5.0 CHEMICAL FATE AND TRANSPORT

5.1 INTRODUCTION

This section presents the results of the fate and transport analyses of principal chemical pathways from the terrestrial portion of the Bremerton Naval Complex OU B to Sinclair Inlet. Two transport pathways were of primary concern: direct groundwater discharge to the inlet and surface water discharge to the inlet (via stormwater and drydock discharges). Section 5.2 focuses on the groundwater pathway in terms of Washington State marine surface water quality standards (WAC 201A) and marine sediment quality standards (WAC 173-204). Section 5.2 also addresses the terrestrial soil to groundwater pathway consistent with applicable state standards (WAC 173-340-747). Section 5.3 summarizes estimated mass loadings from the surface water pathway and compares estimated groundwater and stormwater mass loadings. Section 5.4 analyses the soil vapor pathway, as required by MTCA (WAC 173-340-745).

Although the findings presented in Section 4 show that many of the highest concentrations of chemicals of interest (COIs) are found in the nearshore, the overall spatial pattern of chemical concentrations in marine sediment appears fuzzy and variable in the nearshore areas of OU B. More distinct patterns would be expected if chemicals from the terrestrial environment had been persistently transported to the marine environment. The empirical evidence thus suggests that ongoing groundwater discharges are not the primary cause of observed chemical contamination in sediments and that, by implication, active remediation of groundwater is not justified.

This section summarizes analyses conducted to estimate potential transport of chemicals from terrestrial areas of the naval complex to the marine waters and sediments of Sinclair Inlet. These analyses were undertaken partly as a check on the findings in Section 4.

The Section 4 findings were reviewed to identify a short list of representative chemicals suitable for detailed analysis of chemical fate and transport. An emphasis was placed on chemicals frequently detected in terrestrial and marine OU B at concentrations exceeding regulatory criteria. The intent was to identify a group of chemicals to serve as surrogates for the longer list of chemicals detected at OU B. The implication is that if the fate and transport analyses indicate little or no risk of marine sediment or water column contamination due to transport of the short-listed chemicals from terrestrial OU B, risks associated with other chemicals can also be dismissed.

Five inorganic chemicals—arsenic, copper, lead, mercury, and zinc—and the organic chemical compounds known as PAHs were initially identified as candidates for fate and transport analyses. As shown in Section 4, the inorganic chemicals have been frequently detected and identified as COIs in OU B soil, groundwater, and marine sediment. PAHs were frequently detected in soil and marine sediments and were identified as COIs in soil, sediment, and groundwater. These chemicals were suggested as possible candidates for analysis during discussions in October 1997 with Ecology and EPA staff regarding the approach and scope of the FS. Regulatory agency staff agreed that these were appropriate chemicals for detailed fate and transport studies and suitable surrogates for the full list of chemicals detected at the naval complex . It was subsequently agreed to add PCBs to the short list of chemicals for fate and transport analyses given the role of PCBs as the basis for the recently completed OU B marine sediment cleanup. These seven COIs—arsenic, copper, lead, mercury, zinc, PAHs, and PCBs—thus became the chemicals of concern (COCs) for the fate and transport analyses.

After consultation with regulatory agency staff beginning in 1997, three nearshore, terrestrial groundwater areas were selected as critical source areas for the groundwater pathway. Many of the highest concentrations of the chemicals to be used in the fate and transport analyses were detected in samples collected from these areas of OU B. These areas are also the primary OU B locations where the groundwater to inlet pathway is comparatively direct, without intervening quaywalls or bulkheads. The three areas are Site 10 West in the western part of OU B; Site 1, on the shore in the center of OU B; and Site 8 (demolished Building 106 underground tanks area), in the eastern part of OU B. The results of the analyses of groundwater pathways for Sites 10 West, 1, and 8 are compared with the estimated stormwater mass loadings from the western, central, and eastern areas of the site, respectively.

Estimates of chemical transport for the groundwater and stormwater pathways make use of information contained in Section 5 of the original 1996 draft RI. This draft RI material is included in this final version of the RI as Appendix RR.

5.2 GROUNDWATER PATHWAY

This section summarizes the analytical models used to estimate potential transport of chemicals by groundwater that discharges directly to marine surface waters and sediments of Sinclair Inlet. The analytical models do *not* include either groundwater flow to the drydocks or surface water discharge from the drydocks. The drydocks are discussed as part of the surface water pathway in Section 5.3.

The analytical models were developed to supplement and complement the groundwater chemical fate and transport discussions in the draft RI (URS 1996b, Appendix RR) and to provide a practical means to analyze potential impacts on the marine environment. The numerical MODFLOW/MT3D groundwater modeling used in the draft RI was not extended to estimate potential impacts to the marine environment because of the limitations of that modeling approach to practically deal with the inherent complexity, variability, and uncertainty in conditions that affect marine protectiveness at the Bremerton Naval Complex.

Analytical Models. The analytical models address the groundwater pathway that starts with chemicals in site groundwater. Groundwater that discharges to Sinclair Inlet transports these chemicals to the marine surface waters and sediments of the inlet. The models are used to estimate quantities called "marine protectiveness measures." The three marine protectiveness measures and the models used to estimate them are as follows:

- Chemical mass fluxes to Sinclair Inlet. Estimates are made using the Chemical Flux Model.

- Offshore mixing lengths or distances required to meet Washington State marine surface water standards (SWS) (WAC 201A). Estimates are made using the Surface Water Mixing Length Model.

- Duration of time before groundwater flux raises chemical concentrations in the biologically active upper 10 cm of nearshore marine sediment to levels above Washington State sediment management standards (SMS) sediment quality standards (SQS) (WAC 173-204). Estimates are made using the Sediment Time Duration Model.

Model estimates of these protectiveness measures were intended to provide a technically defensible basis for evaluating the potential need to develop remedial alternatives in the FS for control of chemical transport via groundwater discharge from the Bremerton Naval Complex to the inlet. Neither the analysis nor its results precludes a requirement for appropriate long-term groundwater monitoring based on the remedy selected in the ROD.[1]

Further, the estimates are scientific approximations and, like all estimates, must be interpreted. The model results should be interpreted in the context of their purpose and with an adequate understanding of their assumptions, hypotheses, and limitations. This section is intended to provide that understanding.

[1]It is anticipated that the ROD will incorporate the standard CERCLA requirement that long-term monitoring results be reviewed every 5 years. Monitoring results indicating that site conditions differ from model predictions or that site conditions are changing could trigger future discussions between the Navy and agencies on the possible need for additional cleanup.

The protectiveness models are summarized in Sections 5.2.2 through 5.2.4, with supporting analyses and details in Appendices TT through XX. Results using the analytical models are presented in Section 5.2.5 for five inorganic chemicals of concern (COCs) and two organic COCs.

Chemicals of Concern. As discussed above, seven COCs were selected for detailed analysis based on discussions with regulatory agency staff. The five inorganic COCs were

- Arsenic
- Copper
- Mercury
- Lead
- Zinc

The five inorganic COCs are metals, with the possible exception of arsenic, which is sometimes described as a "metalloid." The two organic COCs were

- PCBs
- PAHs

The PAHs were separated into LPAHs and HPAHs. The analyses were handled somewhat differently for inorganic and organic COCs. These differences are briefly introduced in the following.

Analysis Differences Between Inorganic and Organic Chemicals. Except for mercury, the inorganic COC concentrations in site groundwater were high enough to be detected in laboratory analyses of groundwater samples. This meant that actual, measured COC concentrations could be used to estimate protectiveness measures. Therefore, except for mercury, measured concentrations were used in the analyses. Because there were no detections of dissolved mercury, concentration estimates were based on detection limits.

In general contrast to the inorganics, the organic COC concentrations in site groundwater were generally too low to be detected in laboratory analyses of groundwater samples. The vast majority of samples showed no detections (nondetections made up, respectively, 97, 91, and 84 percent of the PCB, HPAH, and LPAH samples). The few detections of PCBs and PAHs, which are suspected to be artifacts of imperfect sampling,[2] appear to be unrepresentative of true site conditions.

[2]Only unfiltered samples were analyzed for organic compounds, consistent with common practice. Further discussion on this topic is presented in Appendix WW, as well as in Sections 5.2.2.1 and 5.2.4.10.

Representative concentrations of PCBs and PAHs in site groundwater are thus too low to be detected. Because of this, and to avoid potential concerns associated with estimating site concentrations that are below detection limits, the following "parametric approach"[3] was taken for PCBs and PAHs.

For PCBs and PAHs, the protectiveness measures were analyzed and presented as a function of site groundwater concentration taken as an independent parameter. Although ½ the minimum detection limits were considered estimates of *maximum* site concentrations, the entire range of concentrations of laboratory detection limits (i.e., the vast majority of results for PCBs and PAHs) and the detections were graphed on the same graphs of protectiveness measure versus site groundwater concentration. This parametric presentation approach (with groundwater concentration as an independent variable) has the advantage of allowing readers to draw their own conclusions as part of interpreting the results from the analyses, as presented in Section 5.2.5.

The parametric approach was used for chemical fluxes, loads, mixing lengths to meet SWS, and times to reach SQS in marine sediments. Estimates for mixing lengths were limited to PCBs because SWS are not available for PAHs.

Site Areas of Concern. As discussed in Section 5.1, three nearshore, terrestrial groundwater areas of particular concern were selected as critical source areas for the groundwater pathway at the Bremerton Naval Complex:

- Site 10 West in the western area of OU B
- Site 1, in the central area of OU B
- Site 8 (demolished Building 106 underground tanks area), in the eastern area of OU B

These sites were selected because they reasonably represent the geographic extent of OU B that is nearshore with areas containing high concentrations of dissolved inorganic COCs. The inorganic chemical concentrations assigned to each of these areas were considered a reasonable maximum for each area, based on available groundwater measurements. Moreover, because the resulting protectiveness measure estimates are based on maximum measured groundwater concentrations, estimates are effectively representative of "hot spot" conditions, not average or "typical" conditions.

[3]The "parametric approach" simply means that the protectiveness measures are graphed against groundwater concentration taken as an independent "parameter."

5.2.1 Terrestrial Soil to Groundwater Pathway

Shallow groundwater moving onto the site from upgradient upland areas flows through the terrestrial fill material underlying the site before discharging to the inlet. This terrestrial fill material is the primary source of chemicals at the site, as documented in Section 4. Chemicals in the site fill become dissolved in the groundwater in two general ways; that is, as

- Groundwater flows through the fill (saturated zone flow)
- Infiltrated surface water percolates through the fill to groundwater (vadose or unsaturated zone flow)

Via either saturated or unsaturated flow, the transport of chemicals from the solid phase in fill to the dissolved aqueous phase in groundwater is called leaching. Although leaching mechanisms can be complex and variable, it is because of leaching that chemicals in the site fill are the source of chemicals in site groundwater. In addition to leaching, a secondary source of chemicals dissolved in groundwater may result from liquid chemical spills or leaks that percolate directly to the groundwater.

Appendices W, X, and Y discuss and document the leaching characteristics of site soils for chemicals that include the inorganic COCs arsenic, copper, mercury, lead, and zinc. The information in these appendices was supplemented with the following empirical analysis based on measured chemical concentrations in soil and groundwater samples collected during the RI. These measured concentrations represent quasi-equilibrium conditions for OU B—that is, a sufficient amount of time has elapsed for migration of chemicals from soil into groundwater and the characteristics of the site (e.g., depth to groundwater and infiltration) are representative of future conditions.

Soil-Groundwater Concentration Ratios. Soil-groundwater concentration ratios for the site were calculated using measured chemical concentrations in soil and groundwater samples collected during the RI. The soil samples were taken from soil borings that were completed as groundwater monitoring wells. For the five inorganic COCs, the average concentration for the soil samples from each boring was computed. The ratio of this average concentration in soil to the dissolved concentration in groundwater was then computed for each monitoring well. The resulting soil-groundwater concentration ratios represent effective soil-groundwater partitioning coefficients, or K_d values (expressed in units of L/kg), including any associated dilution effects between soil pore water (leachate) and groundwater. Appendix WW provides details.

The soil-groundwater ratios are an estimate of the chemical concentration in groundwater associated with the chemical concentration in source soil. For example, a ratio of 1,000 would indicate that groundwater concentrations are approximately 1/1,000 (one-thousandth) of soil concentrations. These ratios can be reasonably interpreted as only sitewide or large-scale averages for several reasons:

- Natural spatial and temporal variability associated with soil-groundwater partitioning
- Aggregation effects resulting from all upgradient sources (not just the immediate area of a given monitoring well) that influence chemical concentrations in groundwater at any particular location
- Spatially sparse data and statistical variability

Because of these effects, low soil-groundwater ratios at specific monitoring well locations should not be interpreted as groundwater hot spots.

The analysis was limited to the inorganic COCs. Representative or meaningful ratios could not be computed for PCBs or PAHs. The frequency and numbers of PCB and PAH detections were inadequate, and the few detections in groundwater were suspected to be sampling artifacts. As a practical alternative, estimates of upper bound ratios for PCBs and PAHs can be based on equilibrium partitioning theory (U.S. EPA 1996; Lyman et al. 1992; WAC 173-340-747).

Results and Conclusions. Appendix WW presents the analysis results. These results for inorganic COCs indicate that soil concentrations are many thousands of times greater than groundwater concentrations. Source soil concentrations that are protective of the inlet (via the groundwater pathway) are thus thousands of times greater than the corresponding protective concentrations in groundwater. Conversely, if a particular groundwater concentration is protective of the inlet, then the corresponding protective soil concentration is thousands of times greater.

The results of Appendix WW are consistent with the general conclusion for OU B that because measured chemical concentrations in groundwater (under effective or quasi-equilibrium conditions) are protective (of surface water), then so is the source soil that led to those groundwater concentrations. These results are consistent with WAC 173-340-747, which allows deriving soil concentrations for applicable groundwater cleanup levels established under WAC 173-340-720 based on protection of surface water, using an “empirical demonstration,” as established in WAC 173-340-747(3)(f) and (9).

Plainly put, this assessment concludes, as part of the overall groundwater analysis, that site soil is protective of surface water because site groundwater is protective of surface water. The groundwater analysis documented in the remainder of Section 5.2 demonstrates a high probability that site groundwater neither exceeds SWS (WAC 173-201A) nor results in exceedances of SQS (WAC 173-204).[4] To reiterate, site soil concentrations are protective of surface water for the groundwater pathway. The remainder of Section 5.2 deals with the transport of chemicals dissolved in groundwater that discharges directly to the inlet.

5.2.2 Analytical Models of Marine Protectiveness Measures

Intended to be relatively simple yet technically defensible, the analytical models of groundwater transport of chemicals to the inlet were designed to estimate three marine protectiveness measures. To reiterate, these marine protectiveness measures and their associated models were

1. Chemical mass fluxes to the inlet, estimated using the Chemical Flux Model

2. Offshore mixing distance required to meet marine surface water quality criteria (SWS), estimated using the Surface Water Mixing Length Model

3. Elapsed time before chemicals in nearshore surficial sediments exceed marine SQS (sediment quality standard) levels, estimated using the Sediment Time Duration Model

Both the surface water mixing lengths and sediment concentration times were coupled with the chemical flux. Each analytical model included a deterministic formulation[5] and a corresponding probabilistic formulation. These two formulations are introduced in the following discussion.

Deterministic Formulation. The deterministic formulations or models represent the physics of each protectiveness measure and form the basis for the probabilistic formulation. Chemicals were assumed to be transported by groundwater as dissolved species in uniform, one-dimensional advective flow to the inlet. Chemical concentrations in the inlet were based on mass balances for assumed mixing conditions. The deterministic models are summarized in Sections 5.2.2.1 through 5.2.2.3, with supporting analyses and mathematical details documented in Appendices TT through WW.

The deterministic models were formulated as mathematical functions of input parameters that represented spatial and temporal averages over the spatial regions and temporal durations contributing to the protectiveness measures (see Vanmarcke 1983 for a rigorous theoretical development of spatial-temporal averages). The accuracy of the deterministic formulations was

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[4]It is assumed that an appropriate groundwater compliance monitoring program, consistent with WAC 173-340-720(9), will be used to demonstrate protection of surface water, based on the remedy selected in the ROD.
[5]As used here, the terms "model" and "formulation" are functionally synonymous and can be used interchangeably.

constrained by uncertainty in the estimates of the input parameters. Probabilistic methods were adopted as a practical means of dealing with this uncertainty, as discussed next.

Probabilistic Formulation. The probabilistic formulations dealt with uncertainty in the protectiveness measure estimates in a mathematically coherent way. The intent was to quantify and communicate uncertainty and avoid unnecessary or excessive compounding conservatism. Similar probabilistic methods were previously used for estimating groundwater discharge fluxes and offshore mixing distances for OU A at the naval complex (URS 1995n; and Rohrer et al. 1996).

Uncertainty was characterized as parameter uncertainty.[6] The major input parameters required by the deterministic models represent spatial and temporal averages of phenomena that are naturally variable. Generally, natural variability and limitations in available data support a wide *range* of potential parameter values while poorly supporting any single *point* value. Natural variability and data limitations thus result in parameter uncertainty.

The probabilistic formulation dealt with parameter uncertainty by including all potential values, weighted by their relative degree of likelihood. More realistic, robust, and reliable results were obtained by using this range of parameter values rather than single point values.

The following sections describe the protectiveness measures and the analytical models. The deterministic formulations are discussed first, followed by the probabilistic formulations. Table 5-1 lists the protectiveness measures and their model input parameters. Figure 5-1 provides a simple idealized schematic of the input parameters. Mathematical details are presented in Appendix TT, with supporting development in Appendices UU and VV.

5.2.2.1 Chemical Flux Model

Flux measures the rate at which chemical mass in groundwater enters the inlet. Specifically, flux is measured as the mass of chemical per unit area and time that enters the inlet via groundwater discharging directly to the inlet. The flux was measured as a “unit flux” per linear foot of shoreline (mass per unit time per foot). The unit flux was used to estimate the “chemical load” for a given total length of shoreline (mass per unit time).

[6] For the analyses, parameter uncertainty is intended to include effects of *model uncertainty*. There are tradeoffs between model complexity (or “accuracy”), model uncertainty, and parameter uncertainty. For example, parameter uncertainty can increase with model complexity such that total uncertainty is unchanged. In this case, “simple models are better.” Model complexity that increases accuracy without compromising increases in parameter uncertainty decreases uncertainty and error. In these cases, in terms of accuracy, “more complex models are better.” See, for example, NRC 1990.

Because flux measures the rate at which chemical mass is transported to the inlet via groundwater discharge, it directly influences the chemical concentrations in marine surface water and marine sediments. Flux is thus the basic measure of chemical transport for the groundwater pathway.

The chemical flux was estimated using the Chemical Flux Model, which is fundamental to the other two models, the Surface Water Mixing Length Model and the Sediment Time Duration Model. Because of the fundamental importance of this model to the other two models, important common aspects of the analyses are discussed here as part of the flux model.

The Chemical Flux Model was based on Darcian advective transport, a special case of the more general advection-dispersion equation (e.g., NRC 1990). More specifically

- The unit flux for each COC was approximated by the unit advective flux for each COC. The unit advective flux was estimated as the product of the groundwater discharge rate to the inlet and the concentration of the COC dissolved in the groundwater.

- The groundwater discharge rate (water mass per unit time per foot) was estimated as the product of the specific discharge (water mass per unit time per unit area) and the area of discharge. The specific discharge was estimated from Darcy's law as the product of the hydraulic conductivity of the terrestrial soil and the average hydraulic gradient (equivalent to a freshwater gradient).

- The concentrations of the COCs dissolved in groundwater were estimated from monitoring wells sampled during the RI.[7]

Important aspects of the flux analysis are discussed in the following paragraphs. The discussion starts with issues related to chemical concentrations in groundwater.

Chemical Concentrations in Groundwater. The “true” chemical concentration reflects the true chemical mass transported by groundwater flow to the inlet. This true concentration may generally include both a dissolved phase and a transportable colloidal phase. As explained in the following paragraphs, the true concentrations were estimated from filtered or “dissolved” samples for inorganic COCs. Only unfiltered or “total” concentrations were available for organic COCs because filtered samples were not collected in the RI for organic chemicals.

[7]Because maximum measured concentrations were used in the various protectiveness measure analyses, resulting estimates are representative of hot spot conditions, not average or typical conditions.

COC concentrations were estimated from groundwater samples collected in monitoring wells. Seepage and gravity forces during sampling cause soil particles from the formation to enter the monitoring well. Chemicals attached or "sorbed" to these suspended soil particles become part of the groundwater sample and are included in the unfiltered total sample concentration—even though these soil particles do not flow with groundwater moving through the soil. Total (i.e., unfiltered) concentrations can thus include a sampling artifact or bias that overestimates the true concentrations that flow with groundwater.

Filtered or dissolved samples are used to reduce the potential bias associated with total concentrations. Filtering removes suspended particles larger than the filter pore size of 0.45 micron, which is at the very upper end of the colloid range (e.g., E.M. Thurman, *Organic Geochemistry of Natural Waters*, 1985, as cited in Lyman 1992). Particles larger than colloids (i.e., larger than 0.45 micron) are unlikely to flow with groundwater. Filtering does not remove either colloids or suspended particles less than 0.45 micron.

Filtering thus removes suspended particles that are larger than colloid size. These particles are suspended in groundwater samples by sampling procedures and are therefore unrepresentative of the true groundwater concentration. To the extent filtering passes particles that do not flow with site groundwater, chemical concentrations from dissolved analyses will also exceed the true chemical concentrations that flow with site groundwater. Nevertheless, in practical terms, filtering provides the most reasonable measurement of concentrations that are being transported in site groundwater.

With the foregoing in mind, groundwater chemical concentrations were handled differently for inorganic and organic COCs. These differences were introduced in Section 5.2. Building from that introduction, the following discussion expands on the inorganic COCs.

Inorganic COCs. Except for mercury, the inorganic COC concentrations in site groundwater were high enough to be detected in laboratory analyses of groundwater samples. Therefore, measured COC concentrations were used to estimate fluxes and resultant protectiveness measures.

