the river at the same river stage as tritium, α=0.70. The maximum range of the *apparent* downstream river burden *from shoreline springs at HRM 28* would then be

$$1.6\text{x}10^{-3} < \bar{c}_{ground}\bar{Q}_{ground}/\alpha\kappa < 5.2\text{x}10^{-3} \text{ Ci/year} \ .$$

This contribution from the shoreline springs at HRM 28 compares to the annual mean of the downstream-upstream difference in iodine-129 concentration-times-river-flow from Table 2. That apparent river burden is

$$(\bar{C}_{down}-\bar{C}_{up})\bar{Q}_{river} = 6.8\text{x}10^{-3} \text{ Ci/year} \quad .$$

That is, between 24 and 76 percent of this apparent river burden of iodine-129 can be accounted by groundwater entry from shoreline springs at HRM 28.

This disparity between the apparent river burden ($6.8\text{x}10^{-3}$ Ci/year) and the known entry of iodine-129 (ranging from $1.6\text{x}10^{-3}$ to $5.2\text{x}10^{-3}$ Ci/year) implies that the tritium-contaminated springs at HRM 28 cannot account for all of the iodine-129 observed downstream of Hanford. There must be an unreported. source of iodine-129 in the Hanford Reach. Based on this preliminary accounting of iodine-129 from groundwater at HRM 28, it appears that the unreported source is probably responsible for half or more of the iodine-129 added to the Hanford Reach of the Columbia River.

SOURCE OF UNREPORTED IODINE-129

There are at least two possible sources of the iodine-129 which is apparently entering the river elsewhere than at HRM 28: (1) entry from another part of the unconfined aquifer somewhere upstream of the downstream sampler at HRM 42.5 and (2) intrusion from a confined aquifer. These candidates are evaluated as follows:

(1) Iodine-129 is lost more readily from groundwater to chemical, biological, and radioactive decay processes than is tritium, as shown in Table 3. Since the groundwater discharge from HRM 28 apparently represents the fastest pathway from contaminant releases at "200 East" Area to the river, the iodine-to-tritium ratio is expected to be largest for groundwater discharged to the river near HRM 28. (This concept accords with available well data.) Furthermore, any such iodine pathway would be expected to be remarkable in the data from the hundreds of wells in the unconfined aquifer which PNL routinely samples and reports.

One possible exception would be an unreported release of iodine-129 from a source so close to the shoreline that its effect on groundwater would not be detected in PNL's samples from on-site wells. This possibility is effectively ruled out by the observation that intrusion of iodine-129 into the river is clearly dependent of river flow rate, as shown on the logarithmic plot, Fig. 10, below.

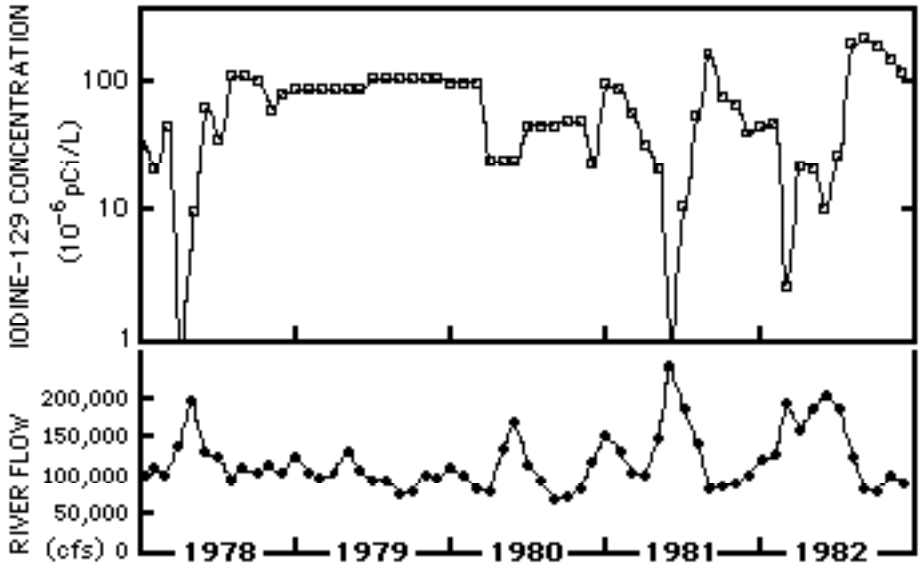


Fig. 10, Iodine-129 added to Hanford Reach vs. river flow [13]

Notice the low concentrations of iodine at the times of peak river flows in 1978, 1980, at the beginning and middle of 1981, and twice in 1982. Likewise, very high concentrations are seen to be associated with low river flow rates. The

effect of monthly river flow on monthly, downstream iodine-129 concentrations was noted by PNL as early as 1982 [13].

The reason for making such a point of this relation between monthly-averaged iodine-129 concentration and monthly-averaged river flow is that it is very difficult to identify any source from Hanford Operations which would discharge large quantities of iodine-129 almost directly into the river but which would be responsive to changes in river flow for periods as long as a month. Thus, it is unlikely for some discharge either from the unconfined aquifer or directly to the river to introduce the necessary iodine to account for the downstream sampler data.

(2) The response time of groundwater from a more or less confined aquifer entering the river bed would depend on interconnections between aquifers. In order to examine the response time of iodine-129 entry into the river versus river flow, in a little more detail, the monthly averages of nominal iodine-129 burden $(\bar{c}_{down}-\bar{c}_{up})\bar{Q}_{river}$ against monthly average river flow $\bar{Q}_{river}$ were scatter plotted for the period, 1978-82, as in Fig. 11, below.

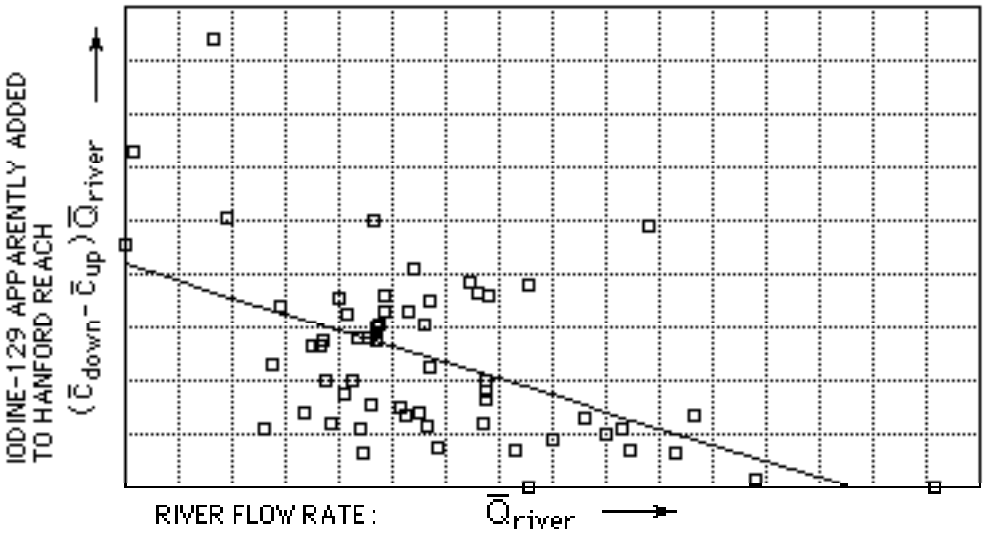


Fig. 11. Scatter plot of monthly iodine-129 burden vs. river flow

The scatter data were least squares fit, as shown, with a correlation coefficient of 0.51. This suggests that about half of the river burden of iodine-129 might be attributed to a source with a river-response time as long as one month. As described in Appendix B, the response time of the unconfined aquifer near HRM 28 is a very few days. No alternatives to a confined aquifer source have been found which would explain the iodine-129 contamination at this long response time.

The possibility of a confined aquifer source of iodine-129 is supported by reduced pressure measured in the (first) confined aquifer below the Columbia River bed [18]. That is, groundwater in the confined aquifer would tend to flow toward the river from both the west (Hanford) side and the east. The obvious

direction for this converging groundwater to continue to migrate is vertical. Since the lower Hanford Reach has about 30 feet [19] less "pressure" (head) than does the first confined aquifer, the groundwater from the first confined aquifer presumably migrates vertically upward.

The iodine/tritium ratios for both unconfined groundwater near the "200" Areas and confined groundwater many places under Hanford have iodine/tritium ratios which are so much higher than at wells 40-1 and 41-1 at HRM 28 that tritium accounting cannot be affected by any conceivable discharge of these waters to the river. The highest iodine/tritium ratios are found near the "200" Areas, in confined aquifer(s), as suggested by a few well data in Table 4 [20], with locations shown in Fig. 12, both on the next page.

Table. 4. Iodine-129 and tritium concentrations in five wells

Well No.	Iodine-129 (pCi/L)	Tritium (pCi/L)	Iodine/tritium ratio
41-1 (unconfined)	0.214	230,000	$9x10^{-7}$
35-70 (unconfined)	101.8	1,600,000	$6x10^{-5}$
DH-8 (confined)	0.041	340	$1x10^{-4}$
31-31P (confined)	4.1*	310	$1x10^{-2}$

* June 1975, iodine datum from >600-foot depth.

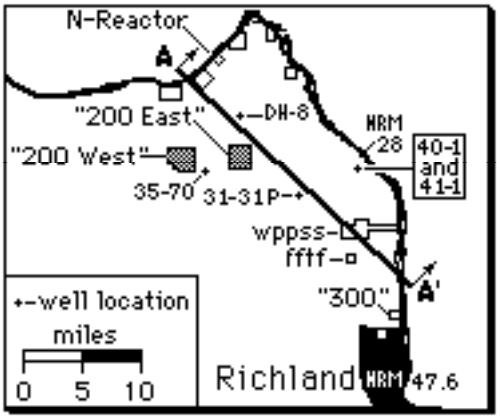


Fig. 12. Well location map for Table 13 and section A-A' location

These well data point toward the "200" Areas as the only locations with (1) high enough iodine-129 concentrations with (2) large enough waste water discharges to carry millicuries of iodine-129 to the Columbia River annually.

Waste water disposals from the "200" Areas have been large enough to elevate the water table under the disposal ponds to greater pressures than in the first confined (Rattlesnake Ridge) aquifer. Table 5 shows that the hydraulic head difference between the unconfined and confined aquifers underlying "200 West" Area reversed direction between 1944 (hindcast) and 1975 [21].

Table. 5. Reference head elevations (feet, Mean Sea Level)

Area	1944 Unconf'd	1985 Unconf'd	1970 Conf'd	84/85 Conf'd
"200 West"	410	470	430	442*

"200 East"	383	405	407	405

* Average hydraulic head of Rattlesnake Ridge aquifer, around "200 West" Area [22].

These data suggest that the relation between the aquifers may have been less affected under the "200 East" area.

A 30-foot hydraulic head difference under the "200 West" Area, as in Table 5, would suffice to carry iodine-laden water downward under "200 West" Area if a large enough and conductive enough structural defect was present there. Thus, the driving forces point to a continuous groundwater pathway connecting the "200" Areas, downward, to confined aquifer(s), then eastward, and finally upward to the river bed.

