6.0 SCREENING-LEVEL ASSESSMENT OF MERCURY BIOACCUMULATION

6.1 Summary

In this section, a sediment screening level for mercury is developed which is conservatively protective of potential bioaccumulation risks to human health and to high trophic level wildlife receptors. This evaluation is site-specific, and is not intended to apply outside of the WW Area.

As discussed in previous sections, historical mercury releases have resulted in a sediment mercury concentration gradient offshore from the Whatcom Waterway. A similar mercury gradient has also been observed in adult male Dungeness crab muscle tissue samples collected in this area. A simple linear regression equation best fit the relationship between measured tissue concentrations and the average surface sediment concentration within the species' home range. This empirical sediment-to-tissue regression relationship was also consistent with regional bioaccumulation data, including age-adjusted bottomfish collected from other areas of Puget Sound containing elevated mercury concentrations in sediment.

Using screening-level risk assessment techniques, a conservative tissue benchmark mercury level was calculated to protect tribal fishers who may consume relatively large amounts of seafood from Bellingham Bay. Based on recent literature reviews, this benchmark level is also protective of fish-eating wildlife. The tissue benchmark level was then input into the empirical sediment-to-tissue regression relationship to determine a site-specific, health-based sediment screening level for mercury. The sediment screening level determined using this conservative process was 1.2 mg/kg. For the WW Area, sediments exceeding this health-based screening level generally fell within those areas of the site also targeted for cleanup to address sediment toxicity concerns identified in Section 5.0. Thus, cleanup necessary to comply with sediment toxicity criteria will also address human health and wildlife food web concerns.

6.2 Bioaccumulative Chemicals of Potential Concern

Of the seven chemicals detected in surface sediments of the WW Area which exceeded SQS chemical criteria (see Section 4.3), only mercury has been regularly detected in fish and shellfish tissue samples (see data compilation provided in Appendix F). Other chemicals of potential concern in sediment at the site such as phenol and 4-methylphenol have not been detected in fish and shellfish tissue. This is consistent with the low bioaccumulation potential of these compounds (Callahan et al., 1979).

Mercury is the principal chemical of concern in the WW Area, and thus was

the focus of this bioaccumulation assessment. Although other potentially bioaccumulative chemicals such as polychlorinated biphenyls (PCBs) and polychlorinated dibenzo-*p*-dioxins and dibenzofurans (PCDDs/PCDFs) have been identified in other urban embayments of Puget Sound (PTI, 1991; O'Neill et al., 1995), these chemicals were not a focus of the WW Area bioaccumulation assessment for the following reasons:

- From previous studies, screening-level data are available on the concentrations of PCBs, PCDDs/PCDFs, and other potentially bioaccumulative chemicals in fish and/or shellfish tissues collected within the WW Area (see Hart Crowser, 1996b and 1996f). The maximum detected tissue concentrations of these chemicals are within the regional background ranges reported for Puget Sound, though relatively little data are currently available on PCDD/PCDF concentrations. Conversely, mercury has been regularly detected in WW Area tissue samples at concentrations above regional background. This condition is depicted on Figure 6-1, which presents a summary plot of the distribution of adult male Dungeness crab muscle tissue mercury concentrations in the greater Bellingham Bay area;

- Historical wastewater discharges to the WW Area, including mercury discharges, were located adjacent to the G-P Log Pond and close to the head of Whatcom Waterway. In sediment deposition environments such as the WW Area (see Section 3.0), releases of PCBs, PCDDs/PCDFs, and other similar chemicals, should they have occurred, would have rapidly settled and remained localized around the discharge location. The localized distribution of similar chemicals such as phenol and 4-methylphenol at the site (see Figure 4-2) is consistent with this hypothesis. Because of the lower solubility (i.e., higher equilibrium partitioning coefficients) characteristic of PCBs and PCDDs/PCDFs relative to 4-methylphenol and phenol (Callahan et al., 1979), even less distribution of PCB and PCDD/PCDF is anticipated;

- Conversely, mercury is more soluble and mobile in the aquatic environment, and therefore is transported a greater distance through the site, resulting in greater bioaccumulation concerns. The relatively widespread distribution of mercury concentrations in sediment at the site supports this hypothesis (see Figure 4-1). Therefore, focusing cleanup efforts on mercury will also address other potential contaminants; and

- Under the state SMS, if a sediment sampling location fails biological testing it must be addressed by cleanup, regardless of the chemical composition. Biological testing as described in Section 5.0 identified contiguous areas of sediment toxicity within the WW Area which exceed state cleanup standards (Figure 5-1). Sediment cleanup actions that address these concerns will also encompass other contaminants.

6.3 Summary of Existing Mercury Bioaccumulation Data

A summary of available data used to evaluate the empirical relationship between tissue concentrations and home range average surface sediment mercury concentrations is presented below. Additional information on the procedure used to pair sediment and tissue data for the regression analysis is also presented in this section.

6.3.1 Data Sources

Synoptic, quality-assured tissue and sediment data collected in Puget Sound areas characterized by elevated mercury concentrations (i.e., above the SQS) are primarily available from five information sources (generally listed in chronological order):

- Bellingham Bay pre-RI/FS studies by Huxley College, Ecology and DNR (1974 and 1990 samplings)(Nelson et al., 1974; SAIC, 1990; and Cubbage, 1991);

- West Eagle Harbor RI/FS assessments by EPA (1989 to 1995 samplings) (CH2M Hill, 1991; and Hart Crowser, 1995);

- Puget Sound Ambient Monitoring Program (PSAMP; 1990 to 1995 samplings) (Patrick, 1996; O'Neill et al., in preparation; and Ecology, unpublished data);

- Sinclair Inlet - Puget Sound Naval Shipyard RI/FS (1994 and 1995 samplings) (URS, 1996); and

- Bellingham Bay Dungeness crab investigations by WDFW and Ecology (1997 sampling generally concurrent with this RI/FS) (Lippert, 1997; L. Weiss, written communication, 1997).