For inorganic COCs, the *maximum* dissolved COC concentrations in samples measured during the RI in nearshore monitoring wells were used as the dissolved COC concentrations in discharging groundwater.[8] Maximum concentrations overestimate the flux contribution from areas having concentrations less than maximum. Because all points have not been sampled, the true spatially averaged concentration is uncertain. Using the maximum measured concentration increases the probability that the true average has not been underestimated. Based on

[8]Because lead at Site 1 and zinc at Site 8 had no reported dissolved concentrations, the highest total result was used to set upper limits on potential dissolved concentrations.

discussions with regulatory agency staff, maximum measured concentrations were used as an expedient and simple way to deal with uncertainty in dissolved COC concentrations.

Organic COCs. In contrast to the inorganics, the organic COC concentrations in site groundwater were generally too low to be detected in laboratory analyses of groundwater samples. Because the vast majority of samples showed no detections, measured organic COC concentrations were not directly used to estimate fluxes and resultant protectiveness measures. Instead, a parametric approach was used, as discussed in Section 5.2.

This distinction between how inorganic and organic COCs were handled in the analyses is kept tacit in the following discussions. This allows the important concepts to be expressed without loss of generality while avoiding the needless distraction of repetitively qualifying that the organic COCs were analyzed using a parametric approach.

Groundwater COC Concentrations and Saltwater Dilution Effects. Chemical flux was approximated as advective flux from groundwater discharging to the inlet. The dissolved COC concentrations in discharging groundwater were estimated from samples measured in monitoring wells. Before the downgradient discharge point (the terrestrial/marine interface), these chemical concentrations may be reduced by seawater mixing due to natural tidal advection and dispersion effects. Groundwater concentrations measured upgradient of the discharge point may thus overestimate the true groundwater concentrations at the discharge point. However, although COC *concentrations* may be reduced, the overall chemical *flux* would not be significantly changed by seawater mixing, as discussed in the following.

It is important to first recognize that chemical flux (mass per unit time per unit area) in groundwater (or surface water) is the sum of two components:

- Advective flux
- Dispersive flux

Advective flux is the product of chemical concentration and specific discharge. Dispersive flux is the product of concentration gradient, effective soil porosity, and a dispersion coefficient. Dispersive flux is positive in the direction of decreasing chemical concentrations. There is no dispersive flux if there is no concentration gradient.

Chemical dilution in groundwater due to tidally induced seawater intrusion and mixing can create a chemical concentration gradient with concentrations decreasing in the direction of groundwater flow. Although such a concentration gradient reduces advective flux in direct proportion to the reduced concentrations, the gradients create a compensating dispersive flux. The net effect is no change in the overall chemical flux.

This conclusion that chemical flux from discharging groundwater would not be significantly affected by seawater mixing is based on the analysis documented in Appendix UU. This appendix analysis focuses on the potential effect of chemical dilution in groundwater resulting from tidally induced seawater intrusion and mixing. This tidally induced mixing is indirectly measured by groundwater salinity. The relationships between chemical concentration, flux, and salinity are developed in the appendix.

Appendix UU also includes results of a data analysis of COC concentrations and salinity measured in groundwater samples collected during the RI from OU B monitoring wells. The measurements constitute the available concentration-salinity data for the Bremerton Naval Complex. The analysis of available data indicates no significant pattern of chemical concentration changes with salinity changes; that is, the RI data show no discernible concentration-salinity gradient. However, because of access and subsurface constraints, nearly all wells are located more than 100 feet from the shore; this generally limits the observed data pattern to site locations more than 100 feet from the inlet, precluding areas closest to the shoreline.

Yet the Appendix UU analysis decisively predicts that a measurable negative concentration-salinity gradient and resultant dispersive flux does occur upgradient of any groundwater discharge point. In practical terms, this clearly indicates that negative concentration-salinity gradients and dispersive fluxes become generally and effectively significant only in mixing zones that are *downgradient* of existing monitoring wells, an area extending inland about 100 feet. As demonstrated in Appendix UU, the upgradient advective flux entering a conservative mixing zone is equal to the total flux within and discharging (exiting) that mixing zone. This means that the total flux discharging OU B to the inlet can be appropriately approximated as advective flux using the RI concentration data, without consideration of salinity. That is, since the RI data show no discernible concentration-salinity gradient, the dispersive flux is effectively zero, making the total flux equal to the advective flux, as detailed in Appendix UU. Simply put, for any COC, the groundwater advective flux estimated using RI concentration data is the best available estimate of total flux discharging to the inlet. The uncertainty in these estimates is handled probabilistically to provide reliable results to the extent practicable.

To summarize and reiterate, the total groundwater chemical flux discharging to the inlet can be reasonably approximated by the advective flux using the available measured COC concentrations in RI monitoring wells without consideration of salinity. In particular, it would be demonstrably incorrect and inaccurate to use so-called “maximum seawater dilution corrected freshwater concentrations” to estimate the chemical flux, as detailed in Appendix UU.

Spatial Averages. Flux estimates represent both spatial averages over the discharge area of concern and temporal averages over the time period of concern. Because the parameters determining flux are naturally variable from point to point, appropriate values represent aggregate properties averaged over the spatial region contributing to the flux. In addition, the hydraulic gradient was a daily average over time, accounting for tidal and seasonal effects.

The chemical load represents the unit flux integrated over a given total length of shoreline. Chemical load was estimated as the sum over the total shoreline length of the product of the unit flux and the corresponding length of shoreline over which the unit flux was occurring. For a constant unit flux, chemical load was estimated as the product of the unit flux and the given total length of shoreline.

Time Effects. Flux was assumed to be constant over time—with respect to time averaging over tidal cycles and other short-term temporal variations as well as longer term changes. Flux time effects are discussed separately for protectiveness measures related to surface water and marine sediment.

Surface Water. At the point of groundwater discharge, the time-dependent or instantaneous flux will generally follow the tidal cycle. The instantaneous flux will exceed the time-averaged flux[9] while the tide is falling, and it will be exceeded by the time-averaged flux while the tide is rising.[10] The Appendix VV analysis demonstrates that the time-averaged flux will overestimate surface water concentrations compared to the time-dependent flux. The time-averaged flux is therefore a conservative representation of the time-dependent instantaneous flux.

Assuming that chemical flux from discharging groundwater is constant over the long term is a conservative approach for estimating chemical concentrations in surface water and for estimating concentrations in marine sediments, as discussed next.

Marine Sediments. For estimates related to marine sediment concentrations, time averaging of flux over short-term variations is appropriate. Chemicals from discharging groundwater accumulate in sediments cumulatively over time periods on the order of decades, centuries, or longer. Short-term variations have negligible effect over these time scales.

Although short-term variations in flux are not important, long-term effects could become significant. Over the long term, chemical concentrations in site groundwater, and the resultant flux, will generally decrease due to source depletion.

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[9]The time-averaged flux is the instantaneous flux integrated over a complete tidal cycle normalized by the time duration of the tidal cycle.
[10]There is a lag time between the tide cycle and the flux cycle, which is not important to this discussion.

Source depletion generally represents all natural attenuation processes. This includes the transport of chemicals out of the source zone, to the extent that it occurs, as well as all biogeochemical effects and, for organic COCs, chemical degradation. Although not strictly part of source depletion, over time the effects of natural degradation will reduce organic concentrations within the marine sediment. Source depletion will occur provided there are no significant new sources—that is, provided source control is effective (or more accurately, that net source depletion exceeds net new sources).

Because of source depletion, assuming that the current flux remains constant over the long term is a conservative approach for estimates of the time to raise marine sediment concentrations to SQS levels. More refined estimates that include source depletion effects would predict longer times.

Drydock Effects. The groundwater discharge rate was estimated from Darcy's law. In limited cases a reduction factor was included to account for flow restrictions to the inlet from quay walls or groundwater interception by drydock dewatering. For these conditions, the groundwater discharge rate was estimated as the product of the reduction factor, the hydraulic conductivity of the terrestrial soil in the discharge region, the hydraulic gradient in the discharge region, and the area of discharge.

The reduction factor (R) represents the theoretical, time-averaged ratio of hydraulic gradient with quay walls and drydock dewatering to the hydraulic gradient in the absence of these two effects. However, because drydock dewatering is considered much more influential than quay walls, the reduction factor is primarily a measure of drydock dewatering.

Setting the reduction factor to 1.0 (R=1.0) simulates groundwater discharge and flux in the absence of quay walls or drydock dewatering. The reduction factor is less than 1.0 to the extent the hydraulic gradient is reduced by quay walls, drydock dewatering, or both, and is less than zero (i.e., negative) for a net gradient into the shipyard from the inlet.

The most important utility for the reduction factor has to do with estimating flux under the "worst case" scenario of no drydock dewatering. That is, by simply setting the reduction factor to an exact value of 1.0 (R=1.0), the effect of quay walls and drydock dewatering disappears from the flux estimates; R=1.0 is equivalent to no (zero) reduction in groundwater discharge.

A reduction factor of less than 1.0 (R=0.02) was used only to estimate groundwater flux and mixing lengths for Sites 1 and 8 for the particular case "with drydock dewatering" because these sites are strongly influenced (in fact, dominated) by drydock dewatering. Estimates for these sites for the case "no drydock dewatering" used no reduction (R=1.0). Although Site 10 West may be affected by a partial quay wall, to be conservative, no reduction was assumed (i.e., R=1.0) for both cases ("with" and "no" drydock dewatering).

It is emphasized that *no* reduction (i.e., R=1.0 only) was used to estimate potential groundwater influences to marine sediments. First, groundwater intercepted by drydock dewatering is, sooner or later, discharged to the inlet; therefore, drydock dewatering does not really alter the longer term influence of groundwater on marine sediments since these influences are cumulative over time. Second, the predicted times for recontamination of marine sediments by direct groundwater discharge to the inlet are on the order of millennia (if not "infinite"), a time horizon likely well beyond drydock dewatering.

Flow Path Idealization. Groundwater flow at the site is unconfined. Flow occurs in the upper surficial layer of random fill that varies from about to 20 to 50 ft in depth, as well as in the underlying native soils. For the following reasons, it is the groundwater flowing through the fill to the inlet that is considered the major source of chemical loading to the inlet from direct groundwater discharge:

- Fill is the source of the COCs in discharging groundwater.
- Groundwater in the fill has the highest measured COC concentrations.
- Flow paths from the fill to the inlet are of minimum lengths and maximum hydraulic gradients compared to deeper flow paths.
- Hydraulic conductivity in the fill is generally higher than in underlying native soils.

The Chemical Flux Model was thus formulated with the shallow flow in mind. Because the model is one-dimensional, the potential effects of more complex flow patterns must be interpreted. The model explicitly represents flow paths along the longitudinal direction of groundwater flow, which, in the absence of drydock dewatering, is idealized as occurring in a direction approximately perpendicular to the shoreline. In vertical cross section, this flow direction defines a groundwater flow plane. Groundwater flow in the flow plane is further idealized as consisting of two components, one relatively shallow and one relatively deep:

- Relatively shallow flow through the fill material that discharges to the inlet along the shoreline, primarily *through riprap or under quay walls*

- Relatively deep flow through natural glacial soils underlying the fill that discharges to the inlet along the nearshore seafloor, *through marine sediments*

Although the Chemical Flux Model represents primarily shallow flow, it implicitly includes a deep flow component that is lumped with the shallow flow. The parameter values chosen for the model determine how much contribution comes from shallow flow as opposed to deeper flow. Because the major source of flux is from the shallow flow, the parameter estimates emphasized the shallow flow component, as discussed in the following paragraphs.

Surface Water. Assuming that the flux estimated by the Chemical Flux Model is *entirely* shallow flow is considered appropriate for analyzing surface water mixing lengths because it gives the most conservative results (i.e., longer hypothetical mixing distances). As such, model inputs for estimating SWS protectiveness should reflect conditions associated with shallow flow, which discharges to the inlet along the shoreline primarily through riprap or under quay walls. Model output needs to be interpreted in a consistent manner, considering probable chemical behavior under shallow flow conditions.

Marine Sediments. The situation for marine sediments is more complicated because of potential influence by flux from deeper flow. However, the influence of deeper flow to the critical top 10 cm of marine sediment[11] is minor compared to the influence of shallow flow because of the following two major reasons:

- Deeper flow would generally have significantly less chemical flux than would shallow flow because of lower chemical concentrations and hydraulic gradients (from longer flow paths) that result in lower specific discharges.

- Deeper flow must travel a long distance through marine sediment before discharge to surface water. These travel distances are on the order of hundreds to thousands of centimeters. To the extent chemical mass was lost to the sediment, the portion lost within the *last* 10 cm before discharge (i.e., the biologically active zone) would be negligible.

[11]The top 10 cm of marine sediments is considered the biologically active zone, where SMS apply.

Flux from shallow flow is therefore also considered the principal source for effects to the upper 10 cm of marine sediment.

The foregoing discussion makes clear that the Chemical Flux Model requires appropriate interpretation for both input characterization and output evaluation. The Chemical Flux Model is integral to the other two marine protectiveness measures and models, which are discussed below.

5.2.2.2 Surface Water Mixing Length Model

Discharging groundwater mixes with marine surface water (seawater) from the inlet. Provided that ambient seawater concentrations are less than groundwater concentrations, the mixing causes the concentrations of chemicals in groundwater to decrease in proportion to the volume of seawater that mixes with the groundwater.

For each COC, estimates were made of the hypothetical volume of seawater needed to decrease the measured groundwater concentrations to marine SWS. These volumes were used to estimate corresponding lengths (per unit discharge area) required to decrease groundwater concentrations to SWS. These lengths, which vary with the COC, were termed “mixing length required to achieve SWS.” The lengths were assumed to start at the groundwater/seawater discharge point to the inlet.

In terms of marine SWS, the mixing lengths were considered to be a protectiveness measure representing the potential impact of chemical mass flux from discharging groundwater on marine surface water in Sinclair Inlet. For example, short mixing lengths can be assumed to represent small impacts in terms of SWS, and longer lengths would represent greater impacts.

A simple model was developed to estimate a mixing length required to achieve SWS for each COC. The model is based on the concept of a continuous mixed reactor located within the inlet at the groundwater/surface water interface. This model can be considered a crude, but conservative, representation of the advection-dispersion phenomena that occur when a chemical in groundwater discharges to surface water. The advection-dispersion equation classically used to describe this kind of phenomena[12] provides the basis of a more accurate analysis that is developed in Appendix VV and used to support the simple model described below.

[12]See, for example, U.S. EPA 1985; Boyle et al. 1974; Chang 1998; Chaudhry 1996; Ward and Montague 1996; Huber 1993; French 1985; Tschobanoglous and Schroeder 1987; Bear 1972, 1979; Tracor 1971. Chemical concentrations in both groundwater and surface water are modeled using the same fundamental differential equation, the advective-dispersion equation.

The model was developed as follows. The COC concentration from discharging groundwater and the existing, ambient seawater concentration was formulated by a mass balance in an imaginary inlet mixing volume (IMV).

The imaginary IMV was defined to start in surface water at the point of groundwater discharge and extend into the inlet a distance equal to the mixing length. By definition, the IMV precludes seawater mixing with groundwater in the terrestrial zone before discharge. This restriction ignores the potential for groundwater concentrations measured in monitoring wells to be reduced before discharge to the inlet. The IMV concept thus forces a non-zero mixing length.

The volume of the IMV was estimated as the product of the mixing length and a vertical mixing area per foot of shoreline. The vertical mixing area was set equal to the groundwater discharge area.

The IMV was assumed to be continuously mixed. This results in homogeneous COC concentrations, reflecting the mix of groundwater entering the IMV and surface water within the IMV.

The mass of COC in the IMV was estimated as follows. The COC mass from discharged groundwater was estimated as the product of the COC flux, as discussed in Section 5.2.2.1, and the time required for the IMV to be "flushed" by advection and dispersion effects. As discussed in Section 5.2.4, estimates of the IMV flushing time were based on interpretation of the advection-dispersion analysis presented in Appendix VV and supported by the complementary results of Appendix UU.

The mass of COC from ambient seawater within the IMV was estimated as the product of the ambient seawater volume (equal to the IMV minus the volume of discharged groundwater in the IMV) and the COC concentration in ambient seawater.

The entire COC mass from the discharging groundwater was assumed to remain dissolved within the IMV. That is, no reduction was assumed in COC mass in the IMV due to chemical precipitation or complexation/sorbing onto marine sediment because of the following:

- Discharging groundwater will have thoroughly mixed with seawater before discharge to the inlet.

- Estimated mixing lengths were on the order of a fraction of an inch to a few inches at most, which would provide minimal opportunity for chemical precipitation in the IMV.

- Critical discharge points for meeting SWS are at or near the shoreline, which consists primarily of riprap that has no or very limited ability to complex/sorb chemicals dissolved in groundwater.

The COC concentration in the IMV was estimated by mass balance as the dissolved groundwater COC mass plus the ambient seawater COC mass divided by the IMV. For each COC, the mixing length to achieve the SWS was estimated by equating the estimated COC concentration in the IMV to the SWS and solving for the required mixing length.

Neglecting the ambient COC concentration in seawater provided the mixing length required to achieve the SWS as if there was no COC in the ambient (mixing) seawater. This mixing length was termed the *incremental* mixing length to represent a mixing length required to reduce the discharging groundwater to the SWS in the absence of COC in the mixing seawater. This parameter was useful when the ambient seawater concentration was already above the SWS, as was the case for copper.

5.2.2.3 Sediment Time Duration Model

A separate protectiveness model was formulated for the biologically active upper 10 cm of nearshore marine sediments. The model focused on sediment chemical concentrations due to dissolved COCs in discharging groundwater becoming permanently mixed or fixed in the upper 10 cm of the sediments.

The model estimates the duration of time before chemical concentrations would exceed the SQS levels. For each COC, this time was considered to be a protectiveness measure representing the potential impact on marine sediments in Sinclair Inlet from chemicals in discharging groundwater. Long times can be assumed to represent small impacts; shorter times, greater impacts.

The duration of time before chemicals in marine sediments exceed SQS levels was formulated from a mass balance in the affected sediments. The COC concentration in affected sediments from the cumulative “fixing” (by precipitation, complexation, sorption, etc.) of dissolved COCs in discharging groundwater was estimated over time as the cumulative chemical mass at a given time divided by the sediment mass at that time. For each COC, the time at which the estimated COC concentration in the sediment reached the SQS was predicted.

The model was initially formulated so that the chemical mass in the sediment over time could also include a contribution from an initial COC mass in sediment at time zero and a COC mass from accumulating sediments before impact by groundwater discharge. Both parameters were estimated as decimal fractions of the SQS. However, because it was desired to estimate the sediment protectiveness measure in terms that were directly related to the potential future effect of groundwater discharge alone, both parameters were set to zero. Thus, the results presented in this RI assume initially clean sediments and initially clean accumulating sediments, before impact by chemicals in groundwater discharge.

The model maintained a constant sediment target thickness or depth below the mud line of 10 cm over time. Although the sediment depth of concern remains constant, the mud-line elevation increases with sediment accumulation (due to cumulative deposition over time). This causes the location of the sediments of concern to vary with time. Until the accumulated depositional sediment exceeds the 10-cm target thickness, the chemical concentration in the sediments of concern is the depth-weighted average of the preexisting (before time zero) sediment (assumed initially clean) and the chemical concentration in the depositional sediment. After the accumulated depositional sediment reaches a thickness of 10 cm, the chemical concentration in sediments from the mud line (which continues to rise with time) to the 10-cm target depth is constant with time and is equal to the chemical concentration of the accumulating depositional sediments, as explained next.

The chemical concentration of accumulating depositional sediments was estimated as the time rate of chemical mass "fixed" onto sediment (by precipitation/complexation/sorption) from discharging groundwater divided by the rate of depositional sediment mass accumulation. The rate of chemical mass was estimated as the product of (1) the chemical mass flux from discharging groundwater and (2) the proportion of flux that fixes onto the sediment. The rate of sediment mass was estimated as the product of (3) sedimentation area (set equal to the affected offshore distance normal to a 1-ft unit width of shoreline), (4) sediment dry density, and (5) sedimentation rate. All parameters were assumed to be constant over time.

Before the *depositional* sediment reaches a total thickness (depth) of 10 cm, the chemical concentration in the upper 10 cm of sediments was estimated as the cumulative chemical mass in the upper 10 cm of sediments divided by the corresponding 10-cm-thick sediment mass. The cumulative chemical mass in the upper 10 cm of sediments was estimated as the product of the chemical mass rate—the product of (1) and (2) above—and the time duration of flux discharge. The corresponding sediment mass was estimated as the product of the sedimentation area (3 above), sediment dry density (4 above), and the sediment thickness of 10 cm.

For the organic COCs (PCBs and PAHs), the sediment mass was multiplied by the percent total organic carbon to yield the mass of organic carbon in the sediment. This was necessary because the SQS for PCBs and PAHs are measured in units of chemical mass per mass of organic carbon.

Zero mixing was assumed between the accumulating sediment layer and the underlying sediment. This assumption is conservative compared to the SEDCAM model, which assumes continuous mixing of the active sediment layer (Jacobs, Barrick, and Ginn 1988). Estimated protectiveness measure times would increase if continuous mixing was assumed.

The estimated duration of time before chemicals in marine sediments exceed SQS levels became infinite above a critical sedimentation rate, termed the "limiting sedimentation rate." At or above the limiting sedimentation rate, the chemical concentration in sediment was predicted to never increase above the SQS. If the sedimentation rate was interpreted as an erosion rate, the same result occurred: erosion rates at or above the "limiting erosion rate" would prevent chemicals in sediment from ever reaching the SQS. Thus, assuming initially clean sediments:

- Erosion and deposition are equally effective at limiting chemical concentrations in marine sediments.

This conclusion also complements the fact that the concentrations represent area-weighted averages, as discussed next.

Area-Weighted Average Concentration. The Sediment Time Duration Model assumes a volumetric average chemical concentration over the affected sediment area and thickness. For a constant sediment thickness, this is equivalent to assuming an area-weighted average chemical concentration in the affected marine sediments. However, these area-weighted averages more accurately represent "hot spot" conditions, explained as follows.