Broad structural defects must be present both under the source area and under the river for this groundwater pathway actually to exist. Unfortunately, details of Hanford stratigraphy are poorly understood. The possibility of a *window* connecting the aquifers near the "200" Areas has been mentioned in newly available documents [23].

The magnitude of connections which are known between the aquifers near the "200" Areas is indicated by Section **A-A'** (located in Fig. 12) in Fig. 13, below.

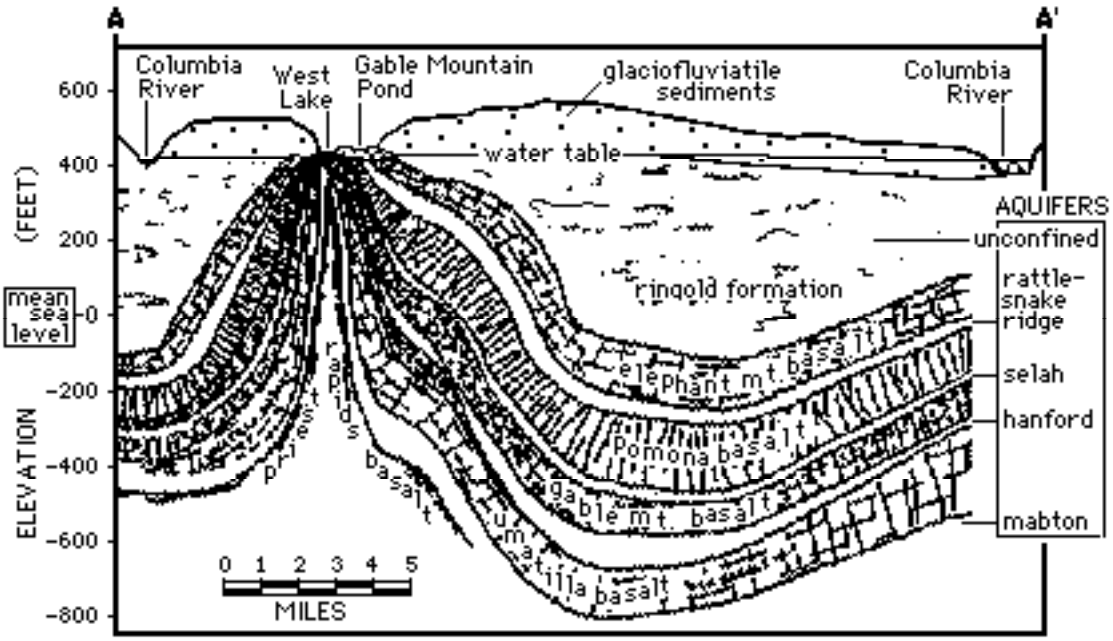


Fig. 13. Section A-A' across Gable Mountain Pond [24]

(The iodine-129 concentration near Gable Mountain Pond, as shown by Well DH-8 in Table 4, appears to be too low to be the unreported source of iodine to the river.)

Well 35-70 which is open to the unconfined aquifer and Well 31-31P which is open to confined aquifers (see Fig. 12 and Table 4) both show exceptionally

high iodine concentrations and high iodine-to-tritium ratios. Both wells are southeast of the "200 West" Area, in the direction of typical groundwater flow in both unconfined and confined aquifers. Thus, "200 West" Area is the likely source region of the unreported iodine-129 which enters the river.

OTHER RADIONUCLIDES

Radionuclides which are reportedly released from N-Reactor and which are sampled in the river are listed in Table 6 [25] on the next page. The tabulated radionuclides are mostly so dilute in river water that the downstream-upstream differences are close to the minimum detection levels. The exceptions are: • tritium and iodine-129 which have already been described, • technetium-99 which is not sampled in river water, and • cobalt-60 and strontium-89, -90 for which reported releases from N-Reactor are greater than observed, downstream sampler values.

The much greater reported release of strontium-90 than is detected downstream has been apparent for the last several years. The contractor's reported strontium-90 release from N-Springs is used by PNL [28] rather than the downstream sampler data to calculate a dose to the "maximally exposed

Table. 6. Accounting for riverborne radionuclides in 1986 (pCi/L)

Radionuclide		Conc. added from HRM 28	2xReported N-Reactor	River Sampler Difference	Approximate Detect.Limit	Comments[a]
Tritium	-3	43.	2.3[b]	50.	10.	accounted
Cobalt	-60	.002	.011	.0028	.002	(1)
Strontium	-89	<0	.038	.00	.02	(1)
	-90	.0004[c]	.17	.01	.02	(1)
Niobium	-95	--	--	<0	.002	(2)
Zirconium	-95	--	.0016	.000	.002	(3)
Technetium	-99	.02[d]	--	--	.002	(4)
Ruthenium	-106	.01	.0025	<0	.01	(3)
Iodine	-129	.000040	--	.000091	.000001	last subsec.
	-131	--	.0027	<0	.003	(3)
Cesium	-134	--	--	<0	.002	(2)
	-137	.0008	.002	<0	.003	(3)
Cerium	-144	--	--	.000	.003	(2)
Uranium	-234	.0004	--	.02	.005	(5)
	-235	.00001	--	.000	.002	(2)
	-238	.0003	--	.02	.004	(5)
Plutonium	-238	.000002	.00000006	<0	.0001	(2)
	-239/240	.0000005	.000019	.000072	.00004	(5)

[a] Comments:
(1) Unidentified pathway leaves river downstream from N-Reactor ?
(2) Radionuclide uninteresting at present detection levels.

(3) Radionuclide may become interesting as detection levels improve.
(4) River sampling would detect this radionuclide released from HRM 28 springs.
(5) An unidentified pathway to the river may exist.

[b] As reported, rather than twice reported value [26].
[c] Value from PNL analysis of Well 44-4 used instead of U.S. Testing values.
[d] Corrected to 230,000 pCi/L tritium [27].

where:

"Conc. Added from HRM 28" is the concentration in nearshore wells $\bar{C}_{ground}$ multiplied by $\bar{Q}_{ground}/\bar{Q}_{river}$ (with $\bar{Q}_{ground}$=8 cfs and $\bar{Q}_{river}$ =107,000 cfs) and divided by $\alpha\kappa$=0.4 to yield an effective concentration at the downstream sampler. "2xReported N-Reactor" is twice the reported release. This doubling of reported releases compensates for a contractor reporting practice of assuming that much longer pathways exist for all radionuclides discharged at 1301N and 1325N cribs (except tritium) than are represented by the contractor's N-Springs sampler. Since no documentation is available to support this assumption, it is disallowed here. This factor of two does not affect the "Comments" below the table. "River Sampler Difference" is the mean 1986 downstream concentration minus the mean upstream concentration. "Approximate Detect. Limit" is an estimate of the minimum detectable concentration of a radionuclide with current techniques. This estimate is based on PNL sources but assumes greater statistical work, more samples, and less confidence than PNL accepts.

individual." This seemingly conservative assumption neglects (1) possible under-reporting by the contractor and (2) the possibility that the largely unaccounted strontium-90 might follow a pathway yielding a very different dose than the assumed river water pathway.

CONCLUSIONS

(1) Studies of lateral mixing and sample averaging allow annual accounting of tritium entering the Hanford Reach of the Columbia River. About 1600 Ci is discharged from groundwater at HRM 28, and about 200 Ci enters from N-Springs at HRM 9. These releases agree with the measured downstream-upstream addition of tritium to within 25 percent.

(2) Tritium accounting provides an independent estimate of 7.7 cfs for the average discharge of groundwater from the unconfined aquifer at HRM 28. This estimate compares with a previous estimate of 10 cfs and a lower bound measurement of 6.3 cfs. Average discharge is now estimated to be 8±2 cfs.

(3) Shoreline springs at HRM 28 probably contribute half or less of the iodine-129 measured downstream of Hanford. The remaining few millicuries per year probably come from confined aquifer(s) below the river bed.

(4) The pathway of half or more of iodine-129 entering the river most likely begins in the "200 West" Area unconfined aquifer, continues downward into confined aquifer(s) below, extends easterly in confined aquifer(s) to strata below the Columbia River, and then upward to the river bed. The initial source of iodine-129 entering this pathway is probably Hanford Operations.

(5) Travel times along the *fastest* confined aquifer pathway between "200 West" Area and the river are likely less than 30 years.

(6) The probable existence of this high speed pathway having vertical legs near both "200 West" Area and under the river precludes Hanford as a potential high-level waste repository site because of the geohydrology disqualifying condition which requires a pre-emplacement groundwater travel time of at least a thousand years along a similar pathway.

(7) Significant quantities of radionuclides which are sampled at the "300" Area at HRM 42.5 may enter the river farther downstream, but still upstream of the Richland pumping stations. Downstream samplers should be located at one site which is demonstrably downstream of all Hanford impacts.

(8) Technetium-99 originating at HRM 28 is probably detectable in downstream samples of river water. Routine monitoring of technetium-99 would provide a valuable tool for accounting Hanford waste pathways.

(9) Releases of cobalt-60 and strontium-89 and -90 from N-Springs at HRM 9 are not accountable at the downstream, river sampler stations. The fate and impacts of these missing radionuclides are unknown.

(10) Some of the methods used by USDOE and Hanford contractors to estimate the groundwater impacts of Hanford Operations are biased to underestimate those impacts. Because of the methods used, the impacts are most underestimated where relatively few data exist.

RECOMMENDATIONS

(1) Technetium-99 should be sampled routinely, in the river upstream and downstream of Hanford and in some wells open to the unconfined aquifer and in other wells open to individual confined aquifers. Technetium samples should be collected at wells which have other special (other than tritium and nitrate) analyses in order to characterize the water present. Wells should be selected for suitability as a data base.

(2) Pathways of radionuclides in waste streams from Hanford should be identified, and their fluxes should be accounted and reported. The most important gaps are: (1)The source area for iodine-129 at "200 West" Area should be confirmed experimentally. (2) The strontium-90 pathway between N-Springs and the downstream sampler needs to be defined. (3) Uranium pathways need to be defined.

(3) There should be a single downstream sampler location for all downstream data which are comparable to the Priest Rapids Dam.

(4) Hanford should be eliminated as a candidate for a high-level waste repository on the basis of the geohydrology disqualifying condition.

(5) Worst case groundwater pathways and fastest groundwater travel times should be identified so that adequacy of long-term disposal of defense wastes already at Hanford can be assured.

(6) The major pathway(s) of cobalt-60 and strontium-89 and 90 from N-Springs should be identified before substantially more of these wastes are released. That is, the pathway(s) should be identified before N-Reactor is restarted.

(7) Methods of evaluating impacts of Hanford Operations should be revised to eliminate major biases which underestimate those impacts.

Appendix A.
LATERAL MIXING DOWNSTREAM OF HRM 28

Tritium-contaminated groundwater enters the west side of the Columbia River at HRM 28. Several processes mix the tritium away from the shore and across the river. Candidate process include turbulent diffusion resulting from eddies in the river flow, a spiral flow caused by centrifugal force on water flowing rapidly around bends in the river, more or less permanent meanders of the main stream between the banks, and mixing induced by the islands in the flow.