For the purpose of this bioaccumulation assessment, data were used to evaluate the empirical relationship between tissue and sediment mercury concentrations if the following criteria were met:

- Sufficient information was available to verify the accuracy and representativeness of each tissue and sediment sample result (e.g., field sampling records; analytical quality control);

- Local habitats present in the WW Area (primarily steep riprap, bulkheads and subtidal soft silt; see Section 7.0) could reasonably support the species sampled. Pile perch and striped seaperch data were not retained in the WW Area bioaccumulation assessment for this reason;

- The specific tissue analyzed was known to be consumed regularly by regional fishers (note: only English sole liver was excluded based solely on this criterion, and is discussed further in Section 6.8 below);

- Sufficient data were available for each species to assess the correlation between tissue and sediment concentrations (i.e., a minimum of three data points spanning sediment mercury concentrations ranging from background to the SQS of 0.41 mg/kg or above). Pile perch and striped seaperch data did not meet this criterion;

- Tissue data were collected in areas not characterized by relatively low bioavailability. The apparently anomalous behavior of the Sinclair Inlet data (i.e., relatively low tissue concentration for the corresponding sediment concentration) resulted in the exclusion of these data from the WW Area bioaccumulation assessment; and

- At least one contemporary surface sediment sample result was available within the home range of the sample collection site (see below).

- Using these data acceptance criteria, the following four tissue data sets were retained for the detailed bioaccumulation analysis:

- Dungeness crab muscle tissue data collected during 1990 and 1997 in the greater Bellingham Bay area by the State of Washington (Ecology, WDFW, and DNR)(SAIC, 1990; Cubbage, 1991; Lippert, 1997; L. Weiss, written communication, 1997);

- Red rock crab muscle tissue data collected in 1974 from the WW Area by Huxley College (Nelson et al., 1974), and in 1990 from Port Madison and West Eagle Harbor by EPA (CH2M Hill, 1991);

- English sole muscle tissue data collected over the period from 1991 to 1995 at numerous sites in Puget Sound by PSAMP (O'Neill et al., in preparation); and

- Mixed hardshell whole body clam tissue data collected over the period from 1990 to 1993 in Bellingham Bay and Puget Sound reference areas by Ecology and DOH (Cubbage, 1991 and Patrick, 1996).

The local and regional locations for which acceptable tissue and sediment samples are available for this mercury bioaccumulation analysis are depicted on Figure 6-2. The sample data are summarized in Table 6-1.

6.3.2 Sediment Exposure Estimates

Because the data with the exception of the hardshell clam data were collected on mobile species, the sediment mercury concentration at the point of capture for these mobile species may not be the most accurate indicator of that organism's exposure to mercury. A better estimate was obtained by averaging sediment concentrations throughout each fish and shellfish species' home range.

The home ranges of Dungeness crab, Red rock crab, and English sole have been studied in several regional investigations, and the results of these studies have been published in the scientific literature and in Puget Sound Dredge Disposal Analysis (PSDDA) reports. Post-larval and juvenile crabs tend to seek cover during their first 1 to 2 years of life. This lack of movement, and residence in intertidal and shallow subtidal areas, probably serves to protect them from predation. However, older crabs, particularly males, move greater distances. Mayer (1973) tagged Dungeness crabs in Similk Bay and found that most crabs resided in the general area of tagging, with movements of generally less than 1.6 kilometers (km) in 50 days. Typical rates of movement reported by various investigators range from 0.1 to 1 km per day (Waldron, 1958; Breen, 1985). With the exception of seasonal onshore-offshore migrations, these studies have shown that Dungeness crab movements are essentially random and non-directed.

In a review of the available literature, PSDDA (1988) reported an average home range for English sole of approximately 9 km^2, and used that value in a quantitative bioaccumulation assessment. Considering the available data, the home range of English sole, Red rock crab, and Dungeness crab in unconstrained areas of Puget Sound and the Straits of Georgia reportedly ranges from roughly 2 to 20 square kilometers (km^2). For the purpose of this analysis, an average unconstrained home range of approximately 10 km^2 was assumed for these three mobile species evaluated. The 10 km^2 area can be approximated as a circle with a radius of 1.8 km (1.1 miles; Figure 6-1).

It should be noted, however, that estimates of the areal extent of fish and shellfish home ranges are uncertain. The average estimate summarized above represents a best approximation of typical home ranges, based on a review of available information.

To estimate sediment exposure corresponding to each tissue sample, available surface (0- to 10-cm) sediment samples collected within the estimated home range radius of the tissue sampling location were used to calculate an areal-weighted (concentration contour-based) average surface sediment concentration. Mercury concentration distributions in West Eagle Harbor are presented in CH2M Hill (1989) and Hart Crowser (1995). Historical mercury distributions within the WW Area are presented by Nelson

et al. (1974), and contrast with the lower concentrations detected during the more recent RI sampling (Figure 4-1). Mercury concentrations in Elliott Bay were estimated using statistical analyses of the large database available for that area (Hart Crowser, unpublished block kriging data). In situations where no mercury gradient was evident in the sediment data (e.g., middle Duwamish River and reference areas such as Port Madison), sediment concentrations of mercury were simply averaged within the home range of the species being evaluated. Only sediment samples collected seaward of the ordinary high water elevation were used in this calculation.

Additionally, an effort was made to select sediment data that were most representative of conditions which existed at the time of tissue sampling. In most cases, paired sediment and tissue data were collected within a two-year time frame. However, a more liberal definition of comparable time frames was applied to tissue and sediment data collected from reference areas. Within these low concentration areas, sediment mercury levels do not appear to have changed significantly over time. Sediment concentrations of mercury identified as non-detects by the laboratory were assigned a value one-half of the reported detection limit.

6.3.3 Average Tissue Concentration Estimates

As previously discussed, the estimated home range of each species was used to determine appropriate pairing of individual sampling locations with sediment chemistry data. Multiple tissue samples collected within the estimated home range radius (for example, where the home range estimates of a species overlapped in adjacent sampling areas) were averaged into a single estimate of tissue concentration for the purposes of this bioaccumulation analysis. In addition, some fish/shellfish tissue results were derived from composite samples of multiple specimens and therefore provided an estimate of average tissue concentrations rather than an estimate of mercury concentrations in individual fish. Tissue concentrations of mercury identified as non-detects by the laboratory were assigned a value one-half of the reported detection limit.

6.4 Other Factors Influencing Tissue Concentrations

As discussed above, the intent of this screening-level bioaccumulation assessment was to assess the empirical relationship between the measured tissue concentrations and the home range average surface sediment mercury concentration. However, previous analysis of the extensive PSAMP database performed by O'Neill et al. (in preparation) has shown that tissue mercury concentrations vary according to both age and sediment concentrations. In many different species, older fish contain significantly higher mercury (primarily methylmercury) concentrations in their tissues than younger individuals.

Figure 6-3 presents a scatter plot of English sole muscle tissue concentrations as a function of the mean composite fish age for all Puget Sound areas with elevated mercury concentrations (i.e., sediment above 0.41 mg/kg). Most of the data presented on Figure 6-3 were collected by PSAMP and URS (1996). The linear regression equation describing these data is summarized as follows:

$$[Hg]_M = -0.002 + 0.0107 \times MCA;\ r^2 = 0.76$$

where:

$[Hg]_M$ = English sole muscle tissue concentration in mg/kg wet weight; and

MCA = Mean composite fish age in years.