The area-weighted averages implicit in the Sediment Time Duration Model have an inherently high probability of overestimating the true average for any given COC. For example, the area-weighted averages implicit in the model are about four times more conservative than the predicted area-weighted averages due to natural recovery (URS 2000) that were used to support the OU B Marine ROD (U.S. Navy et al. 2000).[13] Because the estimates are also based on maximum measured concentrations in site groundwater, *estimates are representative of "hot spot" conditions, not true area-weighted averages*.

[13]Natural recovery modeling for the OU B Marine ROD was based on the less conservative SEDCAM model and assumed a sedimentation rate about four times that used in this analysis.

The functional equivalence of erosion and deposition at limiting chemical concentrations in marine sediments further reduces the sensitivity of the results to localized extreme deviations from "average" conditions.

Proportion of Groundwater COC Mass That Fixes Onto Marine Sediment. A critical parameter in the sediment model is the proportion of COC mass in discharging groundwater that fixes onto the top 10 cm of affected marine sediment. This parameter is discussed in detail in Section 5.2.4.4.

5.2.3 Probabilistic Formulation of Marine Protectiveness Measures

Protectiveness measure estimates based on the foregoing deterministic models would be exact provided the model assumptions were strictly correct and the input parameters precisely known. Unfortunately, this proviso is unrealistic. First, like all models, the model assumptions represented simplifications and approximations. This created model uncertainty. Second, because of natural variability and unavoidably limited site information, the model input parameters were inherently uncertain. This created parameter uncertainty.

Input Parameter Probability Density Functions. Parameter uncertainty was dealt with as follows. The input parameters were characterized by probability density functions (PDFs). The input parameter PDFs included the entire range of potential parameter values. Based on interpretation of available information, the input parameter PDFs weighted each potential value by the relative degrees of confidence that *potential* value was in fact the *true* (but inherently uncertain) value.

Protectiveness Measure PDFs. The deterministic models were then formulated as probabilistic functions of the input parameters, thus allowing the aggregate uncertainty in the model predictions to be quantified. The uncertainty in each input parameter was propagated through the deterministic model using the mathematics of random variables and error propagation (Benjamin and Cornell 1970; Hahn and Shapiro 1967; Harr 1987; Ang and Tang 1975). The result was a protectiveness measure PDF that measured the cumulative or aggregate uncertainty in the protectiveness measure estimates resulting from the input parameter PDFs.

The protectiveness measure PDFs were functionally and probabilistically consistent with the input parameters and their PDFs. The underlying, deterministic functional dependence between the input parameters and the resulting protectiveness measures thus remained unchanged and implicit in the probabilistic formulation.

Model uncertainty was considered by providing compensating conservatism in the parameter estimates. Lumping model uncertainty with parameter uncertainty was both practical and justified because the parameters generally followed from the model assumptions.

Reliable Protectiveness Measure Estimates. The protectiveness measure PDFs were used to make estimates having a 95 percent probability of being "conservative." That is

- For fluxes and mixing lengths, model predictions have a 95 percent probability of *exceeding* the true (but inherently uncertain) values.[14] Equivalently, the 95 percent nonexceedance estimates have a 95 percent probability of *overestimating* the true value. These estimates can be broadly interpreted as having a "high confidence" (or certainty, probability, or likelihood) of overestimating the true values.

- For estimates of the duration of time before chemicals in sediments exceed SQS, model predictions have a 95 percent probability of *not exceeding* the true times, consistent with shorter times being of more concern than longer times.[15] Equivalently, the 95 percent exceedance estimates have a 95 percent probability of *underestimating* the true times. These estimates can be broadly interpreted as having a "high confidence" (or certainty, probability, or likelihood) of underestimating the true times.

Overall, this approach provided a mathematically coherent and reliable estimate of protectiveness while avoiding both the unrealistic compounding conservatism that occurs if parameter "worst case" estimates are used and the uncertain conservatism that occurs if parameter estimates having varying levels of conservatism are used. Ultimately, the protectiveness measure estimates, like all estimates, must be interpreted in context with an understanding of their explicit and implicit assumptions, hypotheses, and limitations.

The mathematical details associated with the probabilistic formulation of the marine protectiveness measures are documented in Appendix TT. The following section describes how the model input parameters were estimated.

[14]The 95 percent probability is termed the "95 percent nonexceedance probability" and the corresponding estimate is termed the "95 percent nonexceedance estimate."

[15]In this case, the 95 percent exceedance estimates have a 95 percent probability of being exceeded by the true (but inherently uncertain) time.

5.2.4 Model Input Parameter Estimates

This section focuses on the model input parameter estimates and the rationale for these estimates. Particular emphasis is placed on the parameters that were estimated probabilistically, which were generally those parameters that were considered significantly uncertain. Obtaining definitive data for these uncertain parameters was considered impracticable for reasons of technical complexity, cost, and timeliness.

The following model input parameters were estimated probabilistically:

- Hydraulic conductivity
- Hydraulic gradient
- Discharge area
- Proportion of inorganic COC mass that fixes onto marine sediment
- Affected sediment length
- Flushing time
- Sedimentation rate
- Sediment density

A Bayesian approach was used to assign a PDF to represent the uncertainty in each model input parameter that was characterized probabilistically. The parameter PDF was used to measure the probability that the *true* but uncertain value of that parameter would not (or would) exceed particular *estimates* of that parameter over the range of possible values. Parameter estimates reflected both natural variability and uncertainty associated with limited data.

Bayesian techniques are practical and well supported in the mathematical, scientific, and engineering literature (e.g., Benjamin and Cornell 1970; Martz and Waller 1982; Ang and Tang 1975, 1984). In estimating parameters, a Bayesian approach allows the use of all available information, both quantitative and qualitative, with professional judgment. Subjective professional interpretations are made explicit.

Input parameter PDFs were assumed to follow lognormal distributions for the following theoretical, practical, and empirical reasons. A lognormal PDF

- Prevents negative parameter values, which are physically impossible

- Allows unbounded positive values, which is generally conservative

- Is consistent with observed forms of physical parameters such as hydraulic conductivity (e.g., Krumblein and Graybill 1965), as well as environmental concentrations of trace chemicals (e.g., U.S. EPA 1992a, 1992b; Gilbert 1987; Jornel and Huijbregts 1978; Ecology 1993; Ott 1995; URSG and CH2M HILL 2000)

- Requires a minimum of statistical assumptions

- Is consistent with the central limit theorem of probability theory for phenomena arising from the product of a large number of variables

- Greatly simplifies the mathematics of products and quotients in the protectiveness models without loss of mathematical rigor or accuracy

The intent was to estimate parameter PDFs that conservatively reflected

- All potential values consistent with the available information

- Technically plausible values

- Model assumptions on which parameters were based, so that model uncertainty could be included with parameter uncertainty

Because the parameter PDFs were considered conservative, any new data can be expected to increase the degree of protectiveness reflected in the estimates.

Not all parameters were treated probabilistically. In particular, chemical concentrations were not treated probabilistically. Instead, the maximum dissolved concentrations measured in groundwater monitoring wells near the shoreline in each of the three areas of concern (Sites 10 West, 1, and 8) were used. Also, the length of shoreline used for total flux was the maximum possible length. Both cases contributed to conservative, high-biased estimates.

The information and assumptions used to estimate the model input parameters are summarized in the following sections. Generally, two exceedance or nonexceedance probability levels were used to estimate probabilistic input parameters. An exceedance probability is the estimated probability that a given parameter estimate or value exceeds the true (but uncertain) value of that parameter; a nonexceedance probability is the estimated probability that a given parameter estimate or value *does not* exceed the true (but uncertain) value of that parameter.

Exceedance and nonexceedance probabilities are mutual complements; that is, an exceedance probability is one minus the nonexceedance probability, and vice versa. As discussed in

Appendix TT, parameter estimates for any two exceedance or nonexceedance probability levels (e.g., 95 percent and 50 percent or 95 percent and 5 percent) determine the parameter's lognormal statistical measures, the expected value and coefficient of variation. These two measures in turn determine the parameter values for all other exceedance or nonexceedance probability levels.

For each input parameter, the determining exceedance/nonexceedance probabilities and corresponding parameter estimates were based on analysis and interpretation of available information. The use of professional judgment was fundamental to this process.

Table 5-2 summarizes the input parameter estimates. For the probabilistic parameters, Table 5-2 includes, for each parameter, the resultant

- Expected or average values
- Coefficient of variation (CV)
- Range of values between the 5 percent nonexceedance (or 95 percent exceedance) estimate and the 95 percent nonexceedance (or 5 percent exceedance) estimate

This range constitutes 90 percent of the potential estimates implicit in the probabilistic estimates. That is, 5 percent of the estimates are below the 5 percent nonexceedance value (with a minimum value of zero) and 5 percent are above the 95 percent nonexceedance value (with an unlimited maximum value).

5.2.4.1 Hydraulic Conductivity

Hydraulic conductivity, K, is a spatially averaged geologic property that is naturally spatially variable. In particular, hydraulic conductivity depends on the scale or volume of the groundwater flow region of interest. The principal groundwater flow region for the marine protectiveness measures is the site fill, which is the source of chemicals in groundwater discharging to Sinclair Inlet. The fill is undifferentiated and may be characterized as random mixtures of gravel, sand, silt, clay, and buried debris.

The natural variability in the hydraulic conductivity of the fill is indirectly reflected in the wide range of K values (1 to 86 ft/day) used in past groundwater analyses:

- K of 86 ft/day, used by the USGS in its groundwater modeling at PSNS (USGS 1997). This value was selected from the upper end of the range of values that were estimated from available slug tests before 1995.

- K of 1 ft/day, used in the draft RI for the sitewide numerical (MODFLOW) groundwater modeling (URS 1996b) and included here as Appendix RR.

- K of 37 ft/day, used in the OU A FS (URS 1995n). This value is the approximate average K of available slug tests (39 ft/day; URS 1996a). A K equal to the USGS value of 86 ft/day was used in subsequent groundwater modeling in support of the proposed plan for OU A (URS 1996a).

For a fill consisting of random mixtures of gravel, sand, silt, clay, and debris, a spatially averaged K of 1 ft/day is considered very low and a K of 86 ft/day is considered high.

Because of natural variability and limited site hydraulic conductivity data, it was considered impracticable to adequately define separate areas of conductivity in the fill. The fill was instead treated as one geologically heterogeneous deposit for the purpose of estimating hydraulic conductivity.[16] The variability was characterized by statistical analysis of available data, as discussed next.

Statistical Analysis of Hydraulic Conductivity Data. To quantify the natural variability, the hydraulic conductivity was estimated from a statistical analysis of conductivities measured in borehole aquifer tests. Those conductivities are presented in Table 3-13 and include the available data from the SI and OU B RI. The estimates were limited to conductivities measured in fill material. This constituted a total of 35 measurements made at 26 separate locations in OU B. At nine locations, measurements were made during the SI and repeated during the RI. For these nine locations, the repeated measurements were combined into one measurement for each location by taking the geometric mean of the repeated measurements.

A fitted linear regression line for natural-log-transformed K data indicated that the hydraulic conductivity data are lognormally distributed (r^2 equaled 0.97). A table and graph of the data and regression line are presented in Appendix XX. Statistical parameters were calculated from the regression line using the method presented in Ecology 1993. The statistical results were used to make the probabilistic K estimate that was used in the Chemical Flux Model, as discussed in the following paragraph.

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[16]That is, statistically, the fill was treated as a single "statistically homogeneous" stratum.

Probabilistic Estimate of Hydraulic Conductivity. For the purpose of estimating the three marine protectiveness measures, the hydraulic conductivity K for the site fill was characterized based on the following assumptions:

- The spatially averaged variability in K follows a lognormal PDF.
- There is a 95 percent probability that K will *not* exceed 147 ft/day and a 95 percent probability that K *will* exceed 0.74 ft/day.

This resulted in an average or expected value for K of 38 ft/day. Note that the average hydraulic conductivity in the natural materials below the fill (to 50-ft depth) was about half that of the fill material.

5.2.4.2 Hydraulic Gradient

Because hydraulic gradients across the site are spatially and temporally variable, there is no single gradient, but a distribution of gradients. The hydraulic gradient of interest is the equivalent freshwater gradient for a condition of no drydock dewatering (R=1) and direct discharge of groundwater to the inlet. The gradient is time averaged over a tidal cycle, as discussed in Appendices UU and VV.

Available data on measured groundwater levels in site monitoring wells are limited in both quantity and quality for the purpose of estimating gradients. These limitations include effects of natural variability and statistical uncertainty, the absence of synoptic data, and the confounding effects of tidal cycles and drydock dewatering on groundwater levels (hydraulic heads) measured in site monitoring wells.

The available data thus require significant interpretation and are considered to be incompatible with unique, single-valued “point” estimates of gradients—particularly for the case of no drydock dewatering, arguably the most important case for assessing gradients. To deal with the limited data, the approach taken, similar to that for the other probabilistic model input parameters, was to estimate a probability distribution for the gradient, recognizing that no single point estimate could be accurate. A gradient distribution for the site was characterized based on interpretation of relevant information presented in the following:

- OU A RI (URS 1995k)
- OU NSC RI (URS 1995l)

- OU A groundwater modeling report (URS 1996a)
- OU B draft RI (URS 1996b), which included a calibrated sitewide groundwater flow model for the Bremerton Naval Complex that analyzed conditions for the case of no drydock dewatering

In particular, the calibrated Bremerton Naval Complex groundwater flow model integrated the available site measurement data in the context of a three-dimensional MODFLOW analysis. The analysis provided a point estimate of water table elevations across the naval complex (Appendix RR, Table 5-12). From these predicted groundwater elevation contours, gradient point estimates across the site were deduced and then interpreted (recognizing their limitations), as discussed in the following. Although predictions from the calibrated Bremerton Naval Complex groundwater flow model have uncertain error, they provide a more complete basis for making gradient point estimates than could be obtained directly from available monitoring well data.

Hydraulic Gradient Estimates. An apparent hydraulic gradient of 0.007 was estimated from data collected at the western edge of OU NSC in an area that appeared to be unaffected by drydock dewatering. Analysis of data for OU A indicated an apparent gradient of approximately 0.0167 (which was used in the OU A FS). The OU A analysis also indicated that discharge of fresh groundwater from OU A was one-third to one-quarter of what it would be in the absence of tidal fluctuations, suggesting that the apparent gradient could be reduced by a factor of 3 to 4. Combining the results of the OU NSC and OU A analyses suggested that the effective gradient for areas not affected by drydock dewatering may be approximately 0.006 or less in the tidal zone.

Water table elevation contours resulting from cessation of all drydock dewatering were simulated as part of the calibrated Bremerton Naval Complex groundwater flow model analysis (Appendix RR, Figure 5-12). This simulation showed a range of nearshore hydraulic gradients from approximately 0.02 in the far eastern area of OU B to approximately 0.002 to 0.005 in the central and western areas of OU B. Because these simulated estimates did not include tidal effects, they represent equivalent time-averaged estimates of the freshwater gradients.

Probabilistic Estimates of Hydraulic Gradient. Using these available estimates and professional judgment, the effective hydraulic gradient I was characterized based on the following conservative assumptions:

- The uncertainty in hydraulic gradient i follows a lognormal PDF.
- There is a 95 percent probability that i will *not* exceed 0.03 and a 95 percent probability that i *will* exceed 0.001.

This resulted in an average or expected value for I of 0.009 ft/ft. Because it appears to be a conservative representation of the available data, this range of estimates for hydraulic gradient was considered appropriate throughout OU B, both with and without drydock dewatering. This estimate reflects the aggregate uncertainty in the interpretation of available information based on analysis and professional judgment.

5.2.4.3 Discharge Area

Discharge area varies with tidal conditions. The discharge area of interest, Aq, is the tidally averaged discharge area. Note that Aq has units of ft^2/ft, equivalent to units of feet, as used here.

The effective Aq was estimated from a conservative interpretation of 1995 bathymetric data collected along the shoreline around Mooring F, directly offshore from Site 10 West. These data show a relatively gradual mud-line slope from the terrestrial fill line to a depth of approximately 20 ft below mllw at about 100 to 150 ft from the fill line. A relatively steep slope then follows to a depth of approximately 35 ft below mllw at about 200 ft from the fill line. The slope becomes gradual to a relatively constant mud-line depth of 40 to 42 ft below mllw at about 250 ft or more from the fill line.

Groundwater modeling for OU A (URS 1996a) using computer codes MT3D and MODFLOW indicated that groundwater discharge to the inlet was maximum near the toe of the riprap fill and decreased with distance into the inlet. Groundwater discharge was negligible about 80 ft seaward of the fill toe. This would suggest an Aq not exceeding about 25 ft at OU A and an Aq less than about 20 ft near Mooring F.

Probabilistic Estimate of Discharge Area. A conservative interpretation of these data was used to make the following assumptions for parameter Aq:

- The uncertainty in Aq follows a lognormal PDF.
- There is a 95 percent probability that Aq will *not* exceed 35 ft and a 50 percent probability that Aq *will* exceed 20 ft.

This resulted in an average or expected value for Aq of 21 ft. The interpretation of Aq for Site 10 West was considered conservative enough to be appropriate for all of OU B, including Sites 1 and 8.

5.2.4.4 Proportion of COC Mass That Fixes Onto Marine Sediment

No direct measurements are available to estimate parameter Ps, the proportion of dissolved COC mass in groundwater that “fixes” onto marine sediments in the biologically active layer (top 10 cm). Fixing mechanisms may include chemical precipitation, complexation, sorption, or any other mechanism. The proportion of dissolved COC mass in groundwater that fixes onto marine sediments is measured relative to the flux based on the COC concentrations measured in site groundwater monitoring wells.

Information in the literature was reviewed to provide general background on mechanisms potentially affecting Ps (e.g., Deutsch 1997; EPRI 1984; Domenico and Schwartz 1998; Bagchi 1990; Freeze and Cherry 1979; U.S. EPA 1996; Boatman and Hotchkiss 1977; Lyman, Reidy, and Levy 1992). Based on a general interpretation of that information, Ps was estimated using professional judgment and the reasoning summarized in this section. Separate estimates of Ps were made for the inorganic COCs and the organic COCs.

Inorganic Chemicals. The potential for inorganic COCs from discharging groundwater to fix onto marine sediment was considered negligible for several reasons. The main thrust of these reasons is that, to the limited extent that chemicals in groundwater are able to fix onto sediments (or soil), the vast majority of that action will occur before the groundwater discharges to the inlet or otherwise affects marine sediments in the biologically active zone. The following discussion elaborates on the reasoning.

Effects in the Nearshore Terrestrial Environment. Before discharge, groundwater mixes appreciably with seawater in the nearshore terrestrial environment. Mixing with oxygen-rich seawater[17] increases the dissolved oxygen in groundwater. This can initiate biogeochemical

[17]Sinclair Inlet seawater is well oxygenated (average 7.9 mg/L; Washington Department of Natural Resources cited by others). Dissolved oxygen in site groundwater is generally less than 1 mg/L, according to Appendix RR, pp 5-19.

reactions[18] that can reduce chemical concentrations and fluxes before groundwater discharge. This biogeochemical reduction effect is discussed by Boatman and Hotchkiss (1977) for the Port of Seattle's Terminal 91 project where a granular berm surrounds dredged contaminated sediments. For arsenic (the only chemical common to the Bremerton Naval Complex COCs), they estimated a reduction of 100 percent in groundwater concentrations due to biogeochemical reactions within the interior half of the berm, more than 30 feet inland from the point of groundwater discharge to the bay. The Terminal 91 experience would relate to a Ps value of zero for arsenic.

It is unlikely that chemical concentrations in OU B groundwater are diminishing in ways comparable to those at Terminal 91, based on the data analysis presented in Appendix UU. The analysis indicates that chemical concentrations in site groundwater do not decrease with increasing salinity, representing greater degrees of seawater mixing.

However, to the extent that it occurs, precipitation induced by seawater would likely happen in the terrestrial seawater-mixing zone, not in the marine sediments. In addition, dissolved COCs in groundwater, already at low (trace) concentrations, would be diluted by seawater, and dilution typically reduces the potential for precipitation. Although changes in pH or temperature can

induce precipitation, significant changes in these groundwater parameters were not expected. To the extent that changes do occur, they would likely occur before discharge.

Sampling and analysis techniques used to measure dissolved COCs in groundwater samples would portray COCs attached to any unfiltered colloidal particles that are mobile in groundwater as "dissolved." COCs attached to such colloidal particles that precipitate or "salt-out" from groundwater onto soil particles would increase COC concentrations in the affected soil. Salting-out of colloidal particles that have an active electrically charged double layer (e.g., hydrophobic clay) can occur as seawater mixes with the groundwater and increases its ionic strength. However, because groundwater would have mixed with seawater before discharge, any salting-out of colloids induced by seawater would likely have occurred before discharge.

Sorption Onto Marine Sediment. The potential for future sorption of COCs onto organic carbon, iron oxides, or clay minerals in marine sediments is considered negligible for two major reasons. First, current conditions at the site are likely to have reached a dynamic equilibrium. This means that there should be no significant future increase in COC sorption in marine sediment from groundwater, particularly given that dissolved COC concentrations in groundwater will decrease in the future due to source depletion (assuming effective source control). The second reason has to do with the flow path required to affect the marine sediments.

[18]Seawater also contains sulfate and nitrate, which facilitate the biogeochemical reactions.

As discussed in Section 5.2.1.1 and Appendix YY, relatively deep flow paths would be required for chemicals in discharging groundwater to affect the upper 10 cm of marine sediments. Deeper flow would generally have significantly less chemical flux than would shallow flow because of lower chemical concentrations and longer flow paths, and therefore lower specific discharges. Also, deeper flow must travel long distances, on the order of hundreds to thousands of centimeters, through marine sediment before discharge to surface water. To the extent that chemical mass was lost to the sediment, it is very unlikely that the mass lost within the *last* 10 cm (3.9 in.) of marine sediments would be significant. Any sorption of chemicals to sediments from discharging groundwater would have occurred at depths deeper than the upper 10 cm of sediment.