Although the theory of turbulent diffusion in rivers is incomplete and requires some *ad hoc* assumptions, enough experiments have been performed in several rivers to allow good estimation of lateral mixing rates in rivers having slow meanders and moderate sidewall irregularities. That is, if the river is simple enough, then the spiral flows and other large-scale cross-river mixing processes can be safely ignored. Unfortunately, the route of the Benton-Franklin County line, which appears to approximate the main channel of the Columbia River reasonably well, suggests that bends and islands may have an important effect on mixing in the lower part of the Hanford Reach; see Fig. A1 which is the next page.

Fischer [29] suggests a condition to test whether river bends and sidewall irregularities are moderate enough to allow the theory of turbulent diffusion to be applied. That condition may be written as

$$Q^{1/10} W^{9/10} / S^{1/20} R \quad \text{usually} < 0.32$$

where

- Q = river flow (cfs)
- W = river width (feet)
- S = slope of river surface in upstream direction (feet/feet)
- R = effective radius of river meanders through bends (feet)

For an average Columbia River flow of Q=112,000 cfs and mapped slope of $S=2.46X10^{-4}$ between HRM 21.0 and HRM 36.4, this condition simplifies to

$$W^{9/10} / R \quad \text{usually} < 0.65$$

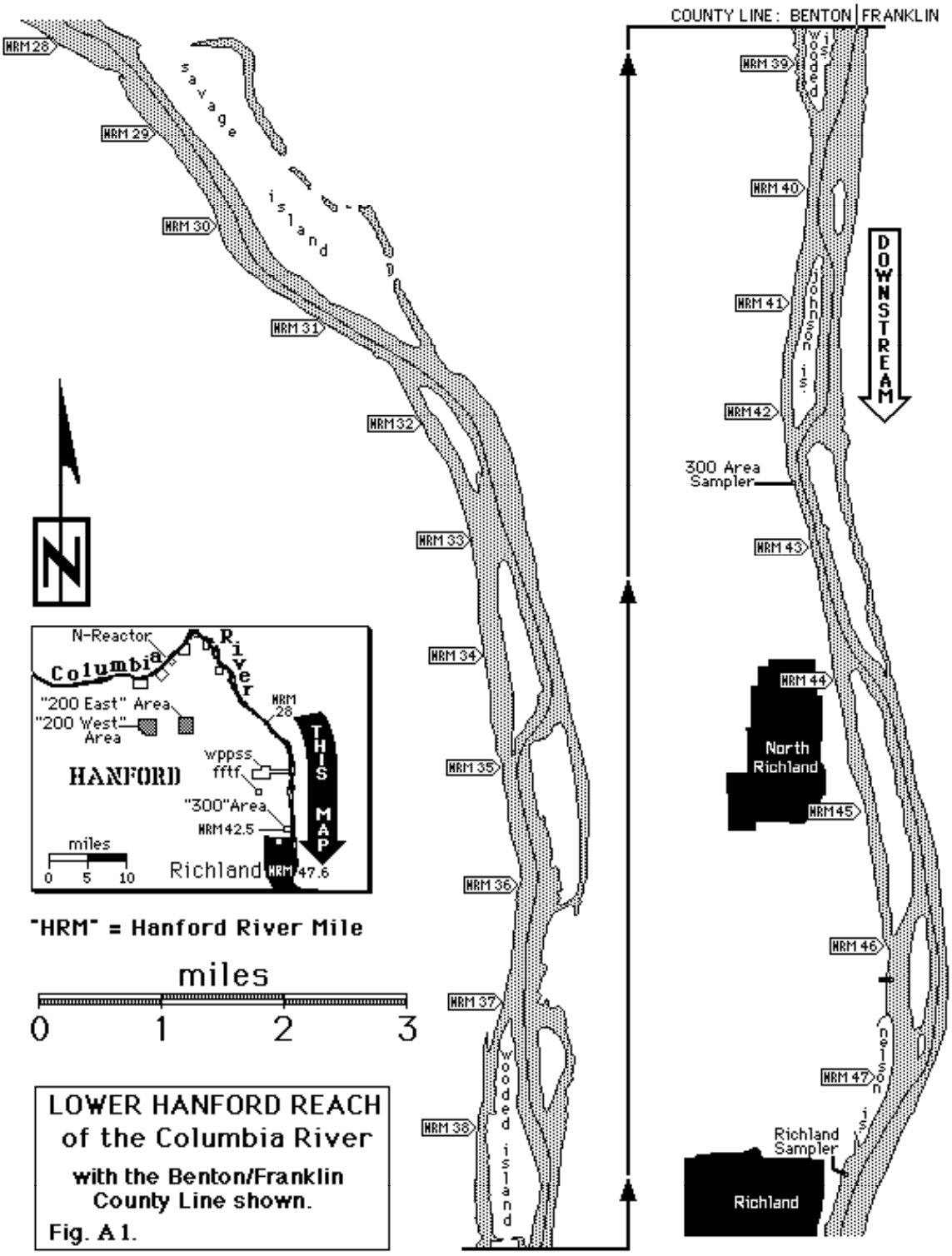
For a typical Hanford Reach width of W=1500 feet, bends in the river channel must have radii of about

$$R \sim 11{,}000 \text{ feet}$$

or greater. This constraint is not satisfied in the lower part of the reach where the river channel bends abruptly, switching sides of the river between adjacent

islands, with radii of the Benton-Franklin County Line often as small as 3,000 feet. Thus, the simple theory of turbulent diffusion is probably inadequate to describe lateral mixing across the river between HRM 28 and Richland.

Fischer also devised a theory to describe spiral flows in rivers which have relatively sharp bends [30]. Unfortunately, the bends between the islands in the



LOWER HANFORD REACH of the Columbia River with the Benton/Franklin County Line shown.

Fig. A1.

Columbia River are too sharp for even this theory to apply. For such a situation, Fisher suggests that lateral mixing should be measured experimentally.

Field measurements of lateral mixing may be performed by several methods: (1) The most direct method is the measurement of the tritium distribution across the river at the downstream sampler at HRM 47.6. This method has been proposed by both PNL and the Geological Survey and is expected to provide accurate mixing rate data. (2) Dye may be released at HRM 28 and its concentration may be measured downstream. This method may be considered to reduce the quality of Columbia River water. (3) Water in the Columbia River at HRM 28 may be tagged with *drogues* which then follow the tagged water down river toward Richland. This last method was selected for a lateral mixing study. A drogue is sketched in Fig. A2, below.

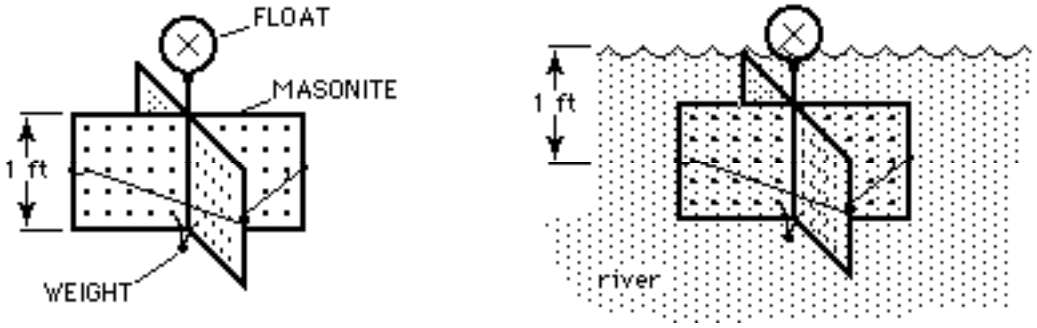


Fig. A2. Drogue

Drogues are inexpensive, collapsible, and highly visible. They are little affected by wind, moving only about $1.4X10^{-3}$ times wind speed, 50° to the left of wind direction [31]. (By limiting study times to early morning hours, this wind speed correction was kept negligible.) The drogues were also numbered so that individual elements of water could be tracked separately. This allowed rates of mixing to be calculated separately for different distances offshore.

In order to use drogues to measure lateral mixing, their offshore distances must be measured. Two angles (A and B) were measured simultaneously (to 0.2 seconds) using matched *C.Plath* sextants at the ends of a 100-foot baseline, Fig. A3, below.

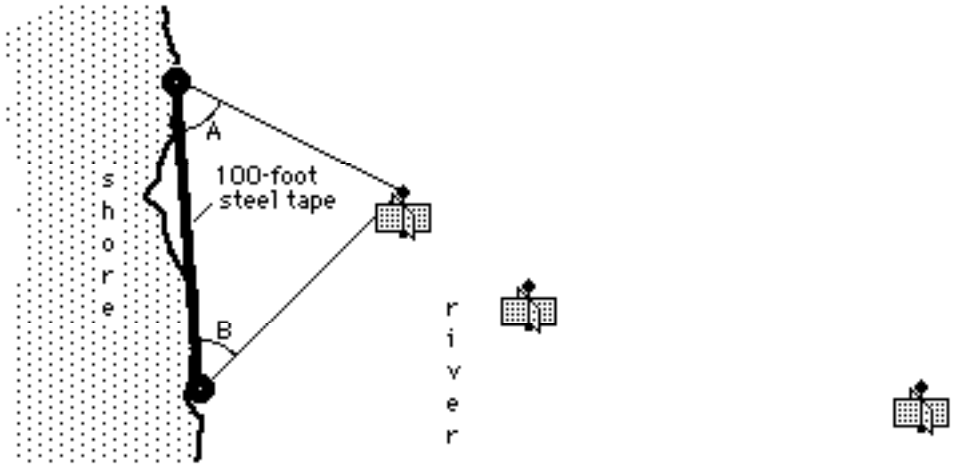


Fig. A3. Ranging technique

Each end of the 100-foot baseline was flagged, and each sextant operator superpositioned the image of the flag at the opposite end of the baseline with the drogue being measured, through a 2.5X telescope. This technique has a nominal accuracy of one percent for distances to 1000 feet. When nearshore drogues moved rapidly past the operators, accuracy was reduced by the spin rate of the verniers on the sextants. Measurements were also hampered by reflection of the sun off the water at sunrise. Overall, this measurement technique was satisfactory.

The lateral dispersion of a set of drogues was used to approximate the lateral mixing of tritium or other contaminant discharged into the river at HRM 28, for most of the candidate processes which are likely to be important in the Columbia River. An exception is the spiral mixing which occurs at sharp river bends. In particular, as the river channel winds its way between islands, the drogues are carried to the near shore. This effect is sketched in Fig. A4, below.