A similar regression relationship, though with a slightly lower slope (0.0085 vs. 0.0107), was observed for English sole muscle tissue collected from regional reference locations, characterized by sediment mercury concentrations at or below 0.1 mg/kg.

As discussed by O'Neill et al. (in preparation), liver tissue mercury concentrations in fish vary significantly with both age and sediment concentration when both variables are evaluated in a multiple regression. The data evaluated by O'Neill also suggest that the average age of English sole within a given embayment may approach 8 years. Therefore, when age data were available from the regional English sole tissue studies, the age regressions summarized above were used to age-normalize each sample result to represent 8-year-old fish. (Note: age-normalization and multiple regression techniques yielded equivalent bioaccumulation estimates in this evaluation.)

Age data were frequently not available for Dungeness crab, Red rock crab, and hardshell clams. In this case, legal-sized shellfish specimens (e.g., adult male Dungeness crab with a carapace width exceeding 160 mm) were segregated, as appropriate, from the remaining data and analyzed separately in the bioaccumulation analysis.

Mercury is not lipophilic, and tissue lipid content is not considered to be a significant factor contributing to the variability in tissue mercury concentration for the species sampled by PSAMP (O'Neill et al., in preparation). Further, no literature reports were located which identified tissue lipids as a significant determinant of mercury bioaccumulation. Accordingly, tissue mercury concentrations were not lipid-normalized for this analysis.

6.5 Tissue-Sediment Regression Analysis

Several statistical techniques were evaluated to assess the empirical relationship between tissue and sediment mercury concentrations. Techniques evaluated included assessment of the sediment:tissue ratio (also known as the biota-sediment accumulation factor [BSAF]) and linear regression analyses. Based on these evaluations, a simple linear regression analysis yielded the best-fit relationship between paired sediment and tissue mercury concentrations. The regression relationships are depicted on Figure 6-4. The best-fit (least squares) equations, correlation coefficients (r^2), and significant levels (P) of the regressions for individual species are reported in Table 6-2.

The most significant regression relationship (lowest P) between sediment and tissue concentrations was observed for adult male Dungeness crab muscle tissue (Table 6-2). The Dungeness crab regression also yielded the most conservative bioaccumulation estimate of the species evaluated (i.e., the highest tissue concentration predicted at sediment mercury concentrations exceeding the 0.41 mg/kg SQS). English sole and Red rock crab muscle exhibited somewhat lower bioaccumulation characteristics, and the regression equations for these tissues were not statistically significant ($P > 0.05$). Nevertheless, as depicted on Figure 6-4, the Dungeness crab regression line bounded the *maximum* English sole and Red rock crab muscle tissue concentrations observed in other areas of Puget Sound containing elevated sediment mercury concentrations (e.g., Eagle Harbor). The Dungeness crab muscle regression equation thus provided a conservative upper-bound estimate of mercury bioaccumulation for a range of species.

Compared with crab and fish muscle tissues, mercury appears to bioaccumulate to a lesser degree in clam tissue. Though the data were characterized by a relatively large degree of scatter (e.g., $r^2 = 0.17$), the regression relationship describing clam tissue concentrations as a function of sediment concentrations was nonetheless significant ($P = 0.03$).

6.6 Sediment Bioaccumulation Screening Level

6.6.1 Screening-Level Human Health Risk Assessment

Screening-level risk assessment procedures outlined in MTCA (WAC 173-340-708) were used to estimate a human health benchmark dose and fish/shellfish tissue concentration which is protective of individuals who may consume relatively large amounts of seafood. The screening-level evaluation incorporated conservative exposure and risk assumptions, as follows:

- **Protective Mercury Intake Determined by EPA.** The existing oral reference dose (RfD) for methylmercury used in this assessment was obtained from the U.S. Environmental Protection Agency's (EPA's) Integrated Risk Information System (IRIS) database. The RfD is an estimate of daily methylmercury intake to a population, including sensitive subgroups, which is likely to be without an appreciable risk of deleterious effects during a lifetime. The methylmercury RfD (1×10^{-4} mg/kg-day) was conservatively applied to assess *total* mercury concentrations in fish and shellfish tissues.

- **Crab, Bottomfish, Clams, and Mussels Harvested on Site.** Based on local habitat characteristics (see Section 7.0), it is possible that commercially and recreationally targeted fish and shellfish species including crab, bottomfish, clams, and mussels may be caught in the vicinity of the WW Area. For the purpose of this screening-level assessment, a fisher was conservatively assumed to derive *all* of his/her crab, bottomfish, clam, and mussel intake solely from the WW Area (i.e., 100 percent diet fraction).

- **Salmon and Other Pelagic Species Not Resident.** Although various pelagic fish including salmon may occur near the site, the physical habitat in the WW Area (predominantly soft silt sediments) is generally not preferred by these species. In addition, the relatively extensive home range of salmon further minimizes the potential for mercury bioaccumulation. Consistent with this expectation, salmon returning to the Nooksack River contain low tissue mercury concentrations (average 0.05 mg/kg), and are indistinguishable from regional background (e.g., Skagit River) returns (based on an analysis of PSAMP data presented in Hart Crowser, 1996f).

- **Upper-bound Tribal Fish and Shellfish Consumption Rates Assumed.** The most comprehensive evaluation of seafood consumption rates by regional tribal fishers is contained in Toy et al. (1996), based on studies of the Tulalip and Squaxin Island Tribes of Puget Sound. Mean, upper confidence level (UCL), and upper-bound (90th percentile) seafood consumption rates of Tulalip Tribe fishers, who may more closely match the fishing behavior of the local Lummi and Nooksack Tribes, are summarized in Table 6-2. The conservative upper-bound (90th percentile) combined consumption rate of crab, bottomfish, clams, and mussels is approximately 70 grams per day. This value is more conservative than seafood consumption rates currently used in the state MTCA and federal Superfund cleanup programs (27 grams/day, incorporating a diet fraction of 50 percent). The upper-bound consumption rates summarized in Table 6-2 are also more conservative than values currently being developed for the state SMS program (42 grams/day; L. Weiss, Ecology, personal communication).