This contention is supported by available sampling data for marine sediments. The data generally indicate that average COC concentrations increase with depth below the sediment surface (mud line). Although this effect is confounded by natural recovery from clean sediment deposition, this observation supports the hypothesis that to the extent COCs from site groundwater affect sediments, the effect does not occur in the upper 10 cm of marine sediments.

Fixing Onto Suspended Water Column Sediments. There is a potential for dissolved chemicals transported to surface water to become attached to suspended sediment in the water column. There would be an increase in chemical concentration in the marine sediment to the extent that such suspended sediment was deposited onto the marine sediment surface. Although this mechanism is considered insignificant for more mobile COCs such as zinc, mercury, and arsenic, it is potentially somewhat significant for less mobile COCs such as lead and copper. It is of significance for organic COCs, as discussed later.

Correlation of Ps With Affected Sediment Length. Whatever the proportion of dissolved COC mass in groundwater that comes to be in marine sediments in the biologically active top 10 cm of marine sediments, it is likely correlated with the offshore affected sediment length, symbolized Lm. Relatively lower values of Ps would be associated with shorter values of Lm, and relatively higher values of Ps would be associated with longer values of Lm. Although Ps is considered small for any credible length Lm, it probably increases with increasing distance Lm. Therefore, estimates of Ps cannot be made independently of estimates of Lm. The following probabilistic estimate of Ps assumes the estimate of Lm given in Section 5.2.4.5.

Probabilistic Estimates of Ps for Inorganic COCs. Overall, the information discussed in the preceding paragraphs argued for a value of Ps close to zero for the inorganic COCs. As a conservative interpretation of that information, the quantity Ps was estimated as follows:

- The uncertainty in Ps is lognormally distributed, which required that the true value of Ps exceed zero and have no theoretical upper bound (although there is a physical upper bound of 1.0).

- There is a 95 percent probability that the true value of Ps will not exceed one-fifth (0.20, or 20 percent) of the dissolved COC mass in discharging groundwater and a 50 percent probability that the true value of Ps will not exceed one-tenth of one-fifth (0.020, or 2 percent).

This resulted in an average or expected value for Ps of 0.053, or 5.3 percent. Note that these estimates are based on the chemical mass in groundwater estimated from COC concentrations measured in site groundwater monitoring wells.

This range of estimates was considered conservative enough to include all inorganic COCs (i.e., arsenic, copper, mercury, lead, and zinc). In a relative sense, the estimates may be most conservative for mercury and zinc, which appear to be generally more mobile than the other COCs, and least conservative for less mobile COCs such as lead and copper (e.g., Bagchi 1990).[19]

Organic Chemicals. The organic COCs consist of PCBs and PAHs, which include HPAHs and LPAHs. The PCBs and HPAHs have extremely high organic carbon–water partitioning coefficients. To the extent that there is a mass of PCBs and HPAHs in discharging groundwater, under *equilibrium conditions* the mass would partition onto organic material in the water column. It was assumed that this organic material would be associated with either depositing or freshly deposited sediments in the region of affected sediment.

Although it is unlikely that equilibrium conditions would prevail, Ps for PCBs and HPAHs was set at a deterministic value of 1.0, the highest possible value. This means that 100 percent of the mass in site groundwater would be fixed onto the affected marine sediment (i.e., over the affected sediment length Lm, as discussed in Section 5.2.3.5). Using Ps=1.0 is a conservative assumption. Because Ps depends on chemical kinetics and mixing effects and is not solely due to equilibrium partitioning (which assumes perfect mixing and adequate reaction time to reach the limiting condition of equilibrium), absent the limiting conditions of equilibrium, the true value of Ps is less than 1.0, the theoretical maximum.

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[19]The relative mobility and tendency to remain dissolved (not fix onto sediments) is considered to be, from highest to lowest: mercury, zinc, arsenic, copper, lead.

Although LPAHs do not have the same affinity for organic material as do PCBs and HPAHs, the affinity is still high. Therefore, the same assumption was made for LPAHs, and Ps was set to 1.0.

To reiterate, Ps=1.0 is the highest possible value, giving minimum (most conservative) times. It may be useful to note that estimates for any other Ps (Ps′, where Ps′<1) can be made by multiplying the concentration corresponding to a given time (held constant) by the ratio Ps/Ps′ (i.e., the reciprocal of Ps′, since Ps=1.0). For example, for any given time estimate the corresponding "allowable" concentration would be doubled for Ps′=½, tripled for Ps′=1/3, and so on. Note that the same linear relationship does not hold for the time associated with a given concentration that is held constant (e.g., holding concentration constant, the corresponding time estimate is not doubled for Ps′=½). For time, the relationship is nonlinear.

5.2.4.5 Affected Sediment Length

The offshore distance over which precipitation and sorption may occur is measured by the affected sediment length, Lm. Although Lm is uncertain, it is probably positively correlated with both parameter Ps (as discussed in Section 5.2.4.4) and Aq.

Because marine sediments are evaluated in terms of area-weighted averages, it could be argued that it is appropriate to consider an area encompassing the entire exclusion zone when evaluating the potential influence of COCs in groundwater discharge. If so, a distance of 1,000 ft would be a conservative approximation of the sediment area most affected by historical Bremerton Naval Complex operations, including potential effects of groundwater discharge. A 1,000- to 1,500-ft affected sediment length is reasonably consistent with the observed pattern of sediment contamination, as supported by statistical analysis of available COC data for marine sediments. The statistical analyses quantify that, with statistical certainty, the average COC concentrations in the exclusion zone are higher than those outside the exclusion zone.

It could also be argued that most of any precipitation or sorption would occur relatively near shore. Although available marine sediment data indicate a possible general tendency toward marginally higher chemical concentrations near shore, the spatial patterns are fuzzy, discontinuous, and not well supported statistically.[20] Spatial patterns are also confounded because sources of existing chemicals in marine sediments include both effects of historical groundwater discharge and direct releases from piers and ships, as well as effects of sediment erosion and deposition. (It is also noted that results of groundwater modeling done for OU A

[20]Beyond about 100 ft from shore, there is a general correlation of gradually decreasing average chemical concentration with distance from shoreline, particularly beyond about 1,000 to 1,500 ft.

[URS 1996a] are not applicable because the modeling did not include effects of sorption or precipitation.)

Probabilistic Estimate of Affected Sediment Length. A probabilistic estimate of distance Lm was made to deal with the lack of definitive data and resulting uncertainty. The bathymetric data used to estimate a practical lower value for Aq (i.e., 20 ft) was used to make a corresponding lower value estimate of 100 ft for Lm. A practical upper estimate was conservatively equated to 1,000 ft based on the observed elevated concentrations in the exclusion zone.

The affected sediment distance Lm was characterized as follows:

- The uncertainty in Lm is lognormally distributed.

- There is a 50 percent probability that the length will (or will not) exceed 100 ft and a 95 percent probability that the length will *not* exceed 1,000 ft. The identical distribution resulted if the lower estimate of Lm was set at 10 ft with a 95 percent probability of exceedance.

This gave an average or expected value for Lm of 266 ft. These estimates for Lm were also considered consistent with the estimates of Ps.

5.2.4.6 Inlet Mixing Volume Flushing Time

Inlet mixing volume (IMV) flushing time, tf, is a model-defined parameter that represents the time required for dissolved COCs in discharging groundwater to mix with inlet surface water and flush from a hypothetical IMV. Although no empirical data are available to estimate this time, it must be very short because the nearshore environment is dynamic, turbulent, and well mixed by advection and dispersion from tidal, current, and other hydrodynamic effects.

The independent but complementary analyses documented in Appendices UU and VV were used to help quantify parameter tf:

- Appendix UU develops a rigorous analysis of groundwater concentrations and fluxes resulting from tidal mixing. The analysis indicates that concentrations in groundwater would be at ambient seawater concentrations at the point of groundwater discharge.

- Appendix VV develops a rigorous analysis of chemical concentrations in surface water at the point of groundwater discharge. The analysis indicates that, relative to groundwater concentrations, the chemical concentrations in the receiving surface water would drop by two or more orders of magnitude at the groundwater discharge point. Because the Appendix VV analysis is based on chemical flux in discharging groundwater, results are independent of the groundwater concentration at the discharge point.

Although the results from the analyses in Appendices UU and VV are considered to be reliable representations of chemical concentrations in inlet surface water at the discharge point, the regulatory agencies requested that the use of these analyses be limited to interpreting a flushing time for the hypothetical IMV.

Based on interpretation of the Appendices UU and VV analyses, it was concluded that

- An equivalent IMV flushing time would be zero
- Any actual IMV is of zero length

In interpreting estimates of parameter tf, it should also be noted that, by definition, the IMV concept precludes seawater mixing with groundwater before discharge, which ignores the potential for groundwater concentrations measured in monitoring wells to be reduced before discharge. By definition, the IMV forces a non-zero mixing length. It is considered more probable than not that mixing *before* discharge is consistent with a tf of zero.

Probabilistic Estimate of IMV Flushing Time. Although the IMV flushing time is considered to be zero, the following values were used in the analysis. Parameter tf is lognormally distributed, which requires that tf exceed zero and have no theoretical upper bound. The analyses used a 95 percent probability that the value of tf is 0.1 day or less and a 50 percent probability that it is 0.01 day or less. This resulted in an average or expected value for tf of 0.03 day (43 minutes). This estimate is considered implausibly high.

5.2.4.7 Sedimentation Rate

Sedimentation rate, Sr, affects estimates of the duration of time before chemicals in marine sediments exceed SQS levels, tc. A high Sr yields a high tc, and vice versa. Parameter Sr is equal to the average net sedimentation rate, which is the difference between the total sedimentation rate and the total resuspension rate, and is a measure of the long-term rate of accumulation of sediments at a specific location.

In terms of the marine sediment time duration analysis, parameter Sr represents a spatial and temporal average. Sr is spatially averaged over the affected sediment area and temporally averaged over the time duration, which is on the order of centuries or longer.

The average of the individual sedimentation rates recorded during lead-210 dating at Stations 475 and 499 indicated a net sedimentation rate of approximately 0.30 in./yr (URS 1996b, Appendix RR, Section 5.5) in the central portion of the inlet, where the lead-210 data were collected. The sedimentation rate used to estimate natural recovery of PCBs in marine sediments in the exclusion zone, and throughout the inlet, in support of the OU B Marine ROD (U.S. Navy et al. 2000) used the point value of 0.30 in./yr, based on the lead-210 measurements.

For this analysis, the 0.30 in./yr rate was very conservatively used as a practical upper bound on the average sedimentation rate in the Bremerton Naval Complex exclusion zone. Other information considered in estimating the sedimentation rate included the following:

- A net positive sedimentation rate is supported by the statistical analysis of available COC data for marine sediments. Those data indicate that natural recovery is occurring, which implies a net deposition of relatively clean sediment.
- The sediment trend analysis (McLaren 1998) concluded that the interpier area was either depositional or mixed depositional/erosional in character and that a portion of the easternmost shipyard may be an erosional environment.
- Prop wash does not influence the average net sedimentation rate or the average concentration of any chemical over the exclusion zone.
- The equations that model the influence of groundwater discharge on chemical concentrations in sediments under conditions of sedimentation remain fundamentally unchanged under conditions of erosion (which represents a negative sedimentation rate). Localized erosion can thus be as effective as deposition for controlling sediment recontamination from groundwater discharge.

Probabilistic Estimate of Sedimentation Rate. Considering the limited data available, the following assumptions were made to conservatively reflect the uncertainty in Sr:

- The uncertainty in Sr is a lognormal PDF for Sr greater than zero.
- There is 95 percent probability that the true Sr will not exceed the available point estimate of 0.30 in./yr, and there is 50 percent probability that the true Sr will not exceed one-tenth of the 0.30 in./yr estimate, or 0.03 in./yr.

This resulted in an average or expected value for Sr of 0.08 in./yr, or about one-quarter of the 0.30 in./yr estimate based on the lead-210 data.[21]

Implicit in the probabilistic estimate of average sedimentation rate for the exclusion zone is the potential influence of localized zones of net sediment erosion, as well as localized areas of zero sedimentation. As an ultra-extreme test of the sensitivity of the time estimates, time duration estimates assuming an Sr of zero were also made for the inorganic COCs. Because of the parametric approach used for the organic COCs, estimates for Sr=0 were not made for the organic COCs.

5.2.4.8 Reduction Factor for Drydock Dewatering

Reduction factor, R, measures the groundwater flow reduction to the inlet because of quay walls, interception by drydock dewatering, or both. R can range from 1.0 for no reduction to 0.0 for complete reduction; R is negative, less than zero, for net flow from the inlet into the terrestrial areas.

Although the reduction factor represents the time-averaged ratio of hydraulic gradient with quay walls and drydock dewatering to the hydraulic gradient in the absence of these two effects, drydock dewatering is much more influential than quay walls. The reduction factor is therefore primarily a measure of drydock dewatering.

Reduction factors are used only for flux, load, and mixing length estimates. R is not used to estimate times for marine sediments to reach SQS values (i.e., R=1.0), as discussed in Section 5.2.2.1. Also, because of the parametric approach used for the organic COCs, reduction factors were not used for any of the organic COC estimates (i.e., R=1.0).

Where used, R values were considered for the three potential groundwater areas of concern at OU B: Sites 10 West, 1, and 8 (demolished Building 106 underground tanks area). At Site 10 West, the effect of drydock dewatering was assumed to be negligible, and although there was evidence that a quay wall extends into Site 10 West, a reduction factor was conservatively set to a point value of 1.0. That is, for Site 10 West no reduction was assumed for drydock dewatering or a quay wall.

At the other extreme, Site 8 (106 tanks area) was considered to have an R less than or near zero. First, groundwater discharge from the Site 8 area is at least partly restricted by sheet-pile quay

[21]The sedimentation rate used to estimate natural recovery of PCBs in marine sediments in the exclusion zone, and throughout the inlet, in support of the OU B Marine ROD was 0.30 in./yr, based on the lead-210 measurements. The average sedimentation rate used here for the marine protectiveness measures is therefore about one-quarter of that used in the natural recovery analyses that support the OU B Marine ROD.

walls in the eastern area of OU B. Second, the 106 tanks were closed in 1994 and filled with grout. In addition, Site 8 lies between Drydocks 1 and 3, and the drydock dewatering process further restricts groundwater discharge to the inlet.

Simulated groundwater movement in the Site 8 area, as described in the draft RI (URS 1996b, Appendix RR, Section 5), indicates that R is less than zero. That is, the simulations indicated that net subsurface flow is *inland* from the inlet, that all groundwater flow is to Drydock 3, and that chemical discharge to the inlet does not occur. Drainage from Site 8 to Drydock 3 was estimated during the SI to be 15 gpm (U.S. Navy 1999d). The pressure pipe historically reported to discharge oily water from Site 8 to Drydock 3 has since been rerouted to the sanitary sewer (U.S. Navy 1999d).

Similar to the R value for Site 8, the R value for Site 1 was considered to be less than or near zero. Based on the groundwater modeling results described in the draft RI (URS 1996b, Section 5), Site 1 straddles a groundwater divide where groundwater in the Site 1 area flows to Drydock 6 to the west and Drydock 5 to the east. Also, the seawater-level salinity in groundwater samples in the Site 1 area indicates that the majority of groundwater flow is into the Site 1 area from the inlet. Based on these factors, the uncertainty in R for Site 1 was conservatively assumed to be the same as for Site 8, as presented in the following.

Estimate for R: Sites 1 and 8 With Drydock Dewatering. The factors discussed in the preceding paragraphs argued for an R of zero or less for the Site 1 and Site 8 areas. However, as a conservative interpretation for these areas, R was set to a point value of 0.02. A less conservative, more realistic estimate of R would include negative values to account for the inward flow of seawater due to drydock dewatering. For Sites 1 and 8 an expected value for R of zero appears to be realistic.

The value of 0.02 can be contrasted to the “bulkhead factor” for sheet-pile quay walls used by the USGS to model groundwater at PSNS (USGS 1997). Functionally, the bulkhead factor is similar to an R factor. However, the USGS point estimate of 0.02 for the bulkhead factor did not include effects of drydock dewatering, which would serve to reduce the factor below that due to bulkheads alone.

No Drydock Dewatering. For the case of no drydock dewatering, R=1.0 was used for all three sites. The case of no drydock dewatering assumes no effects of dewatering and represents groundwater conditions associated with the permanent closure of the site and cessation of drydock operations. Use of R=1.0 is conservative because it neglects the reduction effects of bulkheads and quay walls (for which USGS assigned the equivalent of R=0.02).

5.2.4.9 Sediment Density

Average sediment dry density, parameter Sd (expressed as dry unit weight [lb/ft^3]), affects estimates of the elapsed time before chemicals in sediment exceed SQS, tc, with a high (low) Sd causing a high (low) tc. A dry density for sediments near the mud line at OU B has been estimated at 35 lb/ft^3 (URS 1996b, Appendix II), indicative of sediments of very low density. Although sediment density is likely to increase with depth below the mud line, it was assumed that near-mud-line sediment density is not very variable.

Probabilistic Estimate of Sediment Density. To reflect the uncertainty in the estimate of Sd, the following assumptions were made:

- The uncertainty in Sd is a lognormal PDF.
- There is a 95 percent probability that the true Sd will not exceed the available point estimate of 35 lb/ft^3 by more than 50 percent (i.e., 53 lb/ft^3).
- There is a 95 percent probability that the true Sd will not be less than 80 percent of the 35 lb/ft^3 point estimate (i.e., 28 lb/ft^3).

These assumptions resulted in an average or expected value for Sd of 39 lb/ft^3 in the top 10 cm of marine sediment.

5.2.4.10 Dissolved Concentration of COCs

As discussed in Section 5.2.1, the dissolved concentration of inorganic COCs in discharging groundwater used in the analyses was based on the *maximum* dissolved COC concentration in samples measured during the RI in nearshore monitoring wells. Based on discussions with regulatory agency staff, this was used as an expedient and simple way to deal with uncertainty in dissolved COC concentrations.

Using the maximum measured concentrations was considered conservative because chemical mass fluxes should be estimated using spatially averaged concentrations over the area of concern. The use of maximum concentrations overestimates the flux contribution from areas having concentrations less than maximum. However, where all points have not been sampled, the true spatially averaged concentration is uncertain. Therefore, estimating the true (but uncertain) spatially averaged concentration by using the maximum measured concentration provides confidence that the true average has not been underestimated. Using the maximum measured COC concentrations provided an inherently high level of conservatism in the analyses that did not require an assignment of additional uncertainty. Resultant fluxes, and the protectiveness measures based on these fluxes, were considered to represent maximum values for the areas of concern.

Additional conservatism occurs because historically measured maximum chemical concentrations are expected to decrease over time, given effective source control. Future fluxes are overestimated to the extent future concentrations decrease relative to the historically measured concentrations. Consequently, the estimated times to increase sediment chemical concentrations by SQS levels are overestimated.

For all site areas, the maximum concentration for dissolved mercury was set at ½ its detection limit because dissolved mercury was not detected in any of the 171 groundwater samples analyzed in the RI. In two cases (lead at Site 1 and zinc at Site 8) where no dissolved concentrations have been reported, the highest total inorganic result was used to set upper limits on potential dissolved concentrations. Using total concentrations adds another level of conservatism because total concentrations are very likely an artifact of sampling and result in an overestimate of concentrations susceptible to groundwater transport. Because the 106 tanks were filled with grout during the 1994 closure, data from postclosure sampling round II were used for Site 8.

As discussed in Section 5.2.2.1, “dissolved” analyses are unlikely to underestimate the concentration of colloids that could flow in groundwater. Because dissolved analyses use a 0.45-micron filter, which is at the very upper end of the colloid range,[22] filtering would not remove either colloids or suspended particles less than 0.45 microns. To the extent filtering passes particles that do not flow with site groundwater, chemical concentrations from dissolved analyses will exceed the true chemical concentrations that are dissolved in or otherwise flow with site groundwater. Nevertheless, in practical terms, filtering provides the most reasonable measurement of inorganic concentrations that are being transported in site groundwater.

Total concentrations, in contrast, are very likely to overestimate concentrations susceptible to groundwater transport. Only total analyses were available for PAHs and PCBs because filtered samples were not collected for organic chemicals. The vast majority of these analyses detected no measurable concentrations of these chemicals. The few cases of detected concentrations are certainly biased high from effects of suspended particles (from imperfect sampling) that do not flow with site groundwater. Chemical concentrations from total analyses generally exceed the true chemical concentrations that are dissolved in or otherwise flow with site groundwater.

[22]For example, E.M. Thurman, *Organic Geochemistry of Natural Waters*, 1985, cited in Lyman 1992.

5.2.4.11 Sediment Thickness

Sediment thickness or depth below the mud line (St) affects only the estimated duration of time before chemicals in sediment exceed SQS levels. No uncertainty was assigned to sediment thickness. The sediment thickness was held constant over all time at the target thickness of 10 cm, consistent with the SMS ($St = St_o = 10$ cm).

5.2.4.12 Shoreline Lengths

Appropriate lengths of shoreline (Ls) were estimated to calculate total chemical loads from discharging groundwater. Because chemical loads are useful for comparing groundwater mass loadings to stormwater mass loadings, shoreline lengths were made consistent with the western, central, and eastern stormwater discharge zones. These zones were then associated with the locations of Sites 10 West (western), Site 1 (central), and Site 8 (eastern). The following assumptions were made:

- The western section extends from the western edge of OU NSC (FISC) to the eastern edge of OU A (Mooring G).
- The central section extends from the eastern edge of OU NSC to the western edge of Drydock 4.
- The eastern section extends east from the central section to the eastern boundary of the Bremerton Naval Complex.

Site 10 West was associated with the western section, Site 1 with the central section, and Site 8 with the eastern section.

The shoreline lengths were expected to significantly overestimate the true lengths of shoreline associated with the maximum concentrations of dissolved chemicals measured in groundwater, as used in the analyses. Groundwater concentrations spatially averaged along the assumed shoreline lengths would be the technically appropriate concentrations for this analysis. Although these spatially averaged concentrations are uncertain, they are very likely less than the concentrations used in the analyses, as discussed previously.