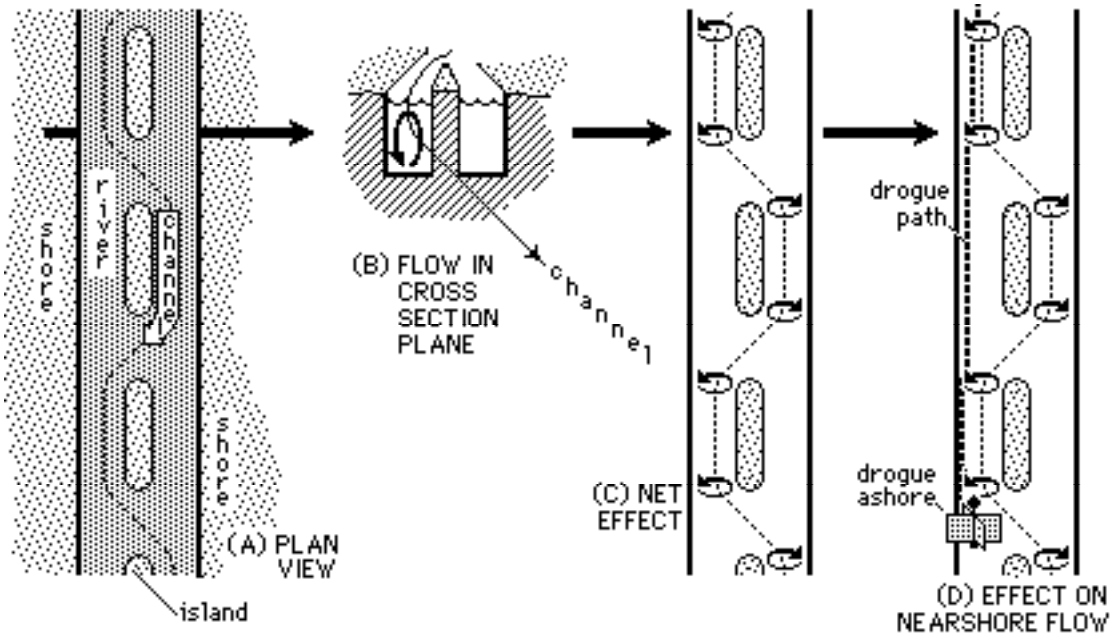


Fig. A4. Spiral flow pattern in idealized, winding river

Fig. A4(A) is an idealized sketch of a river channel winding between islands, as occurs in the Columbia River between HRM 31.5 and HRM 47. In this idealization, the river is straight, and the depth is constant. Fig. A4(B) shows an idealized cross-sectional flow pattern where the river channel turns at the downstream end of the first island in Fig. A4(A). Water near the river surface moves downstream fastest. Thus, it is thrown outward most (centrifugal force) in the bend. This outward flow near the surface is balanced by an inward return flow near the river bed. Fig. A4(C) shows the reversing spiral current pattern around the main flow as it zigzags down the river between the islands. Fig. 11(D) shows the path of a drogue which is so near to shore that it does not follow the main river channel. Notice that ***each*** time the channel zigs or zags, the drogue is displaced closer to shore by the spiral current. In practice, only two or three zigzags are required to put most drogues ashore.

Although this spiral current carries drogues ashore, contaminants are mixed across the river by the spiral. This occurs because contaminants are well-mixed vertically through the water column; whereas, the drogues follow the surface water only. By comparison of Figs. A4(B) and (D), one sees that contaminants in the bottom water are carried offshore by the repeated application of the spirals. Consequently, surface drogues generally underestimate the rate of lateral mixing away from the west bank of the Columbia River downstream of HRM 31.5 where the islands were encountered.

Within two stretches of the study reach, however, the spiral current pattern reverses so that drogues following surface water reflect the actual mixing process. One such stretch is between HRM 28 and HRM 31, upstream of the islands, Fig. A1. In that stretch, the river bends to the right and then to the left an almost equal amount. The second stretch is between HRM 31 and HRM 36.8. In this stretch, the entire river bends to the right at HRM 31.5 so that the main channel crosses the river toward the east while the water flow bends toward the west, reversing the spiral flow pattern of Fig. A4(B). The usual spiral pattern then appears at the channel bend at HRM 35.

Since Columbia River flows can be held to constant values fairly easily only for durations of several hours, the drogue study begun at HRM 28 had to be interrupted and then resumed later. Parts of the study were *jointed* at HRMs 31, 33, and 36.8, and then extrapolated to the downstream sampler locations at HRMs 42.5 and 47.6. Figure A5(A), below, shows the reference coordinate system which is used to describe the study, with "x" the downstream coordinate, "y" the lateral, cross-river coordinate directed roughly to the east, and "z" the vertical coordinate.

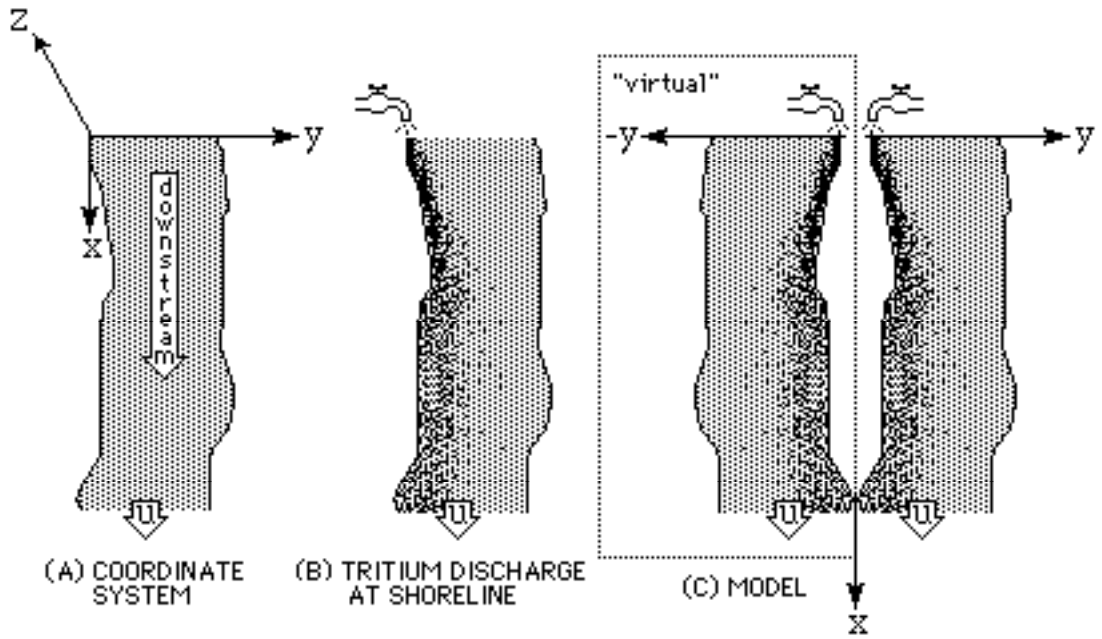


Fig. A5. Coordinate system and "virtual" model

The origin is at the west shore at HRM 28. River flow rate Q and location (x) determine the average river flow speed u (=Q/river-cross-sectional area.)

Figure A5(B) diagrams a contaminant discharge at the west bank, much like the discharge of tritium-bearing groundwater at HRM 28. Figure A5(C) shows the mirror image of that contaminant discharge sketched back to back next to the true discharge. The combination of this "virtual" discharge with a true discharge provides a conceptual model which will be used in this study. This means that every drogue placed in the river has an imaginary twin in the model.

This modeling technique was developed last century to solve boundary problems in electrostatics. It has also been applied to contaminant discharges near a river bank in models of turbulent diffusive mixing. This technique introduces the river bank as a *boundary* into the model by the artifice of removing any lateral concentration gradient at the shoreline. This assures that modeled contaminant remains in the river, just as real contaminant does. (However, this mechanism in the model differs from the mechanism in reality, which is zero diffusivity at the boundary.)

The introduction of an imaginary, mirror flow into the conceptual model does much more than satisfy a "no-flow" boundary condition at the west bank of the river. It also produces a concentration profile in the y-direction across the river, which looks a lot like a bell (Gaussian) curve constrained to having a mean value of y=0.

The theory of turbulent diffusive mixing of a contaminant treats eddies in a river like a random walk problem. After any particle of contaminant enters an eddy, it is considered to be carried a distance roughly equal to the eddy size

and then ejected in a random direction. With such a random walk process, the contaminant would spread across the river as the contaminant was carried downstream. At any distance (x) downstream, the contaminant would have a Gaussian concentration distribution in the y-direction. The center (mean value) of this distribution would remain at y=0, while the width (standard deviation σ of the concentration distribution) would increase as the square root (1/2 power) of the distance downstream:

$$\sigma \sim X^{1/2} \qquad \text{(A1)}$$

For smoothly bending rivers with fairly regular shorelines, this theory allows a good estimate of the constant of proportionality which describes how rapidly σ increases in the x-direction. Although the course of main river flow down the lower Hanford Reach is too irregular for a textbook value of the constant of proportionality to be employed, the various poorly known mixing processes which occur in this reach may still be similar enough to random walk processes that Eq. (A1) applies.

Even if this assumption is not strictly valid, the existence of a variety of presumably important mixing processes suggests that the Central Limit Theorem does apply. For the problem of contaminant mixing across the Columbia River, this theorem might be stated as,

> Provided there are many important, lateral mixing processes which are reasonably well behaved, and provided only locations where the cross-sectional form of the river is simple and similar are considered, then modeled concentration distribution from a shoreline discharge is essentially Gaussian.

In terms of a drogue study, this means that the locations where σ-values are calculated for a drogue array and then extrapolated downstream according to Eq. (A1) must be selected so that the cross-sections of the river must be similar at the selected locations. The present study did not include bathymetric profiling. Survey locations at the reported HRMs 31, 33, and 36.8 were selected visually for "reasonable" bathymetry and stream flow behavior and minimal shoreline irregularities.

Now suppose that n-many drogues are thrown into the river at HRM 28, say, with the i^{th} drogue distance y_i offshore. Then the square of the measured standard deviation σ_{meas} of the drogue distribution is defined as

$$\sigma_{meas}^2 = \frac{\sum_i^n y_i^2}{n - 1/2} \qquad \text{(A2)}$$

The denominator of Eq. (A2) is "n-1/2" rather than the usual "n-1" because each modeled drogue has a virtual twin. The conceptual model has twice as many drogues (half of them imaginary) as are put into the river.

Only half dozen drogues could be tracked at one time because of the rapidity of river flow. These drogues could only be allowed to travel three or four miles before they became so dispersed that some were missed in the morning glare. Even then, drogues were swept ashore and others were lost either temporarily or permanently. When drogues were lost before they passed a survey location, they provided no data for the calculation of σ in Eq. (A2). Therefore, it was impossible to predict exactly what distances offshore drogues should be emplaced to achieve a particular starting array width σ_1 which was sought.

Suppose that some drogues were emplaced at HRM x_1 and that some of them later passed downstream location HRM x_2. All of the drogues which were not sighted at both locations are excluded from the calculations. Thus, neither the measured value of σ_1 nor of σ_2 was the value sought (i.e., the actual, estimated width of contaminant plume which originated at HRM 28). The values are corrected vis Eq. (A1), as follows:

$$(\sigma_2/\sigma_1)^2_{\text{sought}} = 1+ \left[(\sigma_2^2-\sigma_1^2)_{\text{meas}}/\sigma_1^2{}_{\text{sought}}\right] \qquad \text{(A3)}$$

This equation is applicable provided σ_{sought} is not so different from σ_{meas} that the drogues track an irrelevant mixing regime. This constraint is probably not a problem in the present study because measured drogue array widths were close to sought widths. Thus, the drogue study progressed downstream under a fixed river flow (112,000 cfs).