6.6.2 Sediment Screening Level Derivation

By mathematically combining the bioaccumulation regression equations and seafood consumption rates summarized above, the sediment screening level associated with maintaining intake at or below the RfD can be calculated. The equations used for this analysis are presented in Table 6-2. The sediment screening level calculated in this manner varied from 1.2 to 3.7 mg/kg, primarily depending on the probability assumption (e.g., mean versus 90th percentile intake). The most conservative scenario evaluated resulted in a screening level of 1.2 mg/kg (90th percentile combined upper-bound consumption; and use of the Dungeness crab regression for bottomfish).

6.7 Wildlife Risk Benchmark Comparison

The sediment screening level derived from the human health-based analysis described above was compared with ecological benchmark concentrations, to ensure protection of wildlife over and above the benthic infaunal risks addressed through sediment bioassays (Section 5.0). The objective of this screening analysis was to evaluate risks to a range of trophic order wildlife receptors in the WW Area from water contact and food chain transfer of mercury derived from sediment.

The EPA Great Lakes Water Quality Initiative (EPA, 1995) developed wildlife criteria for total mercury concentrations in water that are protective of avian and mammalian wildlife populations inhabiting the Great Lakes basin. The criteria address cumulative adverse effects resulting from the ingestion, contact, and food web transfer of mercury in surface waters. The criteria are based on existing toxicological studies and quantitative exposure information for wildlife species. The Great Lakes Criteria were selected for use in this screening-level assessment as no comparable criteria exist for Puget Sound.

The Great Lakes Wildlife Criterion for total mercury in water is 1.3 ng/L, which is the lower of the derived mammalian wildlife criterion (2.4 ng/L) and the avian wildlife criterion (1.3 ng/L). The total mercury concentration measured in inner Bellingham Bay during this RI was 0.98 ng/L (Table 8-4; from low-level mercury sampling in January 1997). Based on this comparison, wildlife risks are not identified, even under existing conditions.

For the purpose of this screening-level analysis, a wildlife benchmark sediment concentration was calculated by multiplying the Great Lakes criterion (1.3 ng/L) by the mean empirical sediment:water partition coefficient determined by this study (1.0×10^6). The resultant benchmark concentration derived from this analysis of 1.3 mg/kg is slightly higher than the value derived from the human health evaluation (1.2 mg/kg). These data suggest that sediment concentrations that are protective of human health will also be

protective of higher trophic level ecological receptors, including sensitive avian species.

6.8 Uncertainty Analysis

Most of the assumptions incorporated into this screening-level mercury bioaccumulation assessment were intentionally conservative. The following summarizes the main sources of uncertainty identified in both the bioaccumulation analysis and risk screening.

Speciation of Mercury. There are three general forms of mercury in the environment: elemental, inorganic, and organic mercury. Elemental mercury is a silver-white, volatile liquid at room temperature. Inorganic mercury compounds include mercuric and mercurous states. Organic mercury compounds are covalently bound to carbon, such as methylmercury ($HgCH_3$) and phenylmercury (HgC_6H_5). For the purposes of this screening-level risk assessment, the toxicity values used were based upon exposure to methylmercury. The form of mercury most likely found in biological tissue is methylmercury; therefore, this assumption will likely not result in a significant overestimate or underestimate of potential risks. Both inorganic and organic forms of mercury are likely to be found in sediments. It is unknown how this factor would influence the calculation of tissue benchmark calculations or the derivations of tissue-specific bioaccumulation relationships.

Toxicity Value for Mercury. The existing oral RfD for methylmercury listed in IRIS was used in the screening-level human health risk assessment. Because methylmercury is the most toxic form of mercury, the methylmercury RfD was conservatively applied to represent total mercury concentrations. The uncertainty associated with these assumptions would most likely result in an overestimation of risks associated with fish/shellfish consumption. Thus, the sediment screening level developed above (1.2 mg/kg) may be too low.

Target Species Home Range. The home range estimates of the target species included in the bioaccumulation evaluation were determined from information presented in the scientific literature and in PSDDA reports. However, the accuracy of these home range estimates is unknown. The home range estimates were used to identify sediment sampling locations to pair with tissue data in the bioaccumulation evaluation. The uncertainty associated with home range estimation could work to both overestimate or underestimate the regression relationships developed for a particular tissue type.

Fish/Shellfish Consumption Rates. Three assumptions were made regarding the estimated fish and shellfish consumption rates. First, an upper-bound (90th percentile) seafood consumption rate for tribal fishers of approximately 70 grams/day was used in the human health screening

analysis. As outlined above, use of this value is likely to result in an overestimate of exposure, particularly for recreational fishers, and thus would conservatively underestimate the human health-based sediment quality criterion. Second, a conservative assumption of 100 percent diet fraction derived from the WW Area was applied to the screening-level evaluation. Again, the uncertainty associated with this parameter would most likely result in an overestimate of exposure, and would conservatively underestimate the appropriate human health-based sediment quality criterion. Finally, the target species considered in the screening-level analysis were limited to crabs, bottomfish, and clams/mussels. Because other species such as salmon are present in the region, the uncertainty associated with this parameter could result in an underestimate of exposure and an overestimate the appropriate human health-based sediment quality criterion. Nevertheless, because of the migratory behavior of salmon, and considering existing tissue quality data for salmon (see above), the uncertainty associated with this factor is likely to be relatively minor.

Fish/Shellfish Tissues Selected for Bioaccumulation Analysis. The bioaccumulation analysis for crab and bottomfish was focused exclusively on muscle tissue, since muscle is the principal seafood tissue consumed by regional fishers. To the extent that other organs such as skin, liver, and the hepatopancreas are also consumed, these bioaccumulation estimates may not be representative of full exposures. However, since mercury concentrations tend to be higher in protein-rich muscle versus other tissues (Nelson et al., 1974; Hart Crowser, 1995), and also because relatively minor amounts of these other tissues tend to be consumed along with muscle, the uncertainty associated with this factor is likely to be relatively minor.

6.9 Conclusions

Figure 6-5 presents an overlay of the extent of surface sediments exceeding the conservative human health-based screening level of 1.2 mg/kg mercury. Surface sediment mercury concentrations exceeding 1.2 mg/kg are restricted to the immediate vicinity of the G-P Log Pond, nearshore areas adjacent to the ASB, and portions of the former Starr Rock disposal area. Further, sediments exceeding this health-based screening level generally fell within those areas of the site also targeted for cleanup to address sediment toxicity concerns identified in Section 5.0. Thus, cleanup necessary to comply with sediment toxicity criteria will only be slightly expanded to address human health and wildlife food web concerns.