Using the maximum shoreline lengths with the maximum measured groundwater concentrations provided inherently high estimates of chemical loads from discharging groundwater. These estimates were expected to exceed 95 percent nonexceedance estimates and thus have greater than 95 percent probability of exceeding the true chemical loads to the inlet from groundwater discharge.

5.2.5 Results of Marine Protectiveness Analyses

Results of the marine protectiveness analyses are presented in a series of tables and figures. Estimates are based on the input parameter values in Table 5-2. The basis of the input values is discussed in Section 5.2.4. Results are presented separately for the inorganic COCs and the organic COCs.

5.2.5.1 Inorganic Chemicals

Tables 5-3, 5-4, and 5-5 present the marine protectiveness estimates for the five inorganic COCs. In each table, estimates are presented for the three sites:

- Site 10 West (west)
- Site 1 (central)
- Site 8 (east)

Estimates are based on maximum dissolved COC concentrations measured in site groundwater monitoring wells, with the following exceptions. Total lead was used for Site 1; total zinc was used for Site 8; and because there were no detections of dissolved mercury at any of the sites, mercury concentrations were set at ½ the detection limit.

Fluxes and Loads. Table 5-3 presents estimates of chemical mass fluxes and total chemical loads. Results are presented for two conditions: without drydock dewatering (R=1.0 for all three sites) and with drydock dewatering (R=0.02 for Sites 1 and 8).

Unit Fluxes. The unit flux estimates include average or expected values and 95 percent nonexceedance values. The 95 percent nonexceedance values have a 95 percent probability of overestimating the true flux. Equivalently, there is a 95 percent probability that the true fluxes are less than the estimates in Table 5-3.

The unit fluxes presented in Table 5-3 are considered *site maximums* because they were estimated for the areas of concern in Sites 10 West, 1, and 8 using maximum measured COC concentrations, as discussed previously. These maximum unit fluxes effectively represent hot spot conditions for the respective site areas.

Average or expected maximum fluxes without drydock dewatering (R=1.0) ranged from 23 g/yr/ft for zinc at Site 1 to 0.01 g/yr/ft for mercury at all sites (although there were no detections of dissolved mercury at any of the sites). Corresponding 95 percent nonexceedance estimates ranged from 84 to 0.03 g/yr/ft.

Maximum Loads. The maximum chemical load estimates presented in Table 5-3 are the product of the average maximum unit flux and the maximum shoreline length for the western, central, and eastern areas of OU B. Although probabilities were not formally calculated, these maximum load estimates are expected to exceed the true loads with greater than 95 percent probability, as discussed in Section 5.2.4.12.

Because the maximum load estimates in Table 5-3 are expected to overestimate the true loads with practical certainty, there are no entries for average or expected values for maximum loads. The maximum loads are effectively 95 percent nonexceedance estimates.

For each COC, the sum of the maximum load for the three site areas is also presented in Table 5-3. For the case of no drydock dewatering (R=1.0), these maximum sums range from 126 kg/yr for zinc to 0.07 kg/yr for mercury (although dissolved mercury was not detected at any of the sites). The maximum total loads in Table 5-3 are compared with estimated stormwater chemical loads in Section 5.3.

Mixing Lengths. Table 5-4 presents estimates of the mixing length required to achieve the marine SWS for Sites 10 West, 1, and 8. Results are presented for conditions with and without drydock dewatering. Except for copper, the length estimates include ambient seawater concentrations, as measured in Sinclair Inlet. Lengths for copper assume no ambient seawater concentration because the measured inlet value of 5.5 μg/L is already above the SWS of 3.1 μg/L. The length estimates are based on the maximum unit fluxes presented in Table 5-3.

The Table 5-4 estimates are inherently overestimates. All mixing lengths presented in Table 5-4 are based on the Surface Water Mixing Length Model. This mixing model *requires* that mixing lengths be greater than zero. As stated in Section 5.2.2.2, the mixing model also ignores seawater mixing with groundwater immediately before discharge and thus precludes the potential for groundwater concentrations measured in monitoring wells to diminish before discharge to the inlet. The more accurate analyses in Appendices UU and VV demonstrate that the mixing model overestimates surface water concentrations at the groundwater discharge point. This means the estimated mixing lengths in Table 5-4, as summarized next, are too long and the nonexceedance probabilities too low.

Surface Water Mixing Lengths. Table 5-4 includes average or expected values and 95 percent nonexceedance estimates. The 95 percent nonexceedance estimates have a nominal 95 percent probability of not being exceeded by the true mixing length. A more accurate probability is 100 percent, based on the preceding discussion (last paragraph).

Under the condition of no drydock dewatering (R=1.0), the expected values of the mixing lengths range from a maximum of 0.16 ft for copper (incremental length) at Site 1 to a minimum of 0.001 ft for lead at Site 10. These lengths are certain overestimates, based on the preceding discussion (paragraph before the last).

The true mixing lengths are expected to be zero. A zero mixing length is based on the complementary analyses in Appendices UU and VV. These analyses independently predict the same conclusion: chemicals in discharging site groundwater will not cause SWS exceedances in surface water at the groundwater discharge point. This is equivalent to saying that the mixing length is zero.

Times to Reach SQS. Table 5-5 presents estimates of the time before chemicals in initially clean sediment at Sites 10 West, 1, and 8 will exceed SQS due to chemicals transported by groundwater. Time estimates assume no drydock dewatering (R=1.0), initially clean sediments, and no effect of source depletion.

The time estimates are based on the maximum unit fluxes presented in Table 5-3. The time estimates are therefore representative of hot spot conditions, not average or typical conditions in the western, central, and eastern portions of the site.

Time to Reach SQS Concentrations in Marine Sediments. The time estimates include average or expected values and 95 percent exceedance estimates. The 95 percent exceedance values have an estimated 95 percent probability *of being exceeded* by the true (but inherently uncertain) duration of time. Equivalently, there is a 95 percent probability that the true times will equal or exceed the estimates in Table 5-5. The estimates are the following for all chemicals and site areas:

- The expected time is infinite.

- There is a minimum 95 percent probability that the true times will exceed 10,000 years for all site areas and COCs with the exception of zinc at Sites 1 and 8. The estimated times for zinc exceed 5,000 years at Site 8 and 1,000 years at Site 1 (the MS 97 Excel spreadsheet used to implement the analyses displays times only in increments—currently, >10,000 years, >5,000 years, >1,000 years, and so on).

Infinite Time Probability. Estimates of the probability that the times are infinite are also presented in Table 5-5. These infinite time probabilities exceed 90 percent for all chemicals except zinc at Site 8, which has an infinite time probability of 85 percent. An infinite time means that chemical mass from discharging site groundwater will never increase chemical concentrations in marine sediments by an increment equal to the SQS, independent of sediment depth. For initially clean sediment, this means that the SQS would never be exceeded. Thus even for zinc at Site 8, there is an 85 percent probability that clean sediment subjected to groundwater discharge would never exceed the SQS.

Zero Sedimentation Rate. Time estimates for the unrealistic case of a long-term average sedimentation rate of zero (Sr=0) are also included in Table 5-5. These are median estimates conditional on Sr=0. All estimates exceed 3,400 years. For the time estimates to be consistent with the Sr=0 assumption, a condition of zero sedimentation (with neither erosion nor sedimentation) would have to prevail for the estimated times—that is, in excess of 3,400 years.

5.2.5.2 Organic Chemicals

Figures 5-2 through 5-8 present the marine protectiveness estimates for the two organic COCs, PCBs, and PAHs. PAHs are separated into LPAHs and HPAHs. Data for PCBs and PAHs in site groundwater do not allow site distinctions. Therefore, all of OU B was analyzed as one site; equivalently, all sites within OU B are treated the same. Because of these data limitations, the protectiveness estimates are also presented as parametric functions of chemical concentrations in site groundwater, as explained next.

Parametric Presentation for PCBs and PAHs. Protectiveness measure estimates in Figures 5-2 through 5-8 are presented as functions of chemical concentrations in site groundwater. The parametric presentation approach simply means that the protectiveness measures (vertical axes) are graphed against groundwater concentration (horizontal axes) taken as an independent parameter. The parametric presentation approach (with groundwater concentration as an independent variable) has the advantage of allowing readers to draw their own conclusions as part of interpreting the results of the analyses. The parametric approach is used because representative concentrations of PCBs and PAHs in site groundwater are too low to be detected; that is

- The vast majority of OU B groundwater samples showed no detectable concentration of organic COCs: respectively, 97, 91, and 84 percent of the samples had no detectable concentrations of PCB, HPAH, and LPAH.

- The limited detections are unrepresentative of true site conditions—they are both too few and infrequent and may be artifacts of imperfect sampling because only unfiltered samples were analyzed for organic compounds.

- PCBs and HPAHs have very low solubility and corresponding high affinity for soils, which results in the potential for very low concentrations in site groundwater.

These considerations argued for using ½ the minimum detection limit as the *maximum* potential concentration in OU B groundwater. Therefore, a concentration of ½ the minimum detection limit was used to draw the principal conclusions from the figures for PCBs and HPAHs (the minimum detection limit was used for LPAHs). Nevertheless, the full range of chemical detection limits and detections was used for the parametric graphs, as discussed next.

Range of Chemical Detection Limits and Detections. The parametric presentation graphs the *range of chemical detection limits and detections* on the graphs (Figures 5-2 through 5-8) of protectiveness measure versus site groundwater concentration. Although ½ the minimum sample detection limit is considered the most accurate representation of an upper bound on site concentrations, graphing the entire range of detection limits and detections allows readers to make their own interpretations of site concentrations and related protectiveness estimates. The graphed range of chemical detection limits and detections includes the following:

- Number of sample measurements

- Number of nondetections (NDs) and number of detections

- Minimum detection limit (DL) and the number of NDs having the minimum DL

- Concentration at ½ the minimum DL (DL/2)

- Concentration of all detections

- Geomean concentration of all samples, both NDs and detections, with NDs set at ½ DL

Interpreting the Range of Chemical Detection Limits and Detections. To reiterate, the parametric approach is used because representative concentrations of PCBs and PAHs in site groundwater are *below* detection limits. The range of chemical detection limits and detections shown in Figures 5-2 through 5-8 are therefore

- Not representative of true chemical concentrations in site groundwater.

- Indicative of an upper bound or limit on concentrations, after discounting the limited detections that are considered probable artifacts of the total (unfiltered) analyses used to measure organic chemicals. The upper bound on site concentrations is most accurately interpreted relative to the sample detection limits, which represent the lower end of the range of detection limits and detections graphed on the figures.

The parametric approach was used for chemical fluxes, loads, mixing lengths to meet SWS, and times to reach SQS in marine sediments. Estimates for mixing lengths were limited to PCBs because SWS are not available for PAHs.

Fluxes and Loads. Figures 5-2 through 5-4 graph estimated chemical flux and load to the inlet parametrically against the chemical concentration in site groundwater. Figure 5-2 is devoted to PCBs, Figure 5-3 to HPAHs, and Figure 5-4 to LPAHs. In each figure the estimates are presented in the same way. Each figure includes two horizontal (concentration axes) and two vertical (protectiveness measure) axes:

- Lower horizontal (concentration) axis—The chemical concentration in site groundwater is represented on the lower horizontal axis as an independent (parametric) variable; that is, the lower concentration axis says nothing about what the true site groundwater concentration actually *is* (the concentration is uncertain, as already explained). Concentration units are microgram of chemical per liter of discharging groundwater (µg/L).

- Upper horizontal (concentration) axis—The range of detection limits and detections is graphed immediately below the upper horizontal concentration axis (the upper and lower horizontal concentration axes are otherwise identical). The range of detection limits and detections represents the available data for *interpreting* the true but uncertain site groundwater concentration.

- Right vertical (flux) axis—Chemical fluxes are measured by the right vertical axis as a (parametric) function of the chemical concentration on the lower horizontal axis *or* for a given concentration within the range of detection limits and detections graphed immediately below the upper horizontal axis. Flux units are microgram of chemical per day per foot of shoreline (µg/day/ft).

- Left vertical (load) axis—Chemical loads are measured by the left vertical axis as a (parametric) function of the chemical concentration on the lower horizontal axis *or* for a given concentration within the range of detection limits and detections graphed immediately below the upper horizontal axis. Loads are based on 8,850 ft of OU B shoreline; units are kilogram of chemical per year (kg/yr).

Nonexceedance Estimates. Nonexceedance estimates of chemical fluxes and loads are presented in Figures 5-2 through 5-4 as a (parametric) function of site groundwater concentration. The estimates are made at three nonexceedance probability levels that cover a reliability range from "high reliability" (i.e., 95 percent) to "more probable than not" (i.e., very slightly greater than 50 percent),[23] explained as follows.

- 95 percent nonexceedance estimates—These have a 95 percent probability of overestimating the true flux or the true load. Equivalently, there is a 95 percent probability that the true flux or load will not exceed these estimates. In ordinary language, these may be considered high-reliability estimates because it is highly unlikely that the true flux or load would exceed the (95 percent nonexceedance) estimate.

- 75 percent nonexceedance estimates—These have a 75 percent probability of overestimating the true flux or the true load. Equivalently, there is a 75 percent probability that the true flux or load will not exceed these estimates. These may be considered moderate-reliability estimates because it is moderately unlikely that the true value would exceed the estimate.

- 50 percent nonexceedance estimates—These are median estimates, which are interpreted to have a very slightly greater than 50 percent probability of overestimating the true flux or the true load (the "very slight" difference from 50 percent is the probability increment that the true value equals the estimate). It is more probable than not (>50 percent) that the 50 percent nonexceedance estimates will overestimate the true flux or load (equal probability would require the true flux or load to be less than *or equal* to the estimate).[24]

Reading the Graphs. The parametric presentation makes the nonexceedance flux and load estimates conditional on the chemical concentrations represented on the lower horizontal axis or on any particular concentration in the range of detection limits and detections graphed below the upper horizontal axis. As stated, ½ the minimum sample detection limit, which is the lower end

[23]Nonexceedance estimates between these three values could be interpolated.
[24]The practical convenience of this usage should not be lost by torturing a semantic fine point; a 50.1 percent or 51 percent nonexceedance estimate would be equivalent to the 50 percent estimate for all practical purposes.

of the range of detection limits and detections, is considered representative of an upper bound for PCB and HPAH concentrations in site groundwater (the upper bound for LPAH concentrations is the minimum sample detection limit). In general, the flux or load associated with any particular concentration (e.g., ½ the minimum detection level or a given detected concentration) is read on the graph in the same way:

1. Project an imaginary vertical line from the concentration to the nonexceedance curve of interest.

2. From the nonexceedance curve, project an imaginary horizontal line to the right vertical axis to read off the flux estimate or to the left vertical axis to read off the load estimate.

As an instructive illustration, Figure 5-2 presents an example using a site groundwater PCB concentration equal to ½ the minimum sample detection limit of 0.005 µg/L (upper horizontal concentration axis) and the 95% nonexceedance curve. By following the above two steps, the graph shows the estimated 95% nonexceedance load for OU B is approximately 0.0125 (rounded to 0.01) kg/yr (left vertical axis) with a flux of approximately 3.9 (rounded to 4) µg/day/ft (right vertical axis). All graphs are used and read in the same basic way.

With the preceding introduction, the following discussion interprets the results of the parametric analysis for fluxes and loads.

PCBs. As discussed, ½ the minimum detection level was considered a representative upper bound of PCB concentrations in site groundwater. Based on this, the results in Figure 5-2 are interpreted to indicate the following:

- With 95 percent probability
  - PCB fluxes will not exceed approximately 4 µg/day along any foot of OU B shoreline
  - PCB total loads from OU B will not exceed approximately 0.01 kg/yr
- It is more probable than not (>50 percent probability) that
  - PCB flux will not exceed approximately 0.2 µg/day along any foot of OU B shoreline
  - PCB total loads from OU B will not exceed approximately 0.001 kg/yr

These estimates are conservative because they are based on a maximum likely site concentration.

HPAHs. Similar to the upper bound for PCBs, ½ the minimum detection level was considered a representative upper bound of HPAH concentrations in site groundwater. Based on this, the results in Figure 5-3 are interpreted to indicate the following:

- With 95 percent probability
  - HPAH fluxes will not exceed approximately 170 µg/day along any foot of OU B shoreline
  - HPAH total loads from OU B will not exceed approximately 0.6 kg/yr
- It is more probable than not (>50 percent probability) that
  - HPAH flux will not exceed approximately 7 µg/day along any foot of OU B shoreline
  - HPAH total loads from OU B will not exceed approximately 0.02 kg/yr

These estimates are conservative because they are based on a maximum likely site concentration.

LPAHs. LPAHs are more soluble than HPAHs and PCBs, and thus there is greater potential for dissolved-phase transport of LPAHs by groundwater. For this reason the minimum detection level, rather than ½ the detection level, was considered a representative upper bound of LPAH concentrations in site groundwater. Based on this, the results in Figure 5-4 are interpreted to indicate the following:

- With 95 percent probability
  - LPAH fluxes will not exceed approximately 350 µg/day along any foot of OU B shoreline
  - LPAH total loads from OU B will not exceed approximately 1.1 kg/yr

- It is more probable than not (>50 percent probability) that
  - LPAH flux will not exceed approximately 14 μg/day along any foot of OU B shoreline
  - LPAH total loads from OU B will not exceed approximately 0.05 kg/yr

These estimates are conservative because they are based on a maximum likely site concentration.

PCB Mixing Lengths. Mixing lengths for PCBs are presented in Figure 5-5. The mixing lengths are presented similarly to the flux and load estimates.

Estimates in Figure 5-5 are based on the mixing model, as used for the inorganic COCs and discussed earlier. The figure indicates that it is more probable than not that mixing lengths for PCBs would be less than approximately 0.1 ft (under the condition of no drydock dewatering). As discussed for the inorganic COCs, the true length is expected to be zero.

Estimates for mixing lengths were limited to PCBs because SWS have not been established for PAHs. However, under any credible scenario, mixing lengths for PAHs are also likely to be zero.

Times to Reach SQS. Figures 5-6 through 5-8 graph the estimated time to reach SQS concentrations in affected marine sediments against the chemical concentration in site groundwater. Figure 5-6 is devoted to PCBs, Figure 5-7 to HPAHs, and Figure 5-8 to LPAHs. For PCBs, the SQS was reduced from 12 mg/kg OC to an equivalent 3 mg/kg OC to be consistent with the marine cleanup level for OU B.

Exceedance Estimates. The time curves in Figures 5-6 through 5-8 are *exceedance* estimates, graphed as a (parametric) function of site groundwater concentration. Although it may be obvious how exceedance estimates (Figures 5-6 through 5-8) differ from nonexceedance estimates (Figures 5-2 through 5-5), to be clear, the exceedance estimates are explained as follows:

- 95 percent exceedance estimates—These have a 95 percent probability of underestimating the true time to reach SQS (or 3 mg/kg OC for PCBs). Equivalently, there is a 95 percent probability that the true time will exceed these estimates. These may be considered high-reliability estimates because it is highly likely that a true time would exceed the (95 percent exceedance) estimate. At (very) low concentrations the 95 percent exceedance estimate can be infinite; in this case, there would be a *minimum* 95 percent probability that the true time would be infinite (and, clearly, all exceedance estimates below 95 percent [e.g., 75 percent or 50 percent] would be infinite).

- 75 percent exceedance estimates—These have a 75 percent probability of underestimating the true times. Equivalently, there is a 75 percent probability that the true time will exceed these estimates. If the 75 percent exceedance estimate is infinite, there is a minimum 75 percent probability that the true time would be infinite.

- 50 percent exceedance estimates—These median estimates are interpreted as having a very slightly greater than 50 percent probability of underestimating the true time. It is more probable than not (>50 percent) that the true time will exceed these estimates (recall previous discussion regarding 50 percent nonexceedance estimates). If the 50 percent exceedance estimate is infinite, there is a minimum 50 percent probability that the true time would be infinite. And noting the obvious, if the 75 percent exceedance estimate is infinite, so is the 50 percent exceedance estimate.

Reading the Graphs. The estimates in Figures 5-6 through 5-8 are presented similarly to the fluxes and loads in Figures 5-2 through 5-4, except the curves are exceedance estimates. The time associated with any particular concentration is read off the curve by (1) projecting an imaginary vertical line from the concentration to the exceedance curve of interest and (2) projecting from the exceedance curve an imaginary horizontal line to the vertical axis to read off the time estimate. For illustration, Figure 5-6 includes an example using a site groundwater PCB concentration equal to ½ the minimum sample detection limit of 0.005 µg/L (upper horizontal concentration axis) and the 95 percent exceedance curve. Following the above two steps, the graph shows that the estimated 95 percent exceedance time for PCBs is approximately 71 years (left vertical axis). (Recall that for PCBs the SQS was reduced from 12 mg/kg OC to an equivalent 3 mg/kg OC to be consistent with the marine cleanup level for OU B.)

The vertical time axes have been limited to a maximum of 1,000 years, which is considered a reasonable limit of most practical interest. This limitation makes the graphs more generally readable, but means exceedance times greater than 1,000 years cannot be read from the graphs (although they could be roughly inferred by visual extrapolation). This limitation is remedied for the particular PCB and PAH concentrations considered to be reasonable maximums for OU B (i.e., ½ the minimum sample detection limit for PCBs and HPAHs and the minimum sample detection limit for LPAHs). For these concentrations a text box is included in each figure showing the 95%, 75%, and 50% exceedance times and the probability that the time is infinite (75 percent, or 50 percent exceedance times may be infinite). These times may exceed 1,000 years and in certain cases are infinite.

Parameter Ps. All time estimates in Figures 5-6 through 5-8 assume Ps=1.0. Parameter Ps is the proportion of chemical mass in groundwater that fixes onto marine sediment, as discussed in Section 5.2.4.4; Ps=1.0 is the highest possible value, giving minimum (most conservative) times. It may be useful to note that estimates for any other Ps (Ps′, where Ps′<1) can be made by multiplying the concentration corresponding to a given time (held constant) by the ratio Ps/Ps′ (i.e., the reciprocal of Ps′, since Ps=1.0). For example, for any given time estimate the corresponding "allowable" concentration would be doubled for Ps′=½, tripled for Ps′=1/3, and so on. Note that the same linear relationship does not generally hold for the time associated with a given concentration that is held constant (e.g., holding concentration constant, the corresponding time estimate is not generally doubled for Ps′=½).