According to this conceptual model, the width σ of the drogue distribution is described by Eq. (A1) which may be rewritten as

$$\sigma_2^2 = \sigma_1^2 + \gamma_x X_{2\text{-}1} \quad . \qquad \text{(A4)}$$

where $X_{2\text{-}1}$ is the distance downstream between Measurement 1 and Measurement 2. Then an undetermined, mixing rate factor γ_x is solved from the data. With γ_x solved, Eq. (A4) permits σ to be extrapolated farther downstream provided the mixing mechanisms remain invariant. A dimensional sensitivity analysis suggests that the extrapolation of Eq. (A4) downstream to Richland may result in a five percent underestimate of σ at Richland.

This drogue study framework seems adequate for all the obvious mixing processes except spiral flows among the islands downstream of HRM 31.5. If the islands dominate mixing in this stretch of the Hanford Reach, then σ would similarly relate to "n" zigs and zags of the river channel as

$$\sigma_2^2 = \sigma_1^2 + \gamma_n n_{2\text{-}1} \quad . \qquad \text{(A5)}$$

Mixing rates have been calculated for both Eqs. (A4) and (A5) to provide a range of possible lateral mixing rates.

The width (σ) of the tritium release into the river at HRM 28 was calculated by multiplying the corresponding nitrate elevation above background by the water depth at the sample locations in SEARCH's 1986 study [32]. The standard deviation σ of this vertically integrated concentration distribution was calculated to be 66 feet.

With Columbia River flow rate at the same mean annual value (Q=112,000 cfs) at which the tritium discharge into the river was measured, six drogues were placed in the river at 0613 hours on 24 July 1987. They were placed at y=9, 26, 43, 58, 78, and 109 feet offshore. These placements were selected to approximate a Gaussian distribution having the required σ=66 feet. That is, Eq. (A2) gives $s=(9^2+26^2+43^2+78^2+109^2) / 5.5 = 66$ feet. This placement at HRM is shown in Table A1, below.

Table A1. DROGUE STUDY OF 24 JULY 1987

HRM : Drogue #1	#2	#3	#4	#5	#6
28 9 (0613)	26 (0613)	43 (0613)	58 (0613)	78 (0613)	109 (0613)
29 151 (0714)	[HRM 28.2]	[HRM 28.3]	150 (0709)	302 (0644)	329 (0642)
31 122 (0934)			[HRM 30.5]	200 (0804)	129 (0805)
32 77 (1035)				126 (0916)	~1000 (0900)^

format: feet offshore (time) [location aground]

^ Position estimated as passing 300 feet offshore of island.

Two drogues (#2 and #3) were apparently grounded almost immediately by a spiral flow associated with a left (eastward) bend of the current as it approached HRM 28.5, see Fig. A1, before the river bent sharply to the right (westward). This bending of the current in the river had previously been observed, and it was expected to introduce substantial lateral mixing which would appear as increases in the offshore positions of the drogues.

Excluding the two grounded drogues from the calculations at both HRM 28 and 29, the widths of the drogue array were σ_{28}=78.2 feet and σ_{29}=264 feet, where the subscripts refer to HRMs. By HRM 31, the river had resumed its initial direction. Excluding the three grounded drogues from all three calculations, the widths of the array were σ_{28}=85 feet, σ_{29}=298 feet, and σ_{31}=169 feet. This reduction in σ between HRMs 29 and 31 is attributed to a combination of spiral flow and an embayment below HRM 28.5.

These drogues were tracked downstream to HRM 32. The drogues which passed west of the island were seen to hold closely to the shore; whereas, the

drogue which followed the main current to the east of the island was essentially swept away. The effect of the upstream end of the island, then, was to reduce the lateral mixing of the part of the flow held close to shore and to increase the lateral mixing of the part of the flow which was carried far from shore. This effect was observed again, later in the study.

With the flow separated into a high mixing rate part and a low mixing rate part by the island, it became obvious that the concentration distribution could not be even approximately Gaussian on a section across an island. Thereafter, tracking locations were restricted to places in the river where islands were absent.

The study was resumed on 5 August with river flow again stabilized at 112,000 cfs. Six drogues were placed in the river at HRM 30.8 at 0555 hours. Their offshore distances were estimated visually to be roughly y=14, 35, 46, 60, 80, and 112 feet. Then their distances offshore were measured as they passed HRM 31, with the results shown in the first line of Table. A2.

Table A2. FIRST DROGUE STUDY OF 5 AUGUST 1987

HRM :	Drogue #1	#2	#3	#4	#5	#6
31	94 (0619)	118 (0613)	163 (0613)	200 (0613)	254 (0610)	303 (0609)
33	369 (0740)	[HRM36.5]	55 (0739)	654 (0658)	[HRM 31.6]	161 (0710)

HRM :	Drogue #1a	#2a	#3a	#4a	#5a	#6a
31	20 (0850)	40 (0850)	60 (0850)	80 (0850)	100 (0850)	125(0850)
heading:	upstream	upstream	upstream	upstream	upstream	downstream

format: feet offshore (time) [location aground]

As the drogues passed HRM 31, a stable eddy was seen to have developed near the shore. After only 1000 feet of travel downstream to the nominal starting location at HRM 31, the drogues had moved an average of about 130 feet offshore. Drogue #1 had moved the least distance offshore -- 80 feet. An hour and a half later, the six drogues were replaced at HRM 31 to determine the width of the eddy (see the lower part of Table 2). At that time, the edge of the eddy was observed to be 100-125 feet wide (between Drogues #5a and #6a) as indicated by the headings of the drogue movements.

Since each drogue which had been placed in the river at 0555 hours progressed directly downstream, the eddy must have been less than 94 feet wide at that time. Thus, the distances the drogues were offshore at HRM 31 in Table 2 can be treated in two ways: One extreme possibility is to treat the eddy which must have extended almost to Drogue #1 as though the shoreline was actually about 90 feet from the baseline which was set at the actual shoreline. The other extreme possibility is to ignore the existence of the eddy. In the analysis, below, both of these extreme possibilities are followed to bracket the lateral mixing rate in the river.

To examine further the lateral mixing in this important stretch, where the main current crossed from the west side to the east side of the river while the current *bent toward the west*, the drogues were replaced downstream from the eddy at HRM 31, with the results shown in Table A3, on the next page.

Table A3. SECOND STUDY OF 5 AUGUST

HRM : Drogue #	1b	#2b	#3b	#4b	#5b	#6b
31.3	20 (0915)	40 (0916)	60 (0917)	80 (0918)	100 (0919)	120 (0920)
32	[missed]	[HRM31.6]	90 (0954)	165 (0944)	60 (0954)	173 (0945)
33	43 (1059)*		61 (1047)*	214 (1011)	[HRM 32.3]	206 (1012)
36.8	[HRM 33.1]		[HRM 35.4]	311 (1130)		223 (1158)
Correcting for wind:						
36.8				323 (1130)		245 (1158)

* stuck on boulder bar and released; arrival time corrected for time on bar.
format: feet offshore (time) [location aground]

The data for HRMs 31.3 and 33 in Table A3 may be compared to the data for HRMs 31 and 33 in Table A2. The sought width of the drogue array when emplaced at HRM 31 was

$$\sigma_{31,sought}=164 \text{ feet.}$$

The measured starting width of the array from Table A2 which arrived at HRM 33 was

$$134 \text{ feet} < \sigma_{31,meas} < 219 \text{ feet} .$$

The lower end of this range results from subtracting a 90-foot wide eddy from the drogue distances offshore. The upper end ignores the eddy. Thus, the drogues were set either a little too close to shore or a little too far from shore, depending on the interpretation.

At HRM 31.3,

$$\sigma_{31.3,sought}=171 \text{ feet.}$$

The drogues were emplaced with

$$\sigma_{31.3,meas}=84 \text{ feet.}$$

That is, the drogue array was placed substantially too close to shore at HRM 31.3.

The conspicuous consequence of this closer-than-proper emplacement of the drogues at HRM 31.3 was that none of them passed to the east of the island at HRM 31.5; whereas, Drogue #6 from HRM 28 (Table A1) and Drogue #4 from HRM 31 passed to the east of the island.

Both $\sigma_{31.3,meas}$ and the mixing rate factor γ [Eq. (A4)] are much affected by the part of the array passing east of the island: For example, if Drogue #4 were eliminated from Table 2, $\sigma_{33,meas}$ would be reduced from 412 feet to 257 feet. As another example, compare γ_x of the drogue sets of Tables 2 and 3:

Drogues set at HRM 31: Drogue #4 east of Island: 11.5 feet < γ_x <14.4 feet

Drogues set at HRM 31.3: No drogues east of island: γ_x = 2.2 feet

Notice that when the array width was reduced by a factor of two or three between the conditions of Tables A2 and A3, the mixing rate was reduced by a factor of 6.5. Clearly, different lateral mixing processes dominated the two drogue data sets. The first drogue array was divided into two parts, one part which was almost immediately mixed into the main stream and the other part which was constrained to a secondary channel on the west side of the island. Also notice that Drogues #1b and #3b were grounded in the spiral flow south of the island at HRM 34. This spiral flow decreased drogue distances offshore between HRMs 33 and 36.8.

The drogues were reset again under a steady river flow of 112,000 cfs on 12 August, with results shown in Table A4, below:

Table A4. FIRST DROGUE STUDY OF 12 AUGUST 1987

HRM : Drogue #1	#2	#3	#4	#5	#6
33	135 (0609)	147 (0610)	233 (0611)	345 (0614)	488 (0616)	734 (0618)
36.8	[HRM 35.8]	[HRM 35.2]	[missed]	407 (0715)	398 (0716)	[missed]
39.8			[HRM 39.3]	190 (0912)	175 (0855)	[missed]

format: feet offshore (time) [location aground]

Drogues #4 and #6 passed on the east side of Wooded Island. (In the following section, the data for Drogues #4 and #5 were combined with the wind corrected data for Drogues #4b and #6b in Table A3.)

The drogues were reset at HRM 39.8 and measured on the south side of Johnson Island at HRM 42.2, as shown in Table A5, below.

Table A5. SECOND STUDY OF 12 AUGUST

HRM : Drogue #1	#2	#3	#4	#5
39.8	147 (1150)	233 (1150)	345 (1150)	488 (1150)	734 (1150)
42.2	[HRM 40.8]	52 (1331)	165 (1244)	358 (1251)	336 (1236)

format: feet offshore (time) [location aground]

Only Drogue #5 passed east of Johnson Island. The spiral flow at each end of Johnson Island (Compare Figs. A1 and A4.) carried the surface water tagged by these drogues toward shore.

Drogue array widths were calculated with Eq. (A2) for drogue positions in Tables A1 through A5. Positions at HRMs 33 and 36.8 were combined from Tables A4 and A5, as shown in Table A6, below.

Table A6. MEASURED ARRAY WIDTHS (feet)

	HRM:	28	31	33	36.8
drogues set:	σ_{meas}	78.3	118.9*	357.0	
			200.6		
drogues observed:	σ_{meas}		169.1	426.2	373

* 90-foot wide nearshore eddy subtracted from drogue distances offshore to obtain this value.