Table 6-1 - Paired Sediment and Tissue Mercury Concentration Data, Bellingham Bay and Other Puget Sound Embayments (excluding Sinclair Inlet), 1990 to 1997

Species/Tissue Type & Location	Tissue Sample ID	Measured Mercury Tissue Conc. (Data Source) in mg/kg wet wt.	Home Range Average Sediment Mercury Conc. (Data Source) in mg/kg dry wt.
Dungeness crab muscle (a):			
Bellingham 18	97-31	0.081 (Ecology '97)	0.10 (SEDQUAL)
Bellingham 18	97-32	0.027 (Ecology '97)	0.10 (SEDQUAL)
Bellingham 18	97-33	0.031 (Ecology '97)	0.10 (SEDQUAL)
Chuckanut Bay	90-1	0.060 (Cubbage '91)	0.12 (PSAMP)
Lummi Penninsula	90-2	0.090 (Cubbage '91)	0.20 (SEDQUAL)
Post Point	97-14	0.061 (Ecology '97)	0.23 (CH2MHill '97)
Post Point	97-18	0.077 (Ecology '97)	0.23 (CH2MHill '97)
Central Bellingham Bay	97-52	0.126 (Ecology '97)	0.37 (SEDQUAL)
Central Bellingham Bay	97-54	0.056 (Ecology '97)	0.37 (SEDQUAL)
Post Point	90-4	0.110 (Cubbage '91)	0.39 (SEDQUAL)
Post Point Outfall	90-5	0.080 (Cubbage '91)	0.39 (SEDQUAL)
Georgia-Pacific Outfall	90-7-1	0.120 (Cubbage '91)	0.51 (SEDQUAL)
Georgia-Pacific Outfall	90-7-2	0.060 (SAIC '90)	0.51 (SEDQUAL)
Whatcom Waterway	97-2	0.100 (Ecology '97)	0.54 (WW Area RI)
Whatcom Waterway	97-3	0.119 (Ecology '97)	0.54 (WW Area RI)
Whatcom Waterway	97-22	0.211 (Ecology '97)	0.54 (WW Area RI)
Whatcom Waterway	97-24	0.204 (Ecology '97)	0.54 (WW Area RI)
Whatcom Waterway	97-37	0.100 (Ecology '97)	0.54 (WW Area RI)
Whatcom Waterway	97-38	0.108 (Ecology '97)	0.54 (WW Area RI)
Padden Creek	90-6	0.100 (Cubbage '91)	0.55 (SEDQUAL)
Boulevard Park	90-3	0.100 (Cubbage '91)	0.58 (SEDQUAL)
Whatcom Waterway	90-8-1	0.160 (Cubbage '91)	0.91 (SEDQUAL)
Whatcom Waterway	90-8-2	0.150 (Cubbage '91)	0.91 (SEDQUAL)
Red rock crab muscle (b):			
Port Madison	90-1	0.046 (CH2MHill '91)	0.05 (CH2MHill '91)
Port Madison	90-2	0.062 (CH2MHill '91)	0.05 (CH2MHill '91)
Port Madison	90-3	0.034 (CH2MHill '91)	0.05 (CH2MHill '91)
Port Madison	90-4	0.069 (CH2MHill '91)	0.05 (CH2MHill '91)
Port Madison	90-5	0.103 (CH2MHill '91)	0.05 (CH2MHill '91)
Port Madison	90-6	0.059 (CH2MHill '91)	0.05 (CH2MHill '91)
Port Madison	90-7	0.046 (CH2MHill '91)	0.05 (CH2MHill '91)
Port Madison	90-8	0.223 (CH2MHill '91)	0.05 (CH2MHill '91)
Port Madison	90-9	0.101 (CH2MHill '91)	0.05 (CH2MHill '91)
Port Madison	90-10	0.028 (CH2MHill '91)	0.05 (CH2MHill '91)
Port Madison	90-11	0.014 (CH2MHill '91)	0.05 (CH2MHill '91)
Port Madison	90-12	0.074 (CH2MHill '91)	0.05 (CH2MHill '91)
Port Madison	90-13	0.021 (CH2MHill '91)	0.05 (CH2MHill '91)
West Eagle Harbor	90-1	0.139 (CH2MHill '91)	0.65 (CH2MHill '91)
West Eagle Harbor	90-2	0.043 (CH2MHill '91)	0.65 (CH2MHill '91)
West Eagle Harbor	90-3	0.180 (CH2MHill '91)	0.65 (CH2MHill '91)
West Eagle Harbor	90-4	0.062 (CH2MHill '91)	0.65 (CH2MHill '91)
West Eagle Harbor	90-5	0.251 (CH2MHill '91)	0.65 (CH2MHill '91)
West Eagle Harbor	90-6	0.110 (CH2MHill '91)	0.65 (CH2MHill '91)

Table 6-1 - Paired Sediment and Tissue Mercury Concentration Data, Bellingham Bay and Other Puget Sound Embayments (excluding Sinclair Inlet), 1990 to 1997