With the preceding introduction, the following discussion interprets the results of the parametric analysis for the time to reach SQS in marine sediments due to site groundwater discharge.

PCBs. Based on ½ the minimum detection level as a maximum PCB concentration in site groundwater, the results presented in Figure 5-6 indicate the following:

- Time will exceed approximately 71 years with 95 percent probability.
- Time will exceed approximately 4,000 years with 75 percent probability.
- There is a 66 percent probability the time will be infinite.

These estimates are considered reliable because they are based on a maximum likely site concentration, as well as the various other conservative assumptions documented in the analysis.

HPAHs. Based on ½ the minimum detection level as a maximum HPAH concentration in site groundwater, the results presented in Figure 5-7 indicate the following:

- Time will exceed approximately 3,000 years with 95 percent probability.
- There is an 87 percent probability the time will be infinite.

These estimates are considered reliable because they are based on a maximum likely site concentration, as well as the various other conservative assumptions documented in the analysis.

LPAHs. Based on the minimum detection level as a maximum LPAH concentration in site groundwater, the results presented in Figure 5-8 indicate the following:

- Time will exceed approximately 124 years with 95 percent probability.
- Time will exceed approximately 14,000 years with 75 percent probability.
- There is a 70 percent probability the time will be infinite.

These estimates are considered reliable because they are based on a maximum likely site concentration, as well as the various other conservative assumptions documented in the analysis.

5.2.5.3 Conclusions From the Marine Protectiveness Analyses

The results of the marine protectiveness analyses indicate very limited potentially adverse future impacts on marine waters and sediment of Sinclair Inlet from chemicals in groundwater that discharges from terrestrial OU B. In particular, site groundwater discharges are not predicted to result in either SWS exceedances in surface water or SQS exceedances in marine sediment.

These results can be interpreted to apply to all chemicals detected at OU B because the analyses were based on representative chemicals frequently detected at OU B at concentrations exceeding regulatory criteria and adopted as appropriate chemical indicators through consultation with agency staff. The results of these analyses are considered conservative and reliable, as discussed throughout this section.

The analyses support the conclusion that site groundwater is adequately protective of the inlet and that active remediation of groundwater may not be warranted. In particular, site groundwater is predicted by the analyses to be protective of the site remedy for marine OU B.

To help verify these predictions and conclusions by demonstrating protection of surface water, it is assumed that an appropriate groundwater compliance monitoring program (consistent with WAC 173-340-720(9)) will be enacted as part of the remedy selected in the ROD. It is also assumed that a conditional point of compliance for site groundwater under WAC 173-340-720(8)(c) will be required. Presumably, the conditional point of compliance would be located immediately upgradient of groundwater discharge to the inlet because the protectiveness analyses predict that site groundwater does not exceed SWS (WAC 173-201A) at the discharge point and does not result in exceedances of SQS (WAC 173-204).

By implication, as discussed in Section 5.2.1, the terrestrial soil to groundwater pathway does not pose a threat or potential threat to the inlet. Because site soils are the source of chemicals in site groundwater, if groundwater is not a threat, then neither are the site soils. This conclusion does not extend to affected site soils that may be directly transported to the inlet by localized surface water erosion, including stormwater loading, as discussed in the next section.

5.3 SURFACE WATER PATHWAY

This section summarizes mass loadings of COCs from terrestrial OU B to Sinclair Inlet via the surface water pathway. The surface water pathway is separated into "stormwater loading," as discussed in Section 5.3.1, and "drydock discharges," as discussed in Section 5.3.2. Conclusions related to the magnitude of mass loading from groundwater relative to surface water discharges are presented in Section 5.3.3.

5.3.1 Stormwater Pathway

Surface water flowing across the site surface toward storm drains can pick up chemicals both in dissolved form and in association with particulate matter. Surface water that encounters debris accumulated within the storm drain system can also dissolve chemicals and suspend particles with sorbed chemicals from the debris. Potentially, contaminated soils and groundwater also may enter the surface water collection system through breaks or leaks in the system. Stormwater containing dissolved and suspended chemicals flows through the storm drainage system and discharges through permitted outfalls to Sinclair Inlet. The stormwater pathway analysis is discussed in more detail in Appendix RR and Appendix FF.

Stormwater Mass Loading. The principal quantitative results from the surface water analysis in Appendix RR are summarized in Table 5-6, where stormwater loading for the five inorganic COCs are estimated for the western, central, and eastern sections of OU B; the results in Table 5-6 allow comparison with groundwater loadings. The estimated stormwater loadings were based on available data, which is limited to 1995 mass loadings to Sinclair Inlet from surface water discharges from western, central, and eastern OU B. OU A and OU NSC are not included because these sites are not part of OU B and have undergone remedial action under their own RODs. Table 5-6 load estimates do not include loads from drydock discharges or groundwater entering the drydocks via the drydock pressure-relief drainage systems. Inclusion of drydock discharges (which are regulated under the NPDES program, not under CERCLA)[25] would increase the load estimates in Table 5-6 (and further support the conclusion of the analysis, as summarized in Section 5.3.3).

It should be recognized that the load estimates in Table 5-6 are based on extrapolation of limited measurements made at only a fraction of the existing stormwater discharge points during a single year (1995). Although the estimates may therefore overestimate or underestimate the true loads

[25] It is also understood that it would be impracticable to collect and treat either relief drainage water (including groundwater) entering the drydocks or drydock discharges.

under current or future conditions, the following comparison and conclusion appears to be generally valid.

Comparison of Groundwater and Stormwater Mass Loadings. Table 5-7 presents a comparison of estimated groundwater and stormwater loadings for OU B. The comparison is made by computing the ratio of the estimated maximum total load as a result of groundwater discharge from Table 5-3 to the estimated stormwater loadings from Table 5-6. Results are presented for the western, central, and eastern sections, together with the sum of the three sections. Ratios with and without drydock dewatering are included, although the distinction affects only the central and eastern sections.

Ratios in Table 5-7 less than 1.0 indicate that stormwater loadings exceed the groundwater loadings; ratios greater than 1.0 would indicate that groundwater loadings exceed stormwater loadings. However, because the estimated maximum groundwater loads used in calculating the ratios have a high probability of overestimating the true groundwater loads, the ratios in Table 5-7 are biased high. That is, the ratios in Table 5-7 very likely *overestimate* the relative contribution of groundwater.

Even with the high bias, the ratios suggest that groundwater loadings are generally much less than stormwater loadings, particularly for the current and anticipated future OU B operating condition where the drydocks are dewatered. Only for arsenic in the western section, which has the maximum ratio of 0.83, does the estimated groundwater load approach the estimated stormwater load. Because this ratio reflects what is probably a significant overestimate of the groundwater contribution, true ratios would likely be much lower.

From this comparison it appears that stormwater loading to the inlet clearly exceeds direct groundwater loading to the inlet. Additional surface water loading to the inlet via drydock discharges is discussed next.

5.3.2 Drydock Discharges

The drydocks at the Bremerton Naval Complex are fully operational (i.e., dry) more than 95 percent of the time (USGS 1995). The drydock drains act as wells or groundwater sinks that have a large influence on groundwater flow in OU B. Under conditions of drydock operation, USGS estimated from groundwater modeling that all groundwater except shallow groundwater in the western area of the Bremerton Naval Complex discharges to the drydocks (USGS 1997, Figure 17). This implies that virtually all OU B groundwater except shallow groundwater in the area of Site 10 West flows to the drydocks, as discussed in the next two paragraphs.

The continuous pumping necessary to keep shipyard drydocks dewatered draws considerable amounts of seawater into site groundwater. This seawater mixes with groundwater originating in the upland portion of the Bremerton Naval Complex as site groundwater is pulled toward the dewatered drydocks, where it then enters the drydock dewatering systems and is eventually discharged to Sinclair Inlet. Based on groundwater modeling, USGS estimated approximately 8.6 cfs of groundwater discharge to the drydocks, of which approximately 5.7 cfs, or 66 percent, is seawater from the inlet (USGS 1997, Table 5).[26] By implication, approximately 2.9 cfs, or 34 percent, of the discharge to the drydocks would be from groundwater unaffected by (or unmixed with) seawater.

The USGS model-based estimates are consistent with the single set of flow measurements made by USGS at the drydocks in the summer of 1994 (USGS 1995). Based on the USGS measurements and interpretations, approximately 8.4 cfs of groundwater discharged to the drydocks, of which approximately 5.7 cfs, or 68 percent, was seawater from the inlet. By implication, approximately 2.7 cfs, or 32 percent, of the discharge to the drydocks was from groundwater unaffected by (or unmixed with) seawater. Discharge to the inlet was also estimated as part of USGS 1994 study. Total discharge from all drydocks (DD-1 through DD-6) to the inlet was estimated at approximately 14.3 cfs, of which approximately 11.1 cfs, or 78 percent, was seawater from the inlet. These outflow estimates would imply 3.2 cfs of groundwater in the total outflow. The 0.5-cfs difference from the 2.7 cfs estimated for groundwater inflow would appear to be due to various measurement and roundoff errors.

The groundwater analysis documented in Section 5.2 estimated that direct groundwater discharge to the inlet in the western area of OU B (represented by Site 10 West) has an expected value of approximately 0.18 cfs and does not exceed approximately 0.67 cfs with 95 percent probability (see Table 5-3). For all of the Bremerton Naval Complex, based on groundwater modeling, USGS estimated about 4.1 cfs of groundwater recharge from precipitation and about 1.3 cfs of direct groundwater discharge to the inlet, including OU A and OU NSC (USGS 1997, Table 5). As discussed in the preceding two paragraphs, about 2.7 to 3.2 cfs of site groundwater, not including seawater inflow from the inlet, enters the drydocks. With some consistency, these estimates indicate that the Bremerton Naval Complex discharges about 4.1 cfs of groundwater to the inlet either through the drydocks or directly to the inlet, with direct discharge constituting about one-third of the total groundwater flow. Although impractical to differentiate exactly between OU B and the rest of the Bremerton Naval Complex, it appears that when the drydocks are operating, approximately 4 percent (0.18 cfs of 4.1 cfs) to as much as 16 percent (0.67 cfs of 4.1 cfs) of OU B groundwater discharges directly to the inlet along Site 10 West.

[26] 1 cfs = 0.65 MGD.

Based on these flow estimates, it appears reasonable to conclude that most chemicals transported out of the shipyard by groundwater pass through the drydock dewatering systems. Direct discharge of groundwater to Sinclair Inlet would thus appear to be a comparatively minor route restricted to the western shipyard, outside the influence of the drydocks. This further supports the conclusion from Section 5-2 that chemical transport to the inlet from direct groundwater discharge is relatively small compared to transport from surface water discharges. Specifically, the groundwater analysis documented in Section 5.2 estimated that the maximum load of copper and lead to the inlet from the western area of OU B represented by Site 10 West was, respectively, approximately 3.28 and 0.34 kg/yr (Table 5-3). These loads can be cautiously contrasted to a limited extent with estimated past loads from drydock discharge, as discussed next.

As part of the USGS 1994 study (USGS 1995) chemical concentrations were measured in water samples collected at selected locations from the drydock drainage systems. Based on measured concentrations of copper and lead, USGS estimated an equivalent copper load of approximately 200 kg/yr and an equivalent lead load of approximately 300 kg/yr from the drydock discharge to the inlet (USGS 1999 with a subsequent unit conversion correction by URS).

Although the estimated drydock copper loading is 60 times higher than the estimated maximum loading from Site 10 West and the lead loading is 900 times higher, it is cautioned that these drydock load estimates reflect very limited and potentially unrepresentative data. They are based on only one set of flow and chemical measurements and are dominated by results from a few samples. In particular, 70 percent of the copper load and 48 percent of the lead load is based on the single chemical sample at location DD6-W-NB. Furthermore, the limited 1994 data predate systematic improvements in shipyard operations that have likely resulted in greatly reduced chemical concentrations in drydock discharge. Since 1994, improved drydock housekeeping and source control practices have been implemented at the Bremerton Naval Complex. However, even with these qualifications, it appears reasonable to conclude that surface water loading is the major source of chemical transport from OU B to the inlet, which is the principal conclusion of this section as stated next.

5.3.3 Conclusion

The comparison of groundwater and stormwater chemical loading, as discussed in Section 5.3.1, together with consideration of potential drydock discharges, as discussed in Section 5.3.2, indicate that surface water loading is the principal source of chemical loading from OU B to the inlet. Absent drydock discharges, which are regulated under the NPDES program, the conclusion remains that stormwater loading is the principal source of chemical loading from OU B to the inlet.

5.4 SOIL VAPOR PATHWAY

The soil vapor pathway is evaluated in this section for consistency with MTCA under WAC 173-340-745, which requires evaluation of the soil vapor pathway to determine whether air emissions at a site pose a threat to human health or the environment. Specifically, WAC 173-340-745(5)(b)(iii)(C) specifies that the soil vapor pathway shall also be considered in establishing industrial soil cleanup levels.

Because any potential risks associated with this pathway were considered negligible, the OU B risk assessment did not evaluate a soil vapor pathway for ambient air or indoor air. Potential risks from the soil vapor pathway were considered negligible based on the cumulative effects of several factors, including the following:

- Limited exposure for potential receptors because of the industrial nature of the site
- Resistance to vapor transport from structural barriers, including paving and floor slabs, covering virtually 100 percent of the site
- Confounding effects of emissions from industrial processes at the site
- Relatively low concentrations of VOCs in subsurface soil that could be a source of COCs in soil vapor

For the conditions at OU B, consistency with MTCA under WAC 173-340-745(5)(b)(iii)(C) does not require further evaluation of a soil vapor pathway beyond what is included in the remainder of this section. Specifically, the requirements of WAC 173-340-745(5)(b)(iii)(C) are interpreted to preclude a requirement for evaluating the vapor pathway if COC concentrations in site soil are not "significantly higher than a concentration derived for protection of groundwater for drinking water beneficial use." This is the situation at OU B, as is evident from the analysis presented in Table 5-8, which fulfills the requirements of WAC 173-340-745(5)(b)(iii)(C) to preclude further evaluation of the soil vapor pathway.

Table 5-8 compares measured concentrations of VOCs and TPH in OU B soil samples with MTCA soil cleanup levels for protection of groundwater for drinking water beneficial use, based on WAC 173-340, Table 740-1. As Table 740-1 notes, the soil cleanup levels are based on protection of groundwater for drinking water use using the procedures in WAC 173-340-747(4) and (6), which is consistent with WAC 173-340-745(5)(b)(iii)(C)(I) and (III). The exception is TPH-diesel which uses WAC 173-340-745(5)(b)(iii)(C)(II) to preclude the need for further soil vapor analysis. Note that although TPH concentrations may be higher in portions of OU NSC

(FISC) (URS 1995l), OU NSC is not part of OU B; OU NSC is covered by its own RI/FS and existing ROD.

The table clearly indicates that VOC and TPH concentrations measured in OU B soil are not "significantly higher than a concentration derived for protection of groundwater for drinking water beneficial use," consistent with MTCA. In fact, the vast majority of concentrations are below detection limits (excepting TPH-diesel) and are below the Table 740-1 cleanup levels for groundwater protection (last column of Table 5-8). Of the few exceedances, none could be considered "significant" in terms of a soil vapor pathway (or a groundwater pathway). It is thus concluded that further evaluation of the soil vapor pathway for OU B is not required under MTCA, consistent with WAC 173-340-745(5)(b)(iii)(C).

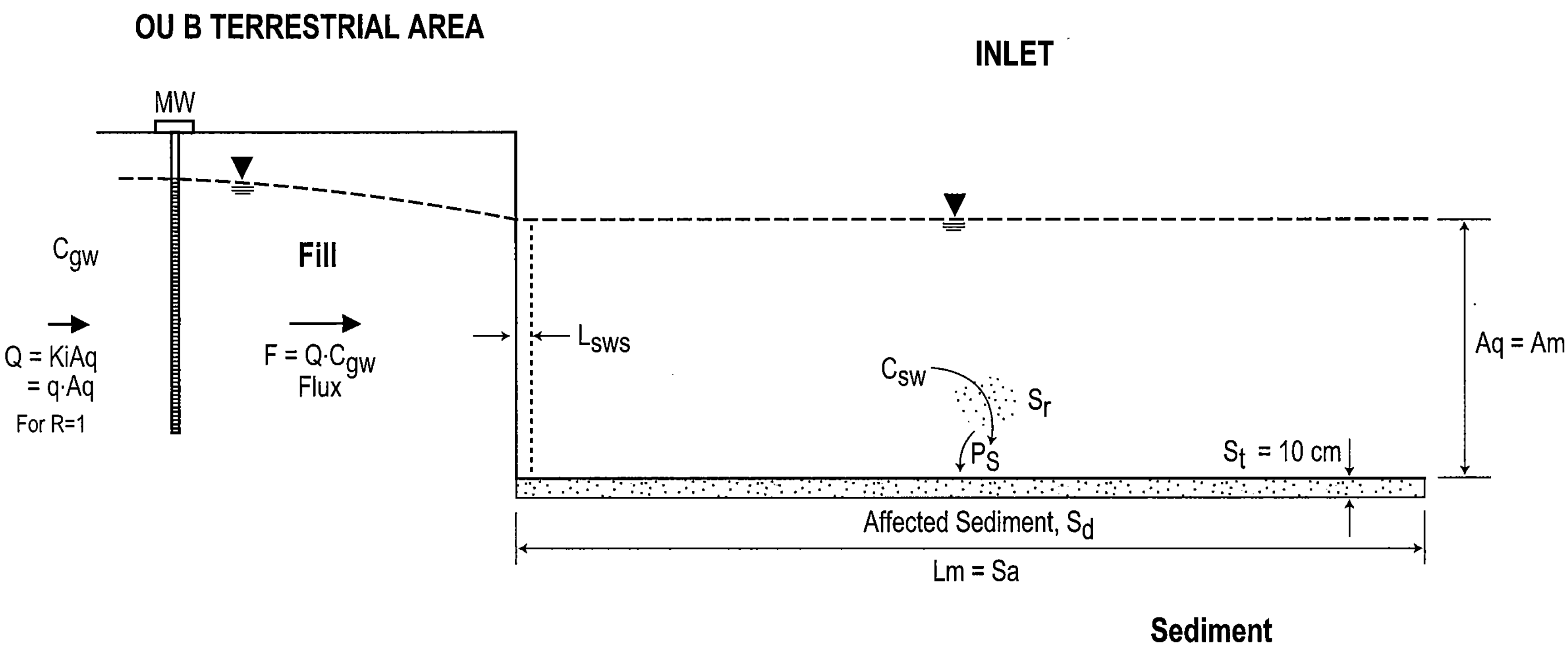
5.5 CONCLUSIONS

The following major conclusions for the terrestrial portion of OU B at the Bremerton Naval Complex are based on the analyses in this document:

- Potentially adverse future impacts on marine waters and sediment of Sinclair Inlet from chemicals in groundwater that discharges from terrestrial OU B are very limited.
- Site groundwater discharges are predicted not to result in SWS exceedances in surface water or SQS exceedances in marine sediment.
- Site groundwater is adequately protective of the inlet to conclude that active remediation of groundwater may not be warranted.
- Site groundwater is predicted to be protective of the site remedy for marine OU B (i.e., recontamination of marine sediments is not expected).
- The terrestrial soil to groundwater pathway is not predicted to pose a threat or potential threat to the inlet. (This conclusion does not extend to affected site soils transported to the inlet by stormwater loading or other surface water erosion effects, including potential mass wasting at exposed areas of the shoreline.)
- The soil vapor pathway is not a threat or potential threat, as concentrations measured in OU B soil do not appear to be significantly higher than a concentration derived for protection of groundwater for drinking water beneficial use.

- In light of the predictions summarized above, stormwater loading appears to be the principal source of chemical loading to the inlet. (Stormwater loading may include erosion or mass wasting at exposed, susceptible areas of the shoreline.)

It is assumed that appropriate site monitoring will be enacted as part of the remedy selected in the ROD to provide adequate verification of these expectations and conclusions.