The width σ_x of an array of drogues some distance x downstream of their emplacement location depends on lateral mixing in the river, on spiral flows, and on river width. Determination of the lateral mixing dependence is the objective of the study. Most of the spiral flows introduce interference which can be nearly removed from the study by selection of particular stretches of the river for measurements (HRM 28 to HRM 31 and HRM 31 to HRM 36.8). In order to extrapolate study results downstream to the samplers at HRMs 42.5 and 47.6, the effect of river width may be extracted from the data, as follows:

Consider an illustrative example in which lateral mixing and shoreline irregularities are ignored. Then if the river geometry did not change, the offshore distances of drogues would be unaltered as they traveled downstream from Location 1 to Location 2 in Fig. A6(A), below.

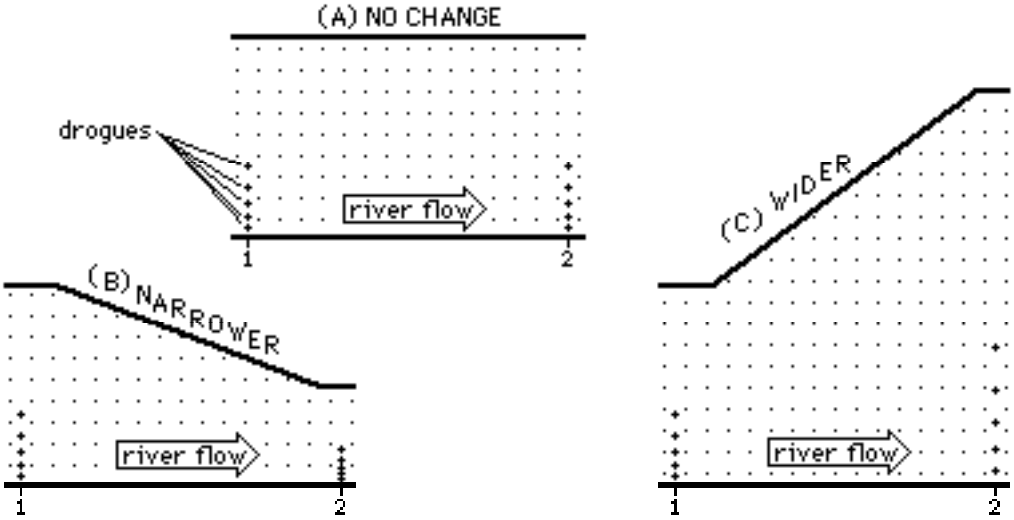


Fig. A6. Effect of river width

Likewise, if the river width slowly decreased at Location 2 to half the width at Location 1, as in Fig. A6(B), above, then one would expect the width of the

drogue array would become half as great. And similarly, the drogue array would double in width if the river did, as in Fig. A6(C).

Since the width of the Lower Hanford Reach is typically close to 2000 feet, this effect of river width on the scaling of the sought array width may be conveniently compensated by multiplying the drogue array width σ_x at distance x downstream by the factor

$$\sigma_{x \Rightarrow 2000 \text{ ft width}} = \sigma_x(2000 \text{ feet})/(\text{river width at x}) \qquad \text{(A6)}$$

Observed river widths at measurement locations are listed in Table A7, below.

Table A7. RIVER WIDTHS (feet)

HRM:	28	31	33	36.8	39.8	42.2
RIVER WIDTH (ft):	1530*	992	2253	1481	2148	2791

* Width at HRM 28 was read from map.

River width has other, secondary effects which are not removed by simple scaling. Further, as may be seen in Fig. 13, changes in river width necessarily introduce shoreline irregularities which increase lateral mixing. The described drogue study likely extended far enough downstream and included adequately representative river widths to have incorporated these secondary effects.

The measured array widths are scaled to a 2000-foot wide river by means of Eq. (A6), as follows:

Table A8. MEASURED ARRAY WIDTHS (feet) CORRECTED TO A CONSTANT RIVER WIDTH OF 2000 FEET

	HRM:	28	31	33	36.8
drogues set:	$\sigma_{meas \Rightarrow 2000 \text{ ft width}}$	102.4	239.7 404.4	316.9	
drogues observed:	$\sigma_{meas \Rightarrow 2000 \text{ ft width}}$		340.9	378.3503.7	

Likewise, the width of the tritium discharge at HRM 28 is corrected to 86.3 feet (=66.0 feetx2000 feet/1530 feet). Sought widths $\sigma_{2000 \text{ ft width} \Rightarrow \text{sought}}$ of the arrays downstream of HRM 28 are then calculated from Eq. (A3) with results listed in Table A9:

Table A9. SOUGHT ARRAY WIDTHS (feet) FOR AN IDEALIZED, 2000 FOOT WIDE COLUMBIA RIVER

HRM:	28	31	33	36.8
$\sigma_{2000\ ft\ width \Rightarrow sought}$	86.3	330.2	391.3	613.0
			364.1	572.9

The mixing rate factor $\gamma_{x,28\text{-}31}$ between HRMs 28 and 31 is calculated from these results and Eq. (A4):

$$\gamma_{x,28\text{-}31} = 6.41 \text{ (feet)}$$

Downstream of HRM 31, islands are present. Thus, lateral mixing can also be described in terms of river channel zig zags by Eq. (A5). Employing the range of $\sigma_{2000\ ft\ width \Rightarrow sought}$ values in Table A9 for the stretch between HRMs 31 and 36.8, the ranges are:

$$7.16 < \gamma_{x,31\text{-}36.8} < 8.71 \text{ feet}$$
$$109{,}591 < \gamma_{n,31\text{-}36.8} < 33{,}368 \text{ square feet}$$

These mixing rates γ correspond to lateral diffusion factors (D_y), which have been measured by means of radionuclide and dye release studies near the upstream production reactors [33]. For the $\sigma_{2000\ ft\ width \Rightarrow sought}$ data, D_y is

$$D_y = (\sigma_2^2 - \sigma_1^2) / 2t_{2\text{-}1} \qquad (A7)$$

where $t_{2\text{-}1}$ is the average time for the drogues to travel from Location 1 to Location 2. With the elapsed times provided with the drogue position data, calculated values of D_y are

$$D_{y,28\text{-}31} = 6 \text{ ft}^2/\text{sec}$$

$$D_{y,31\text{-}36.8} = 15 \text{ ft}^2/\text{sec}$$

These diffusion coefficients compare to values near 2 ft^2/sec measured by Hanford contractors near the production reactors in the upper reach. Considering the increased opportunity for lateral mixing associated with winding of the river channel in the lower reach, these values of lateral diffusion factors are reasonable [34].

If the values of γ (displayed above) are extrapolated downstream to HRM 47.6, the width σ of tritium contaminant in the river is estimated to be about 9% larger based on river channel zig zags (γ_n) instead of distance (γ_x). Therefore, a maximum range of estimates is obtained by setting γ_x=7.16 feet and γ_n=33,368 square feet. Contaminant widths $\sigma_{2000\ ft\ width \Rightarrow 42.5}$ and $\sigma_{2000\ ft\ width \Rightarrow 47.6}$ at the downstream sampler sites at HRMs 42.5 and 47.6 are then extrapolated from Eqs. (A4) and (A5) to lie within the following ranges:

$$737 < \sigma_{2000\ \text{ft width} \Rightarrow 42.5} < 954 \quad \text{feet}$$

$$858 < \sigma_{2000\ \text{ft width} \Rightarrow 47.6} < 1021 \quad \text{feet}$$

Now assume (1) that the time-averaged, added tritium contaminant concentration $\bar{C}(y)-\bar{C}_{up}$ in the lateral direction (y), far downstream of HRM 28 has a Gaussian distribution,

$$[\bar{C}(y)-\bar{C}_{up}] \ / \ (\bar{C}_o-\bar{C}_{up}) \ = \ \exp(-y^2/2\sigma^2) \ , \qquad \text{(A8)}$$

and (2) that river depth is approximately constant, then the mean concentration $\bar{C}_{river}$ of contaminant laterally across the river can be related to the concentration $\bar{C}_o$ at the samplers near the west bank of the river. By averaging Eq. (A8) across a river width of 2000 feet, a concentration factor κ is obtained [35]:

$$\kappa = (\bar{C}_{river}-\bar{C}_{up}) \ / \ (\bar{C}_o-\bar{C}_{up}) \qquad \text{(A9)}$$

The results are

$$0.46 < \kappa_{42.5} < 0.58$$

$$0.53 < \kappa_{47.6} < 0.61$$

The middle values of these ranges are employed, hereafter, as the best estimates of lateral mixing between the groundwater discharge at HRM 28 and the downstream samplers at HRMs 42.5 and 47.6, respectively. That is,

$$\kappa_{42.5;\ 47.6} = 0.52; \ 0.57 \qquad \text{(A10)}$$

Appendix B. AVERAGING

The Accounting Equation (1) relates the difference in the concentration of a contaminant such as tritium between the downstream sampler at HRM 47.6 and the upstream sampler at Priest Rapids Dam to the addition of contaminant entering the river with groundwater at HRM 28, plus contaminant N_N added from N-Springs, plus other sources N_{other} :

$$\alpha\kappa(\bar{c}_{down}-\bar{c}_{up})\bar{Q}_{river} = \bar{c}_{ground}\bar{Q}_{ground} + N_N + N_{other} \qquad (1)$$

This Accounting Equation simply requires that tritium or any other contaminant be conserved in the following sense: The contaminant which enters the Hanford Reach of the river must either leave the Hanford Reach, remain in the Hanford Reach, or be eliminated by breakdown (e.g. decay). The options of remaining in the Hanford Reach and elimination by breakdown are included in the final source/sink term N_{other}. In the case of tritium which is merely heavy hydrogen incorporated into heavy water in the river, the options of remaining or decaying in the reach are negligible. (Tritium has a halflife of 12.3 years, and river water travels through the the Hanford Reach in half a day.)

The units of the terms in Eq. (1) are typically Curies *per year*. This long averaging period is usually employed to average out some of the noise in an individual sample datum and to eliminate effects of transients in the river system. Also, releases from Hanford Operations, such as N_N from N-Springs, are reported on an annual basis.

If each of the three concentrations and both of the flow rates in Eq. (1) were constant over time, then the averaging (indicated by the overbars) would be very easily accomplished. However, none of these concentrations or flow rates are constant. Instead, they are always changing.

These changes over time can be understood by examining the term describing contaminant entry with groundwater at HRM 28 in Eq. (1): $\bar{c}_{ground}\bar{Q}_{ground}$. Mathematically, the annual entry of contaminant is more accurately described as the annual average of the product of the instantaneous concentration c_{ground} of contaminant in the groundwater multiplied by the instantaneous ground water flow rate Q_{ground} . That annual average would be expressed as $\overline{c_{ground}Q_{ground}}$.