Species/Tissue Type & Location	Tissue Sample ID	Measured Mercury Tissue Conc. (Data Source) in mg/kg wet wt.	Home Range Average Sediment Mercury Conc. (Data Source) in mg/kg dry wt.
West Eagle Harbor	90-7	0.078 (CH2MHill '91)	0.65 (CH2MHill '91)
West Eagle Harbor	90-8	0.046 (CH2MHill '91)	0.65 (CH2MHill '91)
West Eagle Harbor	90-9	0.138 (CH2MHill '91)	0.65 (CH2MHill '91)
West Eagle Harbor	90-10	0.098 (CH2MHill '91)	0.65 (CH2MHill '91)
West Eagle Harbor	90-11	0.041 (CH2MHill '91)	0.65 (CH2MHill '91)
West Eagle Harbor	90-12	0.032 (CH2MHill '91)	0.65 (CH2MHill '91)
West Eagle Harbor	90-13	0.057 (CH2MHill '91)	0.65 (CH2MHill '91)
Whatcom Waterway	74-1	0.459 (Nelson et al. '74)	5.94 (Nelson et al. '74)
English sole muscle (c):			
Port Madison	92-3	0.066 (O'Neill et al., '95)	0.05 (CH2MHill '91)
Port Madison	92-2	0.069 (O'Neill et al., '95)	0.05 (CH2MHill '91)
Port Madison	92-1	0.065 (O'Neill et al., '95)	0.05 (CH2MHill '91)
Vendovi Island	94-1	0.074 (O'Neill et al., '95)	0.09 (PSAMP)
Vendovi Island	94-2	0.070 (O'Neill et al., '95)	0.09 (PSAMP)
Vendovi Island	94-3	0.070 (O'Neill et al., '95)	0.09 (PSAMP)
Central Bellingham Bay	91-1	0.091 (O'Neill et al., '95)	0.37 (PSAMP)
Central Bellingham Bay	91-2	0.104 (O'Neill et al., '95)	0.37 (PSAMP)
Central Bellingham Bay	91-3	0.094 (O'Neill et al., '95)	0.37 (PSAMP)
Central Bellingham Bay	92-1A	0.079 (O'Neill et al., '95)	0.37 (PSAMP)
Central Bellingham Bay	92-2A	0.090 (O'Neill et al., '95)	0.37 (PSAMP)
Central Bellingham Bay	92-3A	0.084 (O'Neill et al., '95)	0.37 (PSAMP)
Central Bellingham Bay	93-1	0.086 (O'Neill et al., '95)	0.37 (PSAMP)
Central Bellingham Bay	93-2	0.080 (O'Neill et al., '95)	0.37 (PSAMP)
Central Bellingham Bay	93-3	0.076 (O'Neill et al., '95)	0.37 (PSAMP)
Duwamish River	92-1A	0.075 (O'Neill et al., '95)	0.46 (King Co. '91,'97)
Duwamish River	92-2A	0.079 (O'Neill et al., '95)	0.46 (King Co. '91,'97)
Duwamish River	92-3A	0.082 (O'Neill et al., '95)	0.46 (King Co. '91,'97)
Duwamish River	95-1	0.056 (O'Neill et al., '95)	0.46 (King Co. '91,'97)
Duwamish River	95-2	0.060 (O'Neill et al., '95)	0.46 (King Co. '91,'97)
Duwamish River	95-3	0.065 (O'Neill et al., '95)	0.46 (King Co. '91,'97)
West Eagle Harbor	91-1	0.119 (O'Neill et al., '95)	0.65 (CH2MHill '91/HC '95)
West Eagle Harbor	91-2	0.129 (O'Neill et al., '95)	0.65 (CH2MHill '91/HC '95)
West Eagle Harbor	91-3	0.142 (O'Neill et al., '95)	0.65 (CH2MHill '91/HC '95)
West Eagle Harbor	95-1	0.124 (O'Neill et al., '95)	0.65 (CH2MHill '91/HC '95)
West Eagle Harbor	95-2	0.109 (O'Neill et al., '95)	0.65 (CH2MHill '91/HC '95)
West Eagle Harbor	95-3	0.115 (O'Neill et al., '95)	0.65 (CH2MHill '91/HC '95)
Elliott Bay Waterfront	89-1	0.089 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	89-2	0.058 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	89-3	0.062 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	91-1	0.093 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	91-2	0.080 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	91-3	0.086 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	92-1A	0.092 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	92-2A	0.062 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)

Table 6-1 - Paired Sediment and Tissue Mercury Concentration Data, Bellingham Bay and Other Puget Sound Embayments (excluding Sinclair Inlet), 1990 to 1997

Species/Tissue Type & Location	Tissue Sample ID	Measured Mercury Tissue Conc. (Data Source) in mg/kg wet wt.	Home Range Average Sediment Mercury Conc. (Data Source) in mg/kg dry wt.
Elliott Bay Waterfront	92-3A	0.063 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	93-1	0.083 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	93-2	0.080 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	93-3	0.091 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	94-1	0.088 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	94-2	0.096 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	94-3	0.089 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	95-1	0.074 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	95-2	0.067 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Elliott Bay Waterfront	95-3	0.067 (O'Neill et al., '95)	0.69 (SEDQUAL/HC unpub.)
Composite hardshell clams:			
Eagle Harbor	EH-T-18	0.011 (CH2MHill '91)	0.03 (CH2MHill '91)
Eagle Harbor	EH-T-17	0.013 (CH2MHill '91)	0.04 (CH2MHill '91)
Eagle Harbor	EH-T-1	0.064 (CH2MHill '91)	0.05 (CH2MHill '91)
Eagle Harbor	EH-T-11	0.011 (CH2MHill '91)	0.05 (CH2MHill '91)
Eagle Harbor	EH-T-13	0.020 (CH2MHill '91)	0.05 (CH2MHill '91)
Eagle Harbor	EH-T-15	0.015 (CH2MHill '91)	0.05 (CH2MHill '91)
Eagle Harbor	EH-T-2	0.016 (CH2MHill '91)	0.05 (CH2MHill '91)
Eagle Harbor	EH-T-3	0.022 (CH2MHill '91)	0.05 (CH2MHill '91)
Eagle Harbor	EH-T-4	0.025 (CH2MHill '91)	0.05 (CH2MHill '91)
Eagle Harbor	EH-T-5	0.036 (CH2MHill '91)	0.05 (CH2MHill '91)
Eagle Harbor	EH-T-6	0.069 (CH2MHill '91)	0.05 (CH2MHill '91)
Eagle Harbor	EH-T-9	0.031 (CH2MHill '91)	0.05 (CH2MHill '91)
Semiahmoo	92-1	0.006 (Patrick '96)	0.05 (SEDQUAL)
Semiahmoo	92-2	0.006 (Patrick '96)	0.05 (SEDQUAL)
Semiahmoo	92-3	0.006 (Patrick '96)	0.05 (SEDQUAL)
Semiahmoo	93-1	0.006 (Patrick '96)	0.05 (SEDQUAL)
Semiahmoo	93-2	0.006 (Patrick '96)	0.05 (SEDQUAL)
Semiahmoo	93-3	0.006 (Patrick '96)	0.05 (SEDQUAL)
Sequim Bay	92-1	0.006 (Patrick '96)	0.05 (SEDQUAL)
Sequim Bay	92-2	0.006 (Patrick '96)	0.05 (SEDQUAL)
Sequim Bay	93-1	0.007 (Patrick '96)	0.05 (SEDQUAL)
Sequim Bay	93-2	0.007 (Patrick '96)	0.05 (SEDQUAL)
Sequim Bay	93-3	0.006 (Patrick '96)	0.05 (SEDQUAL)
Eagle Harbor	EH-T-10	0.015 (CH2MHill '91)	0.08 (CH2MHill '91)
Post Point	92-1	0.019 (Patrick '96)	0.39 (SEDQUAL)
Post Point	92-2	0.020 (Patrick '96)	0.39 (SEDQUAL)
Post Point	92-3	0.020 (Patrick '96)	0.39 (SEDQUAL)
Boulevard Park	90-3B	0.010 (Cubbage '91)	0.58 (SEDQUAL)
Eagle Harbor	92-1	0.058 (Patrick '96)	0.77 (CH2MHill '91/HC '95)
Eagle Harbor	92-2	0.056 (Patrick '96)	0.77 (CH2MHill '91/HC '95)
Eagle Harbor	92-3	0.060 (Patrick '96)	0.77 (CH2MHill '91/HC '95)
Eagle Harbor	93-1	0.081 (Patrick '96)	0.77 (CH2MHill '91/HC '95)
Eagle Harbor	93-2	0.075 (Patrick '96)	0.77 (CH2MHill '91/HC '95)