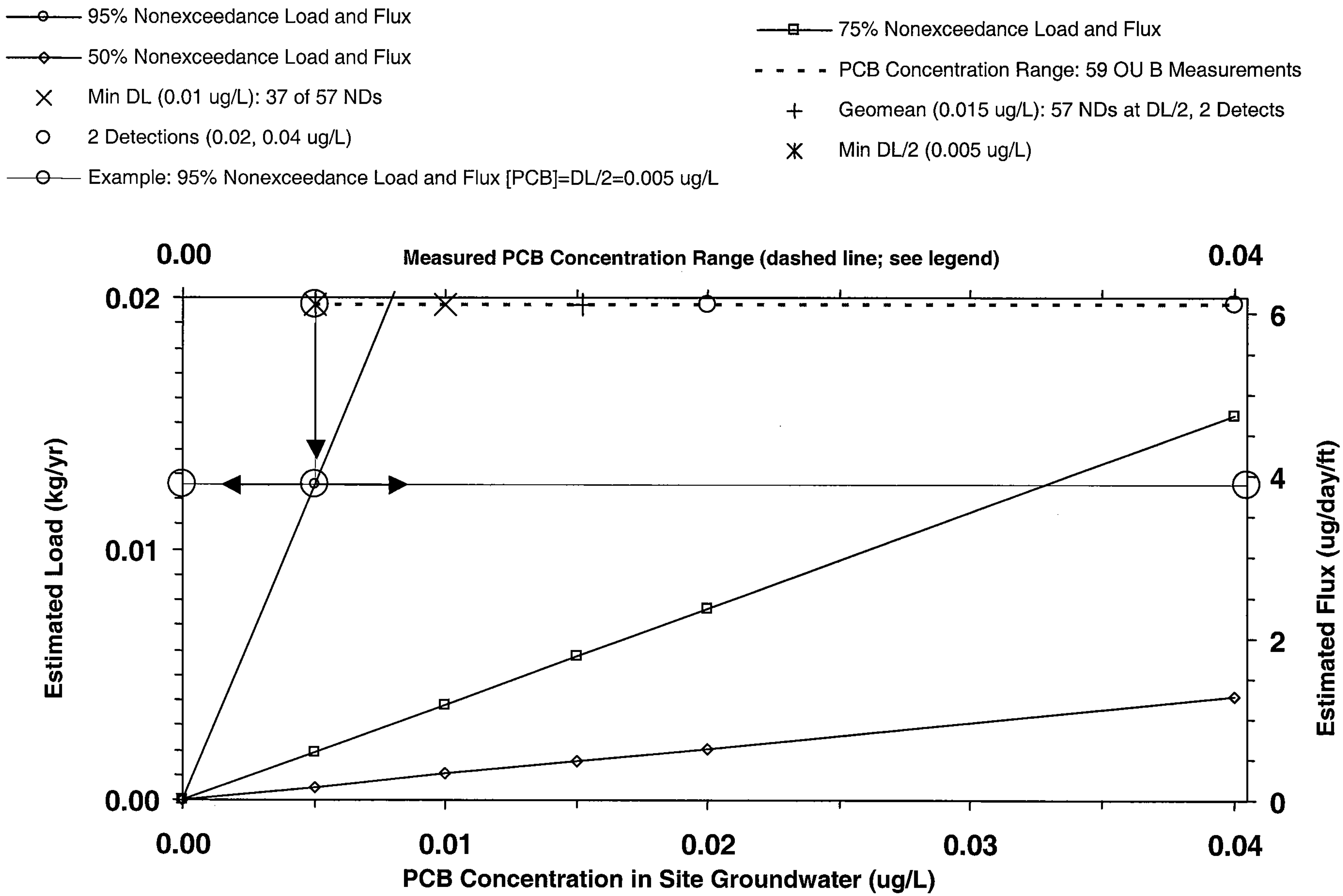


See Table 5-1 for Parameter Definitions.
All Dimensions Per Unit Length of Shoreline.

CLEAN
COMPREHENSIVE LONG TERM ENVIRONMENTAL ACTION NAVY

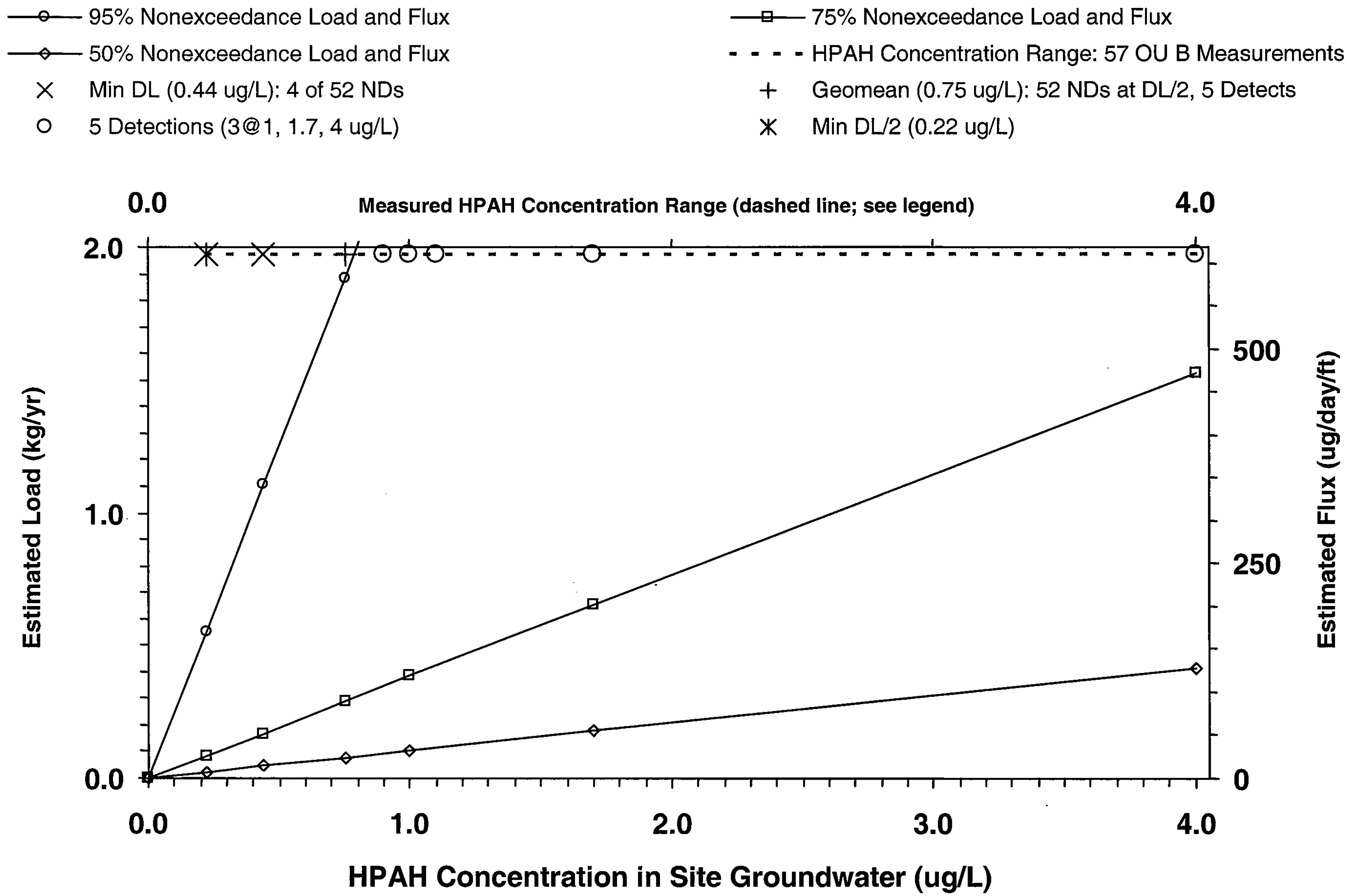
Figure 5-1
Idealized Schematic: Flux/Mixing Length/Sediment Contamination Analytical Models

CTO 0131
Bremerton Naval Complex OU B
Bremerton, WA
FINAL RI REPORT



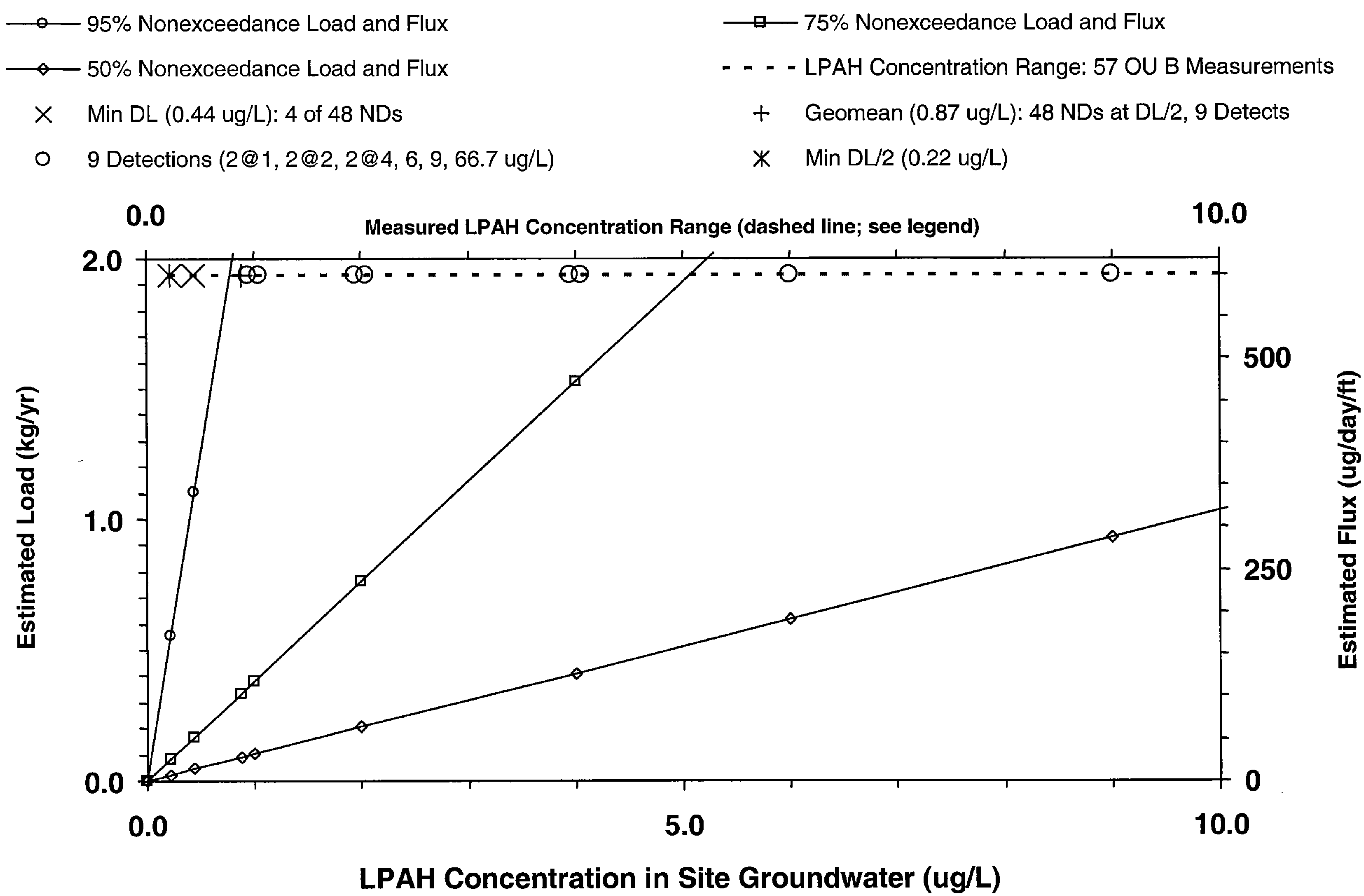
Notes: Load based on total length of OU B shoreline (8,850 ft); flux probabilities same as loading
DL = detection limit; ND = nondetection

CLEAN COMPREHENSIVE LONG TERM ENVIRONMENTAL ACTION NAVY	**Figure 5-2** **PCB Flux and Load From OU B Groundwater Discharge**	CTO 0131 Bremerton Naval Complex OU B Bremerton, WA FINAL RI REPORT



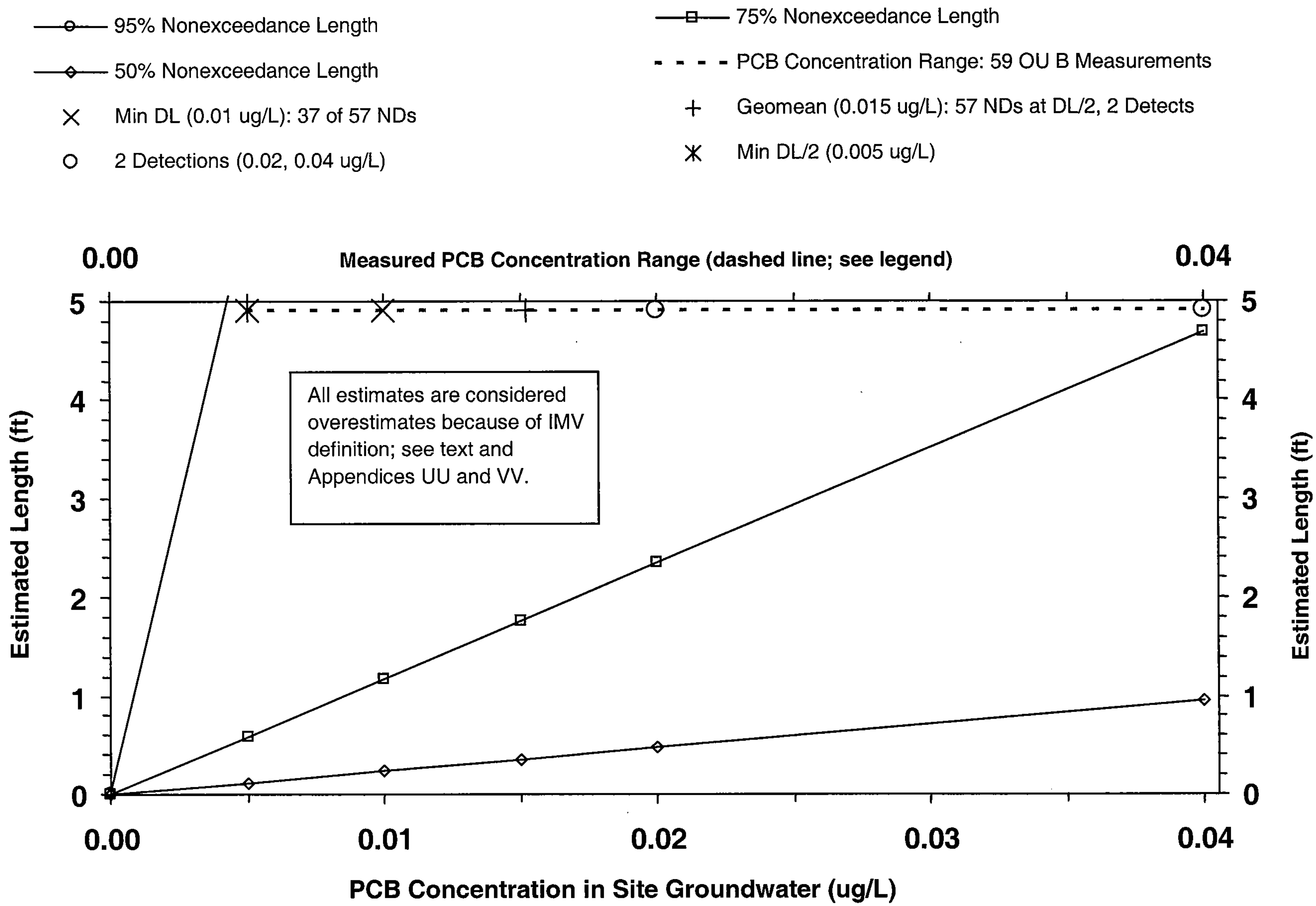
Notes: Load based on total length of OU B shoreline (8,850 ft); flux probabilities same as loading
DL = detection limit; ND = nondetection

CLEAN COMPREHENSIVE LONG TERM ENVIRONMENTAL ACTION NAVY	**Figure 5-3** **HPAH Flux and Load From OU B Groundwater Discharge**	CTO 0131 Bremerton Naval Complex OU B Bremerton, WA FINAL RI REPORT



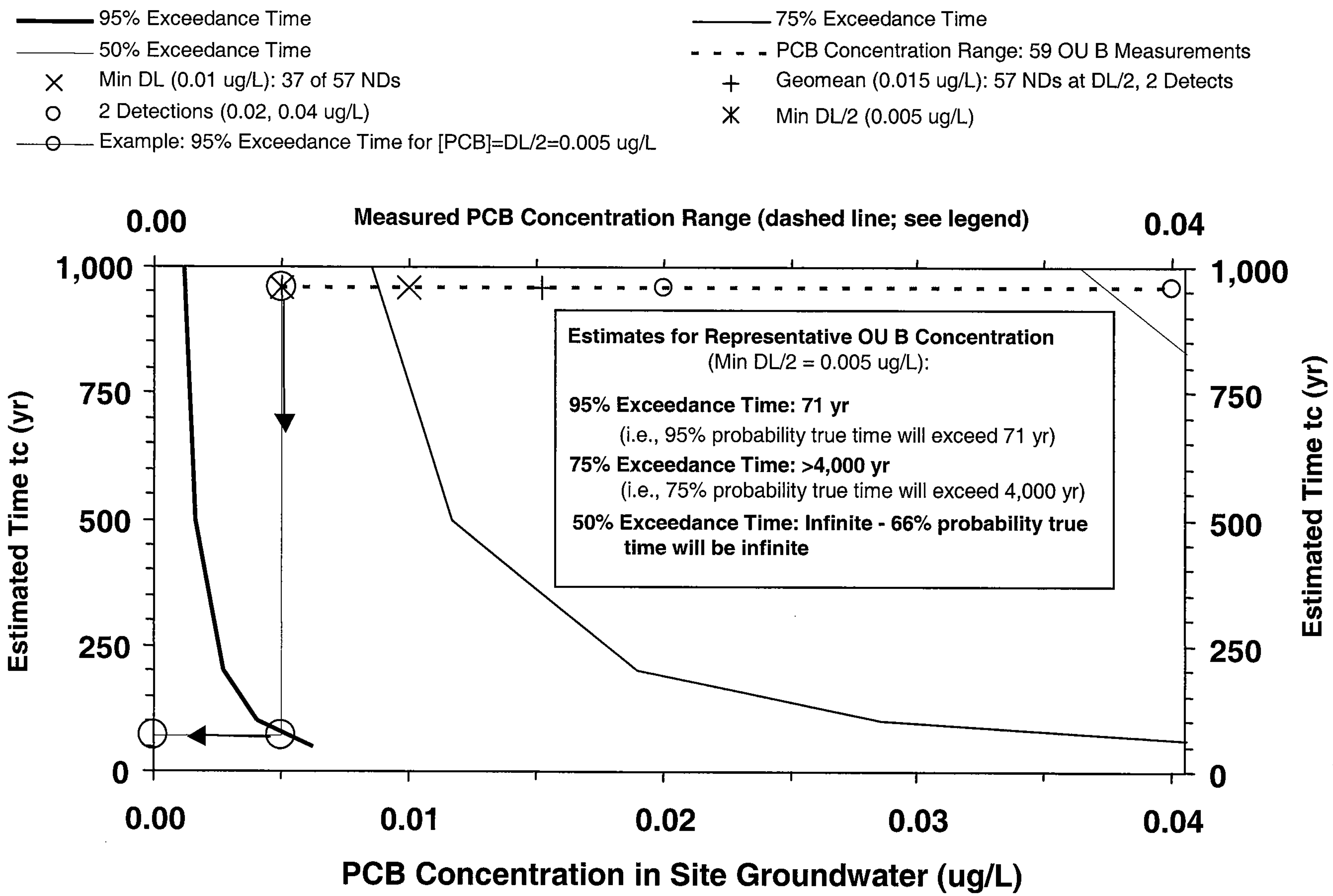
Notes: Load based on total length of OU B shoreline (8,850 ft); flux probabilities same as loading
DL = detection limit; ND = nondetection

CLEAN COMPREHENSIVE LONG TERM ENVIRONMENTAL ACTION NAVY	**Figure 5-4** **LPAH Flux and Load From OU B Groundwater Discharge**	CTO 0131 Bremerton Naval Complex OU B Bremerton, WA FINAL RI REPORT



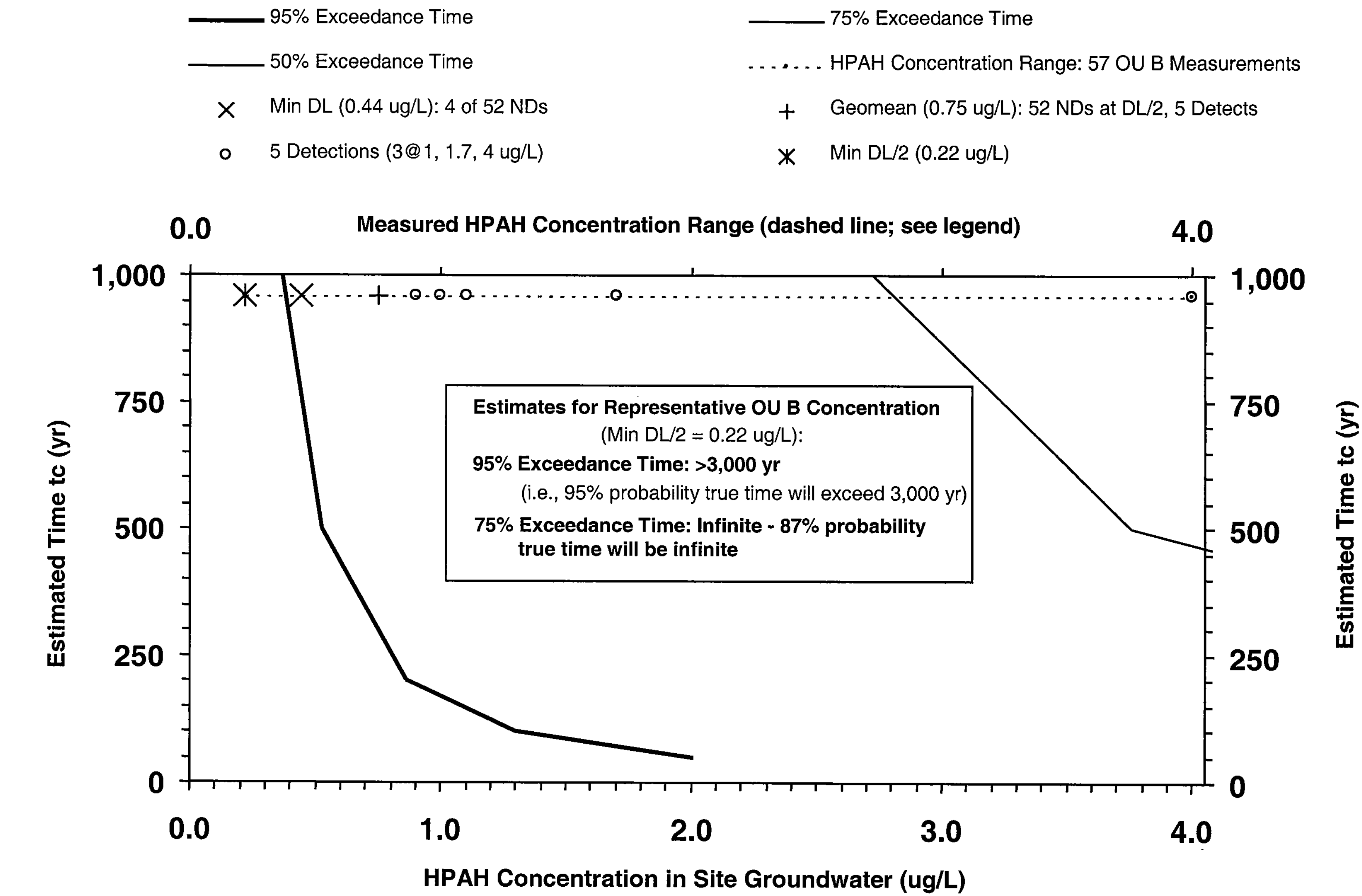
Notes: PCB SWS = 0.000027 ug/L
DL = detection limit; ND = nondetection