This average of the product might conceivably differ substantially from the product of the averages $\bar{c}_{ground}\bar{Q}_{ground}$ in Eq. (1). However, the tritium concentration in groundwater from nearshore wells (such as 40-1 and 41-1 which are roughly 2000 feet inland of HRM 28) is found to change by only four

percent, about its mean value, over a year [15]. That is, c_{ground} is essentially a constant equal to $\bar{c}_{ground}$, and therefore:

$$\bar{c}_{ground}\bar{Q}_{ground} = \overline{c_{ground}\,Q_{ground}} \qquad \text{(B1)}$$

The averaging of the river sample data, c_{down} , c_{up} , and Q_{river} is more difficult, as now is demonstrated. The annual, downstream-upstream difference in river burden of a contaminant is $\kappa\overline{(c_{down} - c_{up})\,Q_{river}}$, where κ is the concentration factor given by Eq. (A9), which relates sampler concentrations to bulk, river water concentrations. This annual difference in river burden of a contaminant can be related to the annual average of the product $\kappa(\bar{c}_{down}-\bar{c}_{up})\bar{Q}_{river}$ by defining an averaging factor α :

$$\alpha = \overline{(c_{down} - c_{up})\,Q_{river}} \;/\; [(\bar{c}_{down}-\bar{c}_{up})\bar{Q}_{river}] \qquad \text{(B2)}$$

which is included in Eq. (1). The problem, then, is to valuate α from measurements and observations.

Downstream and upstream river samples have usually been collected over the same time interval, which ranges between a few weeks and three months, depending on the contaminant and the yearly program. The contaminant concentrations in these samples are not weighted for river flow. Thus, if the annual mean value of the difference between the downstream and upstream contaminant concentrations is to be used, then an *effective* river flow corresponding to $(\bar{c}_{down}-\bar{c}_{up})$ must be determined. This determination is based on consideration of the relations between groundwater discharges and river flow. The following discussion is specific to tritium but applies to other contaminants with some modification.

The flow of groundwater out of the river bank at HRM 28 is visibly stopped when river level rises, with river water recharging the bank. When river level again falls, discharge of water from the bank resumes. This discharge is, at first, almost entirely the river water that had been stored in the bank. As time passes and the river level continues to fall, almost undiluted groundwater begins to flow. Thus, the concentration of tritium in the river bank springs rises as time passes and river level remains low, as shown for accompanying nitrate in Fig. 8.

While the concentration rises as the river water which was stored in the bank is flushed out, the discharge rate diminishes as the river water which was temporarily stored in the bank is depleted. Figure B1, below, shows that the rate of contaminant discharge (=cQ) decreases moderately after river level drops to a constant level.

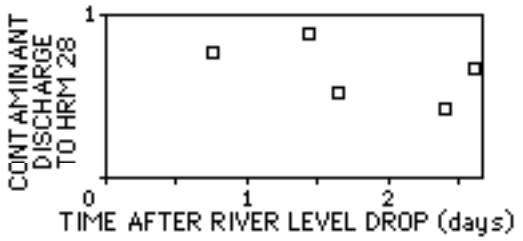


Fig. B1. Nitrate discharge response time at HRM 28

Substantial discharge of a contaminant such as nitrate or tritium from the shoreline springs at HRM 28 is seen to begin within a few hours after river level drops.

River level through most of the Hanford Reach is determined by the flow released from Priest Rapids Dam. That flow varies on a daily cycle to meet electric power demand and on longer term bases to accommodate water delivered to Priest Rapids Reservoir from the upstream dams. The ratio of the average of the minimum gaged flow at Priest Rapids to the mean annual flow (1986) is plotted in Fig. B2, below, for various numbers of days for which the minimum flow is counted.

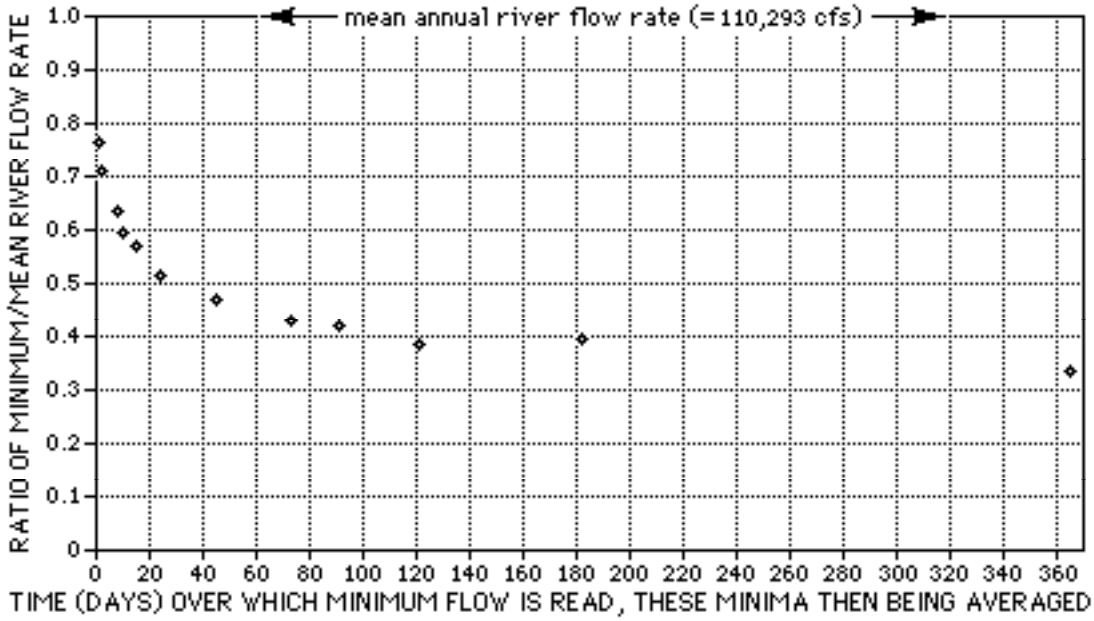


Fig. B2. Ratio of minimum flow to mean river flow for various intervals

For example, the average minimum flow over a two-day period is seen to be 71 percent of the mean annual flow.

The daily cycle of river flow is surprisingly large: The annual range is only 2.8 times the daily range. This large daily range of river level, together with the rapid response of nitrate discharge shown in Fig. B1, implies that tritium is

probably effectively discharged near the time of daily low water. This implication is supported by 7-day records of downstream-upstream tritium differences which do not correlate to river flow, suggesting a time of less than 7 days to discharge riverbank storage.

As an illustration of the situation, suppose that the river has only two flows: a high flow condition, say $\bar{Q}_{river}+\Delta\bar{Q}$ and a low flow condition, say $\bar{Q}_{river}-\Delta\bar{Q}$. Assume that each of these flow conditions occurs half of the time. Further, assume that there is no tritium discharge from HRM 28 at the high flow condition so that the sampler at HRM 47.6 samples only upstream tritium $\bar{c}_{up}$, and that the concentration measured at HRM 47.6 during the low flow condition is $2(\bar{c}_{down}-\bar{c}_{up})$. Then the average tritium concentration above the upstream background measured at HRM 47.6 would be $[0+2(\bar{c}_{down}-\bar{c}_{up})]$ / 2 = $(\bar{c}_{down}-\bar{c}_{up})$, which is the average difference in concentration measured in river water. Similarly, the average flux (Curies) of tritium added to the river would be

$$\frac{0.57(0)(\bar{Q}_{river}+\Delta\bar{Q})+0.57(2)(\bar{c}_{down}-\bar{c}_{up})(\bar{Q}_{river}-\Delta\bar{Q})}{2}$$

with κ=0.57. This reduces to

$$0.57(\bar{c}_{down}-\bar{c}_{up})\bar{Q}_{river}(1-\Delta\bar{Q}/\bar{Q}_{river}) \quad .$$

That is, the average amount of tritium passing HRM 47.6 is less than the average concentration-difference-times-river-flow rate $0.57(\bar{c}_{down}-\bar{c}_{up})\bar{Q}_{river}$ by the amount $(1-\Delta\bar{Q}/\bar{Q}_{river})$. This last factor is the amount that river flow is effectively below average river flow $\bar{Q}_{river}$ when the shoreline springs at HRM 28 discharge tritium to the river.

Based on these considerations, the response time for most river bank discharge is estimated to lie between one and five days. From Fig. B2, the factor $(1-\Delta\bar{Q}/\bar{Q}_{river})$ would correspondingly lie between about 76.3 percent (one-day-minimum/mean) and 63.3 percent (five-day-minimum/mean). A middle estimate is proposed:

$$\alpha = (1-\Delta\bar{Q}/\bar{Q}_{river}) = 0.70 \qquad \text{(B3)}$$

APPENDIX C.
CONCERNING FLOW TUBES

In this appendix, the kind of results which can be expected from water well data used in conjunction with computer flow models are compared to the kind of results obtainable from the contaminant flux/accounting approach.

The thousands of wells which tap the unconfined and confined aquifers at Hanford provide some contaminant concentration c_{well} information. With the

addition of sophisticated computer models which describe groundwater pathways and flow rates Q_{well}, the impacts $c_{well}Q_{well}$ of the contamination can be estimated.

Suppose, for example, that Hanford Operations dispose of waste water at a constant rate Q_{in} near the "200 East" Area, and that the water table is not changing over time. Then the paths which the waste water (now called *groundwater*) will follow can be estimated from the water table. Under most conditions, the groundwater will travel perpendicular to the contours of the water table. Figure C1 shows the locations of wells in which groundwater level is measured and some of the contours near "200 East" Area.

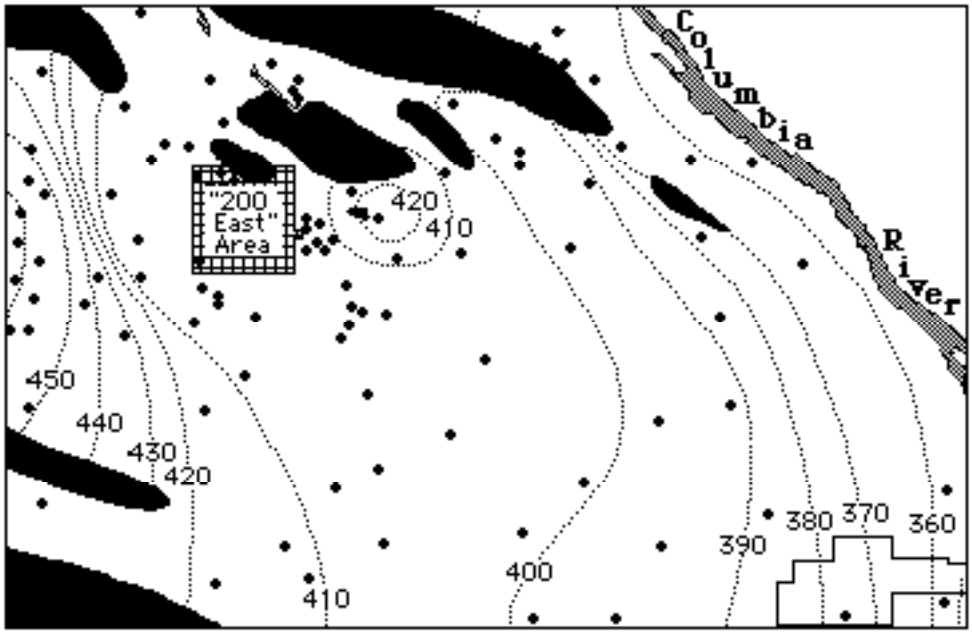


Fig. C1. Water table in 1985 (feet MSL) [36]

Figure C2, below, sketches smoothed flow paths perpendicular to water table contours of Fig. C1.