Table 6-1 - Paired Sediment and Tissue Mercury Concentration Data, Bellingham Bay and Other Puget Sound Embayments (excluding Sinclair Inlet), 1990 to 1997

Species/Tissue Type & Location	Tissue Sample ID	Measured Mercury Tissue Conc. (Data Source) in mg/kg wet wt.	Home Range Average Sediment Mercury Conc. (Data Source) in mg/kg dry wt.
Eagle Harbor	93-3	0.074 (Patrick '96)	0.77 (CH2MHill '91/HC '95)
Eagle Harbor	EH-T-19	0.055 (CH2MHill '91)	0.77 (CH2MHill '91/HC '95)
Eagle Harbor	EH-T-20	0.159 (CH2MHill '91)	1.30 (CH2MHill '91/HC '95)
Eagle Harbor	EH-T-8	0.091 (CH2MHill '91)	2.85 (CH2MHill '91/HC '95)
Eagle Harbor	EH-T-7	0.091 (CH2MHill '91)	12.44 (CH2MHill '91/HC '95)

NOTES:

a) Legal adult male Dungeness crabs only (greater than 160 mm carapace width)

b) Large adult male Red rock crabs only (greater than 130 mm carapace width)

c) English sole muscle tissue concentration adjusted to reflect Year-8 individuals (see Figure 6-3 and text).

Table 6-2 - Derivation of Bioaccumulation-Based Sediment Mercury Cleanup Screening Levels

Bioacumulation Regression Data (a):	Number of Sample Composites	(y) y-intercept mg/kg wet	(s) slope dry/wet	Adj. r^2	P
1. Legal Dungeness crab muscle only	12	0.047	0.116	0.73	0.0002
1a. Red rock crab muscle only (b)	3	0.060	0.067	N/A	N/A
2. English sole muscle only (8-year-old fish)	15	0.070	0.027	0.04	0.2
3. Clams and Mussels only	25	0.032	0.007	0.17	0.03

a) Excluding Sinclair Inlet data, since slope estimates for Sinclair Inlet were significantly lower than other Puget Sound embayments.

b) Since the Rock crab statistics were based on few data points and were less conservative than the Dungeness crab only regression, and because of relatively low Rock crab consumption rates, Rock crab data were excluded from further bioaccumulation analyses.

Tulalip Tribe Seafood Consumption Data (c):	Consumption Rate in gms/day (c) n	Mean	UCL (d)	90%-tile
1. Dungeness crab	73	12.0	19.3	23.4
1a. Red rock crab	73	0.1	0.4	0.0
2. Total Bottomfish	73	2.3	3.2	7.8
3. Clams and mussels	73	14.4	21.8	38.5

c) Consumption rate normalized to a 70-kg adult. From Toy et al. (1996) and Pollisar, written communication (1997).

d) 95% upper confidence interval of the mean

Sediment Mercury Screening Levels Calculated for Different Consumption Scenarios (e)	Sediment Screening Levels in mg/kg dry weight Mean	UCL (d)	90%-tile
1. Crab consumption only	4.6	2.7	2.2
2. Bottomfish (8-year-old fish) consumption only	108	77	30
3. Clam and mussel consumption only	68	43	22
Total crab, bottomfish, and clams/mussels combined (f):			
Using tissue-specific regression equations	3.7	2.1	1.3
Substituting Dungeness crab for bottomfish regression	3.3	1.9	**1.2**

e) Sediment cleanup screening levels for bioaccumulation protection were calculated for different tribal consumption rates, to maintain total intake levels below the oral reference dose for methylmercury of 1×10^{-4} mg/kg-day.

f) Conservatively estimated assuming complete interdependence between crab, bottomfish, and clam/mussel consumption rates, using the following equations:

$$Intake_1 + Intake_2 + Intake_3 = 1 \times 10^{-4} \text{ mg/kg-day}$$

$$Intake_1 = c_1 \times (y_1 + s_1X) \times Z$$

$$Intake_2 = c_2 \times (y_2 + s_2X) \times Z$$

$$Intake_3 = c_3 \times (y_3 + s_3X) \times Z$$

where Intake = total mercury intake in mg/kg-day, and

c = tissue-specific consumption rate in gms wet weight/day

y = y-intercept from bioaccumulation regression in mg/kg wet weight

s = slope from bioaccumulation regression in dry/wet weight

X = sediment concentration in mg/kg dry weight

Z = proportionality constant (normalized to a 70-kg adult)

Subscripts 1, 2, and 3 denote crab, bottomfish, and clam/mussel tissues, respectively.

Since all other values were known, the above equation was then solved for X, the sediment cleanup screening level.

1.2 - bolded value denotes the sediment cleanup screening level conservatively calculated using 90%-tile consumption rates.

English Sole Muscle Tissue Mercury Concentration Increases with Fish Age Puget Sound Embayments with Elevated Sediment Mercury Concentration

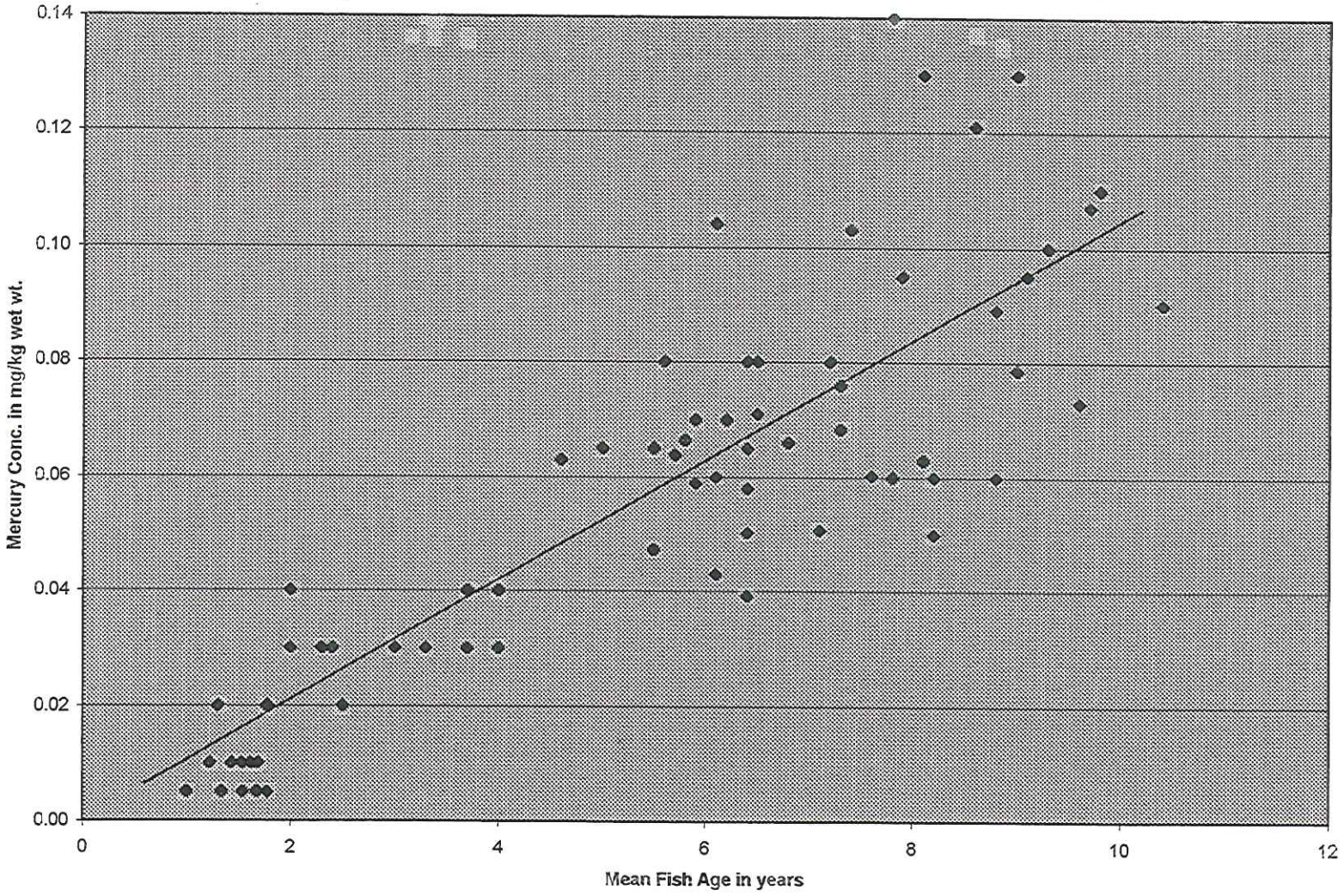

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Figure 6-3

Crab and Bottomfish Muscle Tissue Mercury Concentrations Correlation with Sediment Level

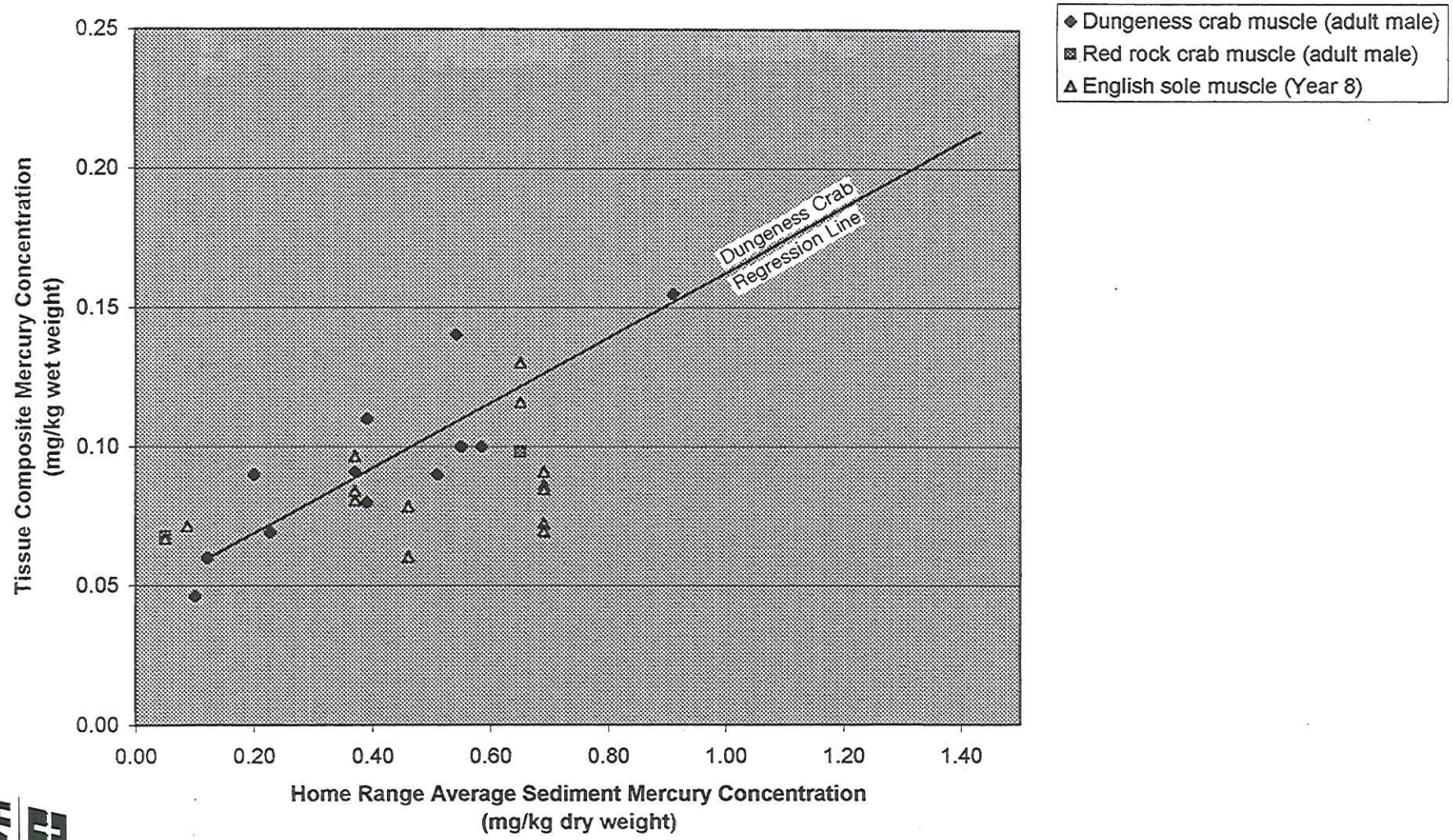

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