CLEAN COMPREHENSIVE LONG TERM ENVIRONMENTAL ACTION NAVY	**Figure 5-5** **Mixing Length for Groundwater Discharge to Meet PCB SWS**	CTO 0131 Bremerton Naval Complex OU B Bremerton, WA FINAL RI REPORT



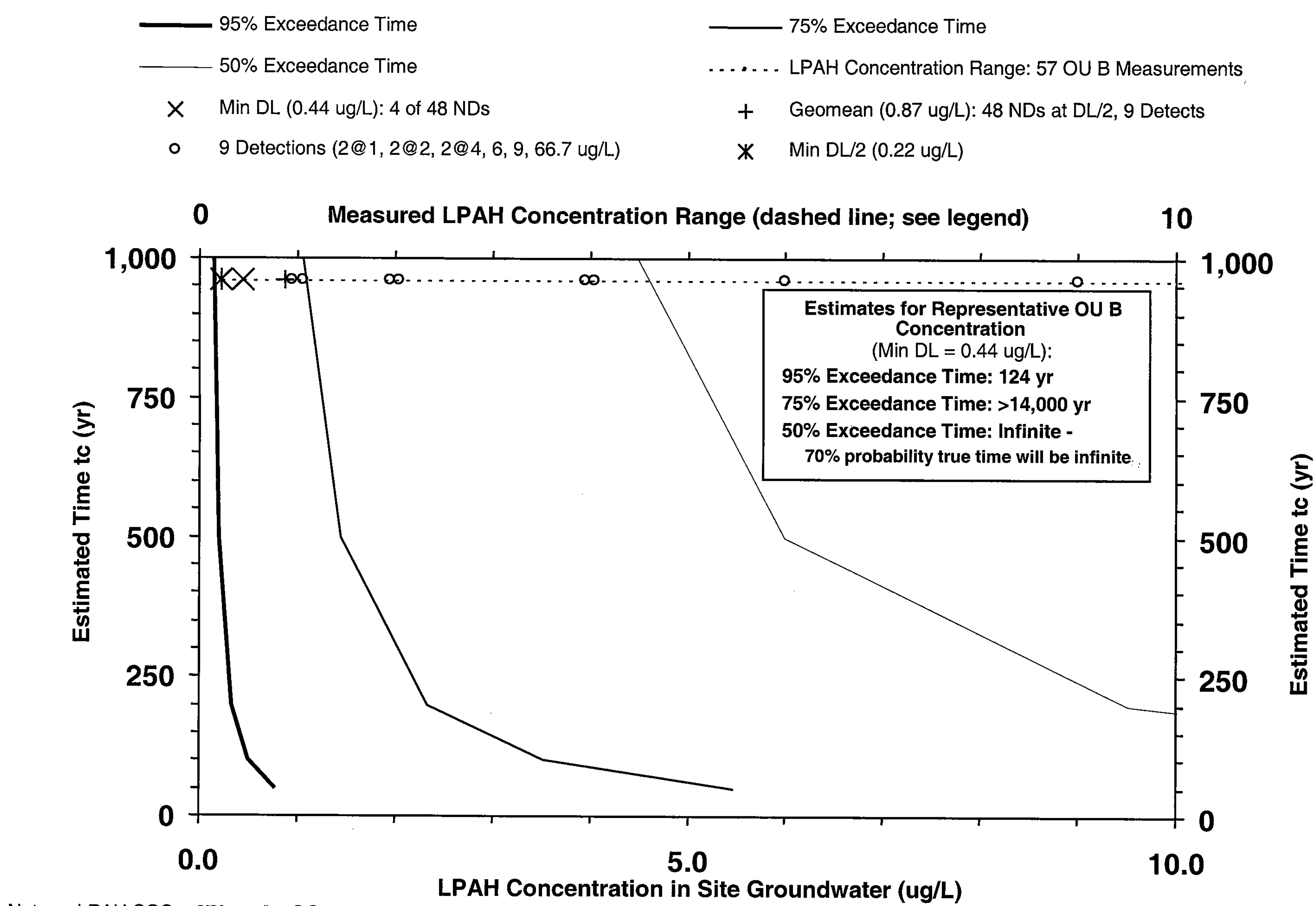
Notes: PCB SQS = 12 mg/kg OC
DL = detection limit; ND = nondetection

CLEAN COMPREHENSIVE LONG TERM ENVIRONMENTAL ACTION NAVY	**Figure 5-6** **Time for Site Groundwater Discharge to Raise PCB Concentration in Marine Sediments by 3 mg/kg OC**	CTO 0131 Bremerton Naval Complex OU B Bremerton, WA FINAL RI REPORT



Notes: HPAH SQS = 960 mg/kg OC
DL = detection limit; ND = nondetection

CLEAN COMPREHENSIVE LONG TERM ENVIRONMENTAL ACTION NAVY	**Figure 5-7** **Time for Site Groundwater Discharge to Raise HPAH Concentration in Marine Sediments to SQS**	CTO 0131 Bremerton Naval Complex OU B Bremerton, WA FINAL RI REPORT



Notes: LPAH SQS = 370 mg/kg OC
DL = detection limit; ND = nondetection

CLEAN COMPREHENSIVE LONG TERM ENVIRONMENTAL ACTION NAVY	**Figure 5-8** **Time for Site Groundwater Discharge to Raise LPAH Concentration in Marine Sediments to SQS**	CTO 0131 Bremerton Naval Complex OU B Bremerton, WA FINAL RI REPORT

Table 5-1
Marine Protectiveness Measures and Model Input Parameters

Protectiveness Measures	Model Input Parameters
Unit chemical mass flux into inlet, F	Q[a], Cgw
Chemical load (mass) into inlet, F_T	F, Ls
Mixing length required to achieve SWS, Lsws	F, Am, tf, Cas, SWS
Duration of time before chemicals in sediment exceed SQS, tc	F, Lm, Sd, St_0, Sr, Ps, SQS

[a]Q is the groundwater discharge rate to the inlet; Q = K I Aq R = q Aq R

Model Input Parameters:

Am - effective vertical mixing area per foot of shoreline; set equal to Aq (ft^2/ft)
Aq - effective unit discharge area (per 1-ft unit length along shoreline) (ft^2/ft)
Cas - COC concentration in ambient (background) Sinclair Inlet seawater (µg/L)
Cgw - dissolved concentration of COC in groundwater that directly discharge to the inlet (µg/L)
I - hydraulic gradient in groundwater discharge region (daily average, accounting for tidal effects) (ft/ft)
K - hydraulic conductivity of terrestrial fill in groundwater discharge region (ft/day)
Lm - affected sediment length normal to shoreline (ft)
Ls - shoreline length (ft)
Ps - proportion of COC mass in discharging groundwater that fixes onto marine sediment (unitless)
q - specific discharge
R - reduction factor for flow restriction to the inlet by quay walls or interception by drydock dewatering (unitless)
Sd - sediment dry density (dry unit weight) (lb/ft^3)
SQS - SMS sediment quality standard (concentration) for COC (mg/kg)
Sr - sedimentation rate (in/yr)
St_0 - initial sediment thickness (in)
SWS - marine surface water standard (concentration) for COC (µg/L)
tf - time to flush hypothetical inlet mixing volume, IMV (day)

Table 5-2
Model Input Parameter Values

	Units	Site 10W West	Site 1 Central	Site 8 East		All Sites 5% to 95% Range		CV
		Average or Expected Values				5%	95%	
Hydraulic Parameters								
Hydraulic Conductivity, K	ft/day	38	38	38		0.74	147	3.49
Hydraulic Gradient, i	ft/ft	0.009	0.009	0.009		0.001	0.03	1.38
Default Reduction Factor, R	unitless	1.0	1.0	1.0		na	na	0.00
R with Drydock Dewatering	unitless	1.0	0.02	0.02		na	na	0.00
Discharge Area, Aq	ft^2/ft	21	21	21		11	35	0.35
Flushing Time, tf	days	0.03	0.03	0.03		0.001	0.11	2.47
Maximum Shoreline Length, Ls	ft	2,100	3,300	3,450		na	na	0.00
R values = 1.0 for no drydock dewatering								
Sediment Parameters								
Affected Sediment Length, Lm	ft	266	266	266		10	999	2.47
Sediment Thickness, St	in	3.94	3.94	3.94		na	na	0.00
Sediment Density, Sd	lb/ft^3	39	39	39		28	53	0.20
Sedimentation Rate, Sr	in./yr	0.08	0.08	0.08		0.003	0.30	2.47
COC Proportion, Ps, for Inorganics	unitless	0.053	0.053	0.053		0.0020	0.20	2.47
COC Proportion, Ps, for Organics	unitless	1.00	1.00	1.00		1.00	1.00	0.00
Total Organic Carbon, TOC (for organic COCs)								
Existing sediments	percent	3.0	3.0	3.0		na	na	0.00
Depositional sediments	percent	1.4	1.4	1.4		na	na	0.00
Dissolved Concentrations		**Maximum Measured Values**						
Inorganic COCs					**Maximum**			
Arsenic	µg/L	10.9	5.9	7.35	10.9			0.00
Copper	µg/L	20.4	48.2	6	48.2			0.00
Mercury	µg/L	0.1	0.1	0.1	0.1			0.00
Lead	µg/L	2.1	51.3	4.8	51.3			0.00
Zinc	µg/L	54.8	296	153	296			0.00
Multiples of OUB Sitewide Average Concentrations								
Arsenic	unitless	4.7	2.5	3.2				
Copper	unitless	5.1	12.0	1.5				
Mercury	unitless	1.0	1.0	1.0				
Lead	unitless	3.0	72.6	6.8				
Zinc	unitless	1.8	9.5	4.9				
OU B Sitewide Concentrations								
Latest Post-1993 Data w/NDs=DL/2		**Avg**	**Geomean**	**Median**	**N**	**%NDs**		
Arsenic	µg/L	2.32	0.98	1.00	58	45%		
Copper	µg/L	4.02	1.73	1.50	57	30%		
Mercury	µg/L	0.1	0.1	0.1	58	100%		
Lead	µg/L	0.71	0.48	0.25	58	90%		
Zinc	µg/L	31.09	8.01	5.00	57	84%		
Concentration Reductions, Pd	unitless	1.00	1.00	1.00		1.00	1.00	0.00

Notes:
Total lead used for Site 1
Total zinc used for Site 8
DL/2 for mercury
Prob. Sr=0 = 0
CV = coefficient of variation
TOC from U.S. Navy 2000

N = number of samples; ND = nondetections; DL = detection limit
na = not applicable

Table 5-3
Chemical Mass Flux and Loading Estimates (Inorganic COCs)

No Drydock Dewatering (R=1)									
		Site 10W	**Site 1**	**Site 8**		**Site 10W**	**Site 1**	**Site 8**	
		West	**Central**	**East**		**West**	**Central**	**East**	
	Units								
		Average or Expected Values				**95% Nonexceedance Estimates**			
					Sum	**95% Probability Rate Less Than**			**Sum**
Unit Groundwater Discharge Rates	ft^3/day/ft	7.41	7.41	7.41		27.45	27.45	27.45	
Total Groundwater Discharge Rates	cfs	0.18	0.28	0.30	0.76	0.67	1.05	1.10	2.81
	MGD	0.12	0.18	0.19	0.49	0.43	0.68	0.71	1.82
Maximum Unit Flux						**95% Probability Flux Less Than**			
Arsenic	g/yr/ft	0.83	0.45	0.56		3.09	1.67	2.09	
Copper	g/yr/ft	1.56	3.69	0.46		5.79	13.68	1.70	
Mercury	g/yr/ft	0.01	0.01	0.01		0.03	0.03	0.03	
Lead	g/yr/ft	0.16	3.93	0.37		0.60	14.56	1.36	
Zinc	g/yr/ft	4.20	22.67	11.72		15.55	83.98	43.41	
		Maximum Values							
Maximum Loads					**Sum**				
Arsenic	kg/yr	1.75	1.56	1.86	5.17				
Copper	kg/yr	3.28	12.74	1.52	17.53				
Mercury	kg/yr	0.02	0.03	0.03	0.07				
Lead	kg/yr	0.34	13.56	1.21	15.11				
Zinc	kg/yr	8.81	78.22	38.67	125.70				
With Drydock Dewatering									
		Site 10W	**Site 1**	**Site 8**		**Site 10W**	**Site 1**	**Site 8**	
		West	**Central**	**East**		**West**	**Central**	**East**	
Reduction Factor R:		1.00	0.02	0.02		1.00	0.02	0.02	
	Units	**Average or Expected Values**				**95% Probability Rate Less Than**			
Unit Groundwater Discharge Rates	ft^3/day/ft	7.41	0.15	0.15	**Sum**	27.45	0.55	0.55	**Sum**
Total Groundwater Discharge Rates	cfs	0.18	0.01	0.01	0.19	0.67	0.02	0.02	0.71
	MGD	0.12	0.004	0.004	0.12	0.43	0.01	0.01	0.46
Maximum Unit Flux						**95% Probability Flux Less Than**			
Arsenic	g/yr/ft	0.83	0.01	0.01		3.09	0.03	0.04	
Copper	g/yr/ft	1.56	0.07	0.01		5.79	0.27	0.03	
Mercury	g/yr/ft	0.01	0.0002	0.0002		0.03	0.001	0.001	
Lead	g/yr/ft	0.16	0.08	0.01		0.60	0.29	0.03	
Zinc	g/yr/ft	4.20	0.45	0.23		15.55	1.68	0.87	
		Maximum Values							
Maximum Loads					**Sum**				
Arsenic	kg/yr	1.75	0.03	0.04	1.82				
Copper	kg/yr	3.28	0.25	0.03	3.57				
Mercury	kg/yr	0.02	0.001	0.001	0.02				
Lead	kg/yr	0.34	0.27	0.02	0.63				
Zinc	kg/yr	8.81	1.56	0.77	11.15				

Notes:
Maximum Load = Maximum Unit Flux x Maximum Shoreline Length
MGD = million gallons per day (1 cfs = 0.65 MGD)

Table 5-4
Surface Water Mixing Lengths to Achieve SWS (Inorganic COCs)

No Drydock Dewatering (R=1)							
		Site 10W	**Site 1**	**Site 8**	**Site 10W**	**Site 1**	**Site 8**
		West	**Central**	**East**	**West**	**Central**	**East**
	Units						
		Average or Expected Values			**95% Nonexceedance Estimates**		
					95% Probability Lengths Less Than		
Mixing Length to Achieve SWS							
Arsenic	ft	0.03	0.01	0.02	0.08	0.04	0.05
Copper*	ft	0.07	0.16	0.02	0.21	0.49	0.06
Mercury	ft	0.04	0.04	0.04	0.13	0.13	0.13
Lead	ft	0.001	0.08	0.01	0.004	0.23	0.02
Zinc	ft	0.01	0.04	0.02	0.02	0.12	0.06

With Drydock Dewatering							
Reduction Factor R:		1.00	0.02	0.02	1.00	0.02	0.02
		Site 10W	**Site 1**	**Site 8**	**Site 10W**	**Site 1**	**Site 8**
		West	**Central**	**East**	**West**	**Central**	**East**
		Average or Expected Values			**95% Nonexceedance Estimates**		
					95% Probability Lengths Less Than		
Mixing Length to Achieve SWS							
Arsenic	ft	0.03	0.0003	0.0003	0.08	0.001	0.001
Copper*	ft	0.07	0.003	0.0004	0.21	0.010	0.001
Mercury	ft	0.04	0.001	0.001	0.13	0.003	0.003
Lead	ft	0.001	0.002	0.0001	0.004	0.005	0.000
Zinc	ft	0.01	0.001	0.0004	0.02	0.002	0.001

*incremental length since ambient seawater concentration exceeds SWS for copper

Notes:
Estimates based on maximum unit fluxes, as presented in Table 5-3
Regulatory Criteria:
Marine surface water quality standards (SWS)
Arsenic from MTCA Method A
All others from Washington WAC 201A

COC	SWS (mg/L)	Ambient Seawater (mg/L)
Arsenic	5.0	1.37
Copper	3.1	5.5
Mercury	0.025	0.0
Lead	8.1	1.25
Zinc	81	5.0

Table 5-5
Time Before Chemicals in Marine Sediments Exceed SQS (Inorganic COCs)

Time (yr) Until Concentrations in Upper 10 cm Exceed SQS
No Drydock Dewatering (R=1) All Estimates

		Site 10W West	Site 1 Central	Site 8 East	Site 10W West	Site 1 Central	Site 8 East
	Units						
Time for Sediments to Reach SQS		**Average or Expected Values**			**95% Exceedance Estimates**		
					95% Probability Time Exceeds		
Arsenic	yr	Infinite	Infinite	Infinite	>10,000	>10,000	>10,000
Copper	yr	Infinite	Infinite	Infinite	>10,000	>10,000	>10,000
Mercury	yr	Infinite	Infinite	Infinite	>10,000	>10,000	>10,000
Lead	yr	Infinite	Infinite	Infinite	>10,000	>10,000	>10,000
Zinc	yr	Infinite	Infinite	Infinite	>10,000	>1,000	>5,000
Time for Sediments to Reach SQS		**Probability Time is Infinite**					
Arsenic		0.93	0.95	0.95			
Copper		0.97	0.95	0.99			
Mercury		0.92	0.92	0.92			
Lead		1.00	0.95	0.99			
Zinc		0.94	0.85	0.90			
Time to Reach SQS for Sr=0		**Median Estimates for Sr=0**					
Assumes No Sedimentation (included for sensitivity)							
Arsenic	yr	12,866	23,769	19,080			
Copper	yr	47,035	19,907	159,920			
Mercury	yr	10,087	10,087	10,087			
Lead	yr	527,210	21,582	230,654			
Zinc	yr	18,407	3,408	6,593			

Notes:
Estimates assume initially clean sediments and clean depositional sediments
Estimates based on maximum unit fluxes, as presented in Table 5-3
Regulatory Criteria (WAC 173-204):
Sediment Management Standards (SMS) SQS

	mg/kg
Arsenic	57
Copper	390
Mercury	0.41
Lead	450
Zinc	410

Table 5-6
Stormwater Mass Loading Estimates

	West (kg/yr)	Central (kg/yr)	East (kg/yr)	Sum (kg/yr)
Arsenic	2.12	5.90	4.29	12.31
Copper	123.94	167.90	134.81	426.65
Mercury	0.46	0.18	0.16	0.79
Lead	38.94	221.48	156.34	416.76
Zinc	317.42	756.89	251.20	1,325.52

Data from Appendix RR, Table 5-6
Does not include drydock discharge
Does not include OU A or OU NSC (FISC)

Table 5-7
Comparison of Mass Loadings Ratios From Groundwater Discharge and Stormwater Discharge

	No Drydock Dewatering			
	West	Central	East	Sum
Arsenic	0.83	0.26	0.43	0.42
Copper	0.03	0.08	0.01	0.04
Mercury	0.04	0.15	0.16	0.09
Lead	0.01	0.06	0.01	0.04
Zinc	0.03	0.10	0.15	0.09

	With Drydock Dewatering			
	West	Central	East	Sum
Arsenic	0.83	0.01	0.01	0.15
Copper	0.03	0.002	0.0002	0.01
Mercury	0.04	0.003	0.003	0.02
Lead	0.01	0.001	0.0002	0.002
Zinc	0.03	0.002	0.003	0.01

Ratios are based on maximum loads
from Table 5-3 divided by stormwater
load estimates from Table 5-6

Table 5-8
Comparison of VOCs and TPH Detected in OU B Soils to Soil Cleanup Levels for Groundwater Protection

Analyte	Quantity Tested[a]	Quantity Detected[a]	Frequency of Detections (percent)	Max. Detect Value[a] (mg/kg)	MTCA Method A Soil Cleanup Level for Groundwater Protection[b] (mg/kg)	Frequency Above Groundwater Protection Level[c] (percent)
Subsurface Soil						
1,1,1-Trichloroethane	378	5	1.3	5.3	2	<1.3
BTEX (total)	378	50	13	1.99	--	
Benzene	388	0	0	(all NDs)	0.03	0
Ethylbenzene	378	20	5.3	0.79	6	0
Tetrachloroethene	378	21	5.6	1.3[e]	0.05	<5.6
Toluene	378	16	4.2	0.057	7	0
Trichloroethene	378	14	3.7	0.051	0.03	<3.7
Xylene (total)	378	47	12	1.5	9	0
TPH-Diesel	90	58	64	2,100	10,000[f]	0[f]
TPH-Gasoline	80	8	10	24	100	0
Surface Soil						
BTEX (total)	7	1	14	0.014	--	
Benzene[d]	7	0	0	(all ND)	0.03	0
Ethylbenzene	7	1	14	0.002	6	0
Tetrachloroethene	7	2	29	0.003	0.05	0
Toluene	7	1	14	0.003	7	0
Xylene (total)	7	1	14	0.009	9	0
TPH-Diesel	4	3	75	55	10,000[f]	0[f]

[a]OU B soil sample data from Table 4-21, which includes chemicals detected only at OU B
[b]Cleanup levels from WAC 173-340, Table 740-1
[c]Basis to preclude further soil vapor pathway analysis
[d]Benzene data from Appendix M (all NDs)
[e]The maximum PCE soil concentration is at 28-30 feet bgs and is related to the PCE plume originating off site
[f]Exclusion of soil vapor analysis for TPH-Diesel based on WAC 173-340-745(5)(b)(iii)(C)(II)

Note:
Cleanup level based on WAC 173-340-745(5)(b)(iii)(C) unless otherwise noted