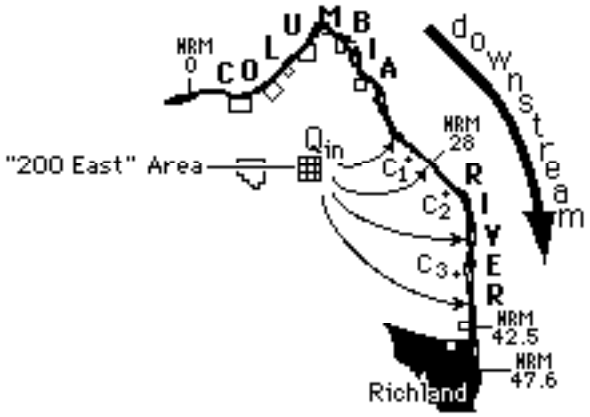


Fig. C2. Approximate flow paths

If groundwater flows parallel to the four flow paths (called *streamlines*) sketched, then the groundwater which leaves "200 East" Area between any two streamlines will enter the river between those same two streamlines. The region between the streamlines may be called a *stream tube*, here considered in two dimensions only.

Now suppose that there are wells which are sampled between each pair of streamlines, yielding contaminant concentrations c_1 , c_2 , and c_3 , respectively, as shown in Fig. C2. Finally, suppose that the corresponding flow rates in the three streamtubes are Q_1 , Q_2 , and Q_3 such that

$$Q_1 + Q_2 + Q_3 = Q_{in} \quad , \tag{C1}$$

Then the impact on the river is calculated to be

$$c_1Q_1 + c_2Q_2 + c_3Q_3$$

The major problem which a computer model builder faces is the assignment of accurate flow rates Q_1 , Q_2 , and Q_3 in Eq. (C1). This assignment requires a "very subjective" interpretation of fairly difficult measurements of soil properties [37].

For groundwater contaminants such as tritium, concentrations at locations such as the three wells marked in Fig. C2 may vary by a factor of 1000 or more. For this example, suppose the actual impact is

$$1000Q_1 + 10Q_2 + 1Q_3$$

and the total flow is

$$Q_1+Q_2+Q_3=Q_{in}=1 \quad .$$

If most of this flow follows the first stream tube, then the impact might be

$$1000(0.8)+10(0.1)+1(0.1)=601.1 \quad .$$

If most of this flow follows the third stream tube, then the impact might instead be

$$1000(0.1)+10(0.1)+1(0.8)=101.8 \quad .$$

Thus, the reported impact of the contamination can easily change by a factor of 6 at the discretion of the hydrologist. Clearly, the hydrologist faces a difficult decision in the application of professional opinion.

Given the difficulty of apprising the relative flows down the different flow tubes, the hydrologist may be forgiven for assuming that the flows in tubes of similar widths are similar. In this example, the resulting estimate would be

$$1000(0.33)+10(0.33)+1(0.33)=333.63 \ .$$

This example shows how the use of well data in conjunction with flow models introduces substantially subjective opinions describing Hanford impacts. In the case of the discharge at HRM 28, the error was probably close to a factor of 25 [38]. Such an error is distressingly large for a groundwater flow regime as well studied as the unconfined aquifer at Hanford. Further, the well data/model approach is seen inherently to underestimate Hanford impacts rather than simply to erroneously estimate those impacts.

The flux/accounting approach which has been used in this report allows direct measurement of flow rates Q_i and impacts c_iQ_i at the river ends of "i" important stream tubes and measurement of a combined impact $(\bar{c}_{down}-\bar{c}_{up})\bar{Q}_{river}$ in the river. The difference, suitably averaged, provides a measure of the net impact of all unknown sources:

$$(\bar{c}\bar{Q})_{unknown} = (\bar{c}_{down}-\bar{c}_{up})\bar{Q}_{river} - \sum_i \bar{c}_i\bar{Q}_i \qquad \text{(C2)}$$

This allows an assessment of the importance of unknown contaminant sources and unidentified pathways (e.g., iodine-129 described in this report). In short, flux accounting addresses the most significant pathways first and allows lesser pathways to be quantified as the accounting is refined.

REFERENCES and NOTES

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[2] Figure is digitized from, D.J. Brown and W.A. Haney, **The movement of contaminated ground water from the 200 Areas to the Columbia River**, HW-80909, 1960, Fig. 5.

[3] Figure is digitized from Fig. 5 of W.D. McCormack and J.M.V. Carlile, **Investigation of ground-water seepage from the Hanford shoreline of the Columbia River**, PNL-5289, Nov. 1984.

[4] For example, see D.R. Friedrichs, C.R. Cole, and R.C. Arnett, **Hanford pathline calculational program - theory, error analysis and applications**, ARH-ST-149, Atlantic Richfield Hanford Company, June 1977, App. E.

[5] Compare Fig. II.3-15 of Ref. [18] to Fig. 3 of K.R. Price, **Environmental monitoring at Hanford for 1985**, PNL-5817, May 1986.

[6] **Subsurface transport of radionuclides in shallow deposits of the Hanford Nuclear Reservation, Washington -- review of selected previous work and suggestions for further study**, U.S. Geological Survey Open-File Report 87-222, 1987, pp.28-35.

[7] N. Buske and L. Josephson, **Summer 1986 data report**, DR-2, SEARCH T.S., 1986.

[8] N. Buske, letter to K.R. Price, dated 24 Dec. 1986; see also PNL's annual environmental monitoring and surveillance reports.

[9] Ref. [6], pp.43-46, App. D.

[10] H.R. Fischer, et al., **Mixing in inland and coastal waters**, Academic Press, 1979, Chap. 5.

[11] Ref. [7], Fig. 11.

[12] P.A. Eddy and J.S. Wilbur, **Radiological status of the ground water beneath the Hanford Site**, PNL-3768, April 1981, Table 4.

[13] Figure processed from M.J. Sula, et al., **Environmental Surveillance at Hanford for CY 1982**, PNL-4657, May 1983, Fig. 5; and other USGS and PNL data.

[14] see **Disposal of Hanford defense high-level, transuranic and tank wastes**, DEIS, DOE/EIS-0113 (vol.3), App. Q.

[15] Ref. [25], Fig. 3.14 and Table A.12.

[16] Ref. [3], Tables B.2 and C.1.

[17] M.J. Sula, et al., **Environmental Surveillance at Hanford for CY-1981**, PNL-4211, May 1982, p.14.

[18] **Waste management operations, Hanford Reservation, Richland, Washington**, ERDA-1538-VI, vol. 1, Dec. 1975, Fig. II.3-21.

[19] Compare Ref. [18] to the map, **Richland Quadrangle, Washington**, 7.5 minute series, AMS 2176 IV SE, Geological Survey, 1978.

[20] Ref. [23], Table A.21; **Significance of confined aquifer iodine-129 data to the Basalt Waste Isolation Project: a Rockwell/BWIP position**, USDOE FOIA release Item 000470, undated, Appendix, 1977 and 1978 data; **Environmental monitoring report on the status of groundwater beneath the Hanford Site January-December 1975**, BNWL-2034, Table 3.

[21] Ref. [18], Fig. 3; 1984/85 data from Ref. [22], Table 3-2; A.G. Law and A.L. Schatz, **Results of the Separations Area ground-water monitoring network for 1985**, RHO-RE-SR-86-24 P, Fig. 22.

[22] **Environmental assessment, reference repository location, Hanford Site, Washington**, Vol. 1, DOE/RW-0070, May 1986, Table 3-12.

[23] For example, see **Significance of confined aquifer iodine-129 data to the Basalt Waste Isolation Project: A Rockwell/BWIP position**, undated, USDOE release 000470.

[24] Figure is digitized and simplified from R.K. Ledgerwood and R.A. Deju, **Hydrogeology of the uppermost confined aquifers underlying the Hanford Reservation**, ARH-SA-253, March 1976, Fig. 4.

[25] **Environmental Monitoring at Hanford for 1986**,PNL-6120, May 1987, Tables A.30, A.31, and G.5.

[26] The N-Reactor operating contractor has assumed that about half of the water discharged to the trenches follows pathways along which contaminant concentrations are low. The alternative of

assuming that high contaminant concentrations identify areas of high, groundwater flow rates is more conservative and probably more reasonable. See Ref. [7].

[27] Ref. [3], Table B.4.

[28] Ref. [25], p.4.3.

[29] Ref. [10], p.112.

[30] **Water Resources Research**, *The effect of bends on dispersion in streams,* **5**(2), April 1969, 496-506.

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[32] see N. Buske and L. Josephson, **Spring 1986 Data Report**, Table 3, and Ref. [7], Table 2.

[33] J.C. Sonnichsen, Jr., D.A. Kottwitz, and R.T. Jaske, **Dispersion characteristics of the Columbia River between River Miles 383 and 355**, BNWL-1477, October 1970.

[34] The Washington Public Power Supply System declined to make diffusion data relating to their nuclear plants at Hanford available for review.

[35] For an example of this averaging, see USGS Table D2. The concentration factor κ does not actually apply to discharges from N-Springs: Employing a lateral diffusivity D_y=2 ft^2/sec from N-Springs at HRM 9 downstream to HRM 12, D_y=6 ft^2/sec from there to HRM 31, and D_y=15 ft^2/sec from there to HRM 47.6, groundwater from N-Springs is found to be mixed across the river at the downstream sampler. Thus, the Accounting Equation (1) should be written,

$$\alpha(\bar{C}_{down}-\bar{C}_{up})\bar{Q}_{river} = (\bar{C}_{ground}\bar{Q}_{ground})/\kappa + N_N + N_{other}$$

so that κ is applied only to the groundwater intrusion at HRM 28. This technically correct form has not been used in this report because neither $\alpha(\bar{C}_{down}-\bar{C}_{up})\bar{Q}_{river}$ nor $(\bar{C}_{ground}\bar{Q}_{ground})/\kappa$ is an actual burden of contaminant in Curies/year. Furthermore, for the contaminants described in this report, the effect is minimal. The greatest effect is on the estimation of $\bar{Q}_{ground}$=7.7 cfs leading to Eq. (3). Using the preferred form of of the conservation equation, the average groundwater flow at HRM 28 is calculated to be $\bar{Q}_{ground}$=8.4 cfs. However, this change does not affect Eq. (3), namely: $\bar{Q}_{ground}$=8±2 cfs.

[36] A.L. Schatz, **Unconfined aquifer water-table map, June 1985**, H-2-38396 Rev. 19, Rockwell.

[37] D.B. Cearlock, K.L. Kipp, and D.R. Friedrichs, **The transmissivity iterative calculation routine - theory and numerical implementation**, BNWL-1706, May 1975. PNL's smooting technique eliminated high-speed pathways from the model:

> The method, as used in the application to the Hanford unconfined aquifer, is very subjective and allows the hydrologist to use his knowledge on the hydrology of the aquifer in determining the hydraulic conductivity distribution [p.2].
>
> It is physically reasonable that there should be a smooth variation of conductivity along a reference line crossing a group of adjacent streamtubes. Thus, the hydrologist draws what he feels is the best curve defining the conductivity values along the reference line using all of the data he has available to him [p.23].

[38] Refs. [7], p.16 and [6], pp. 38-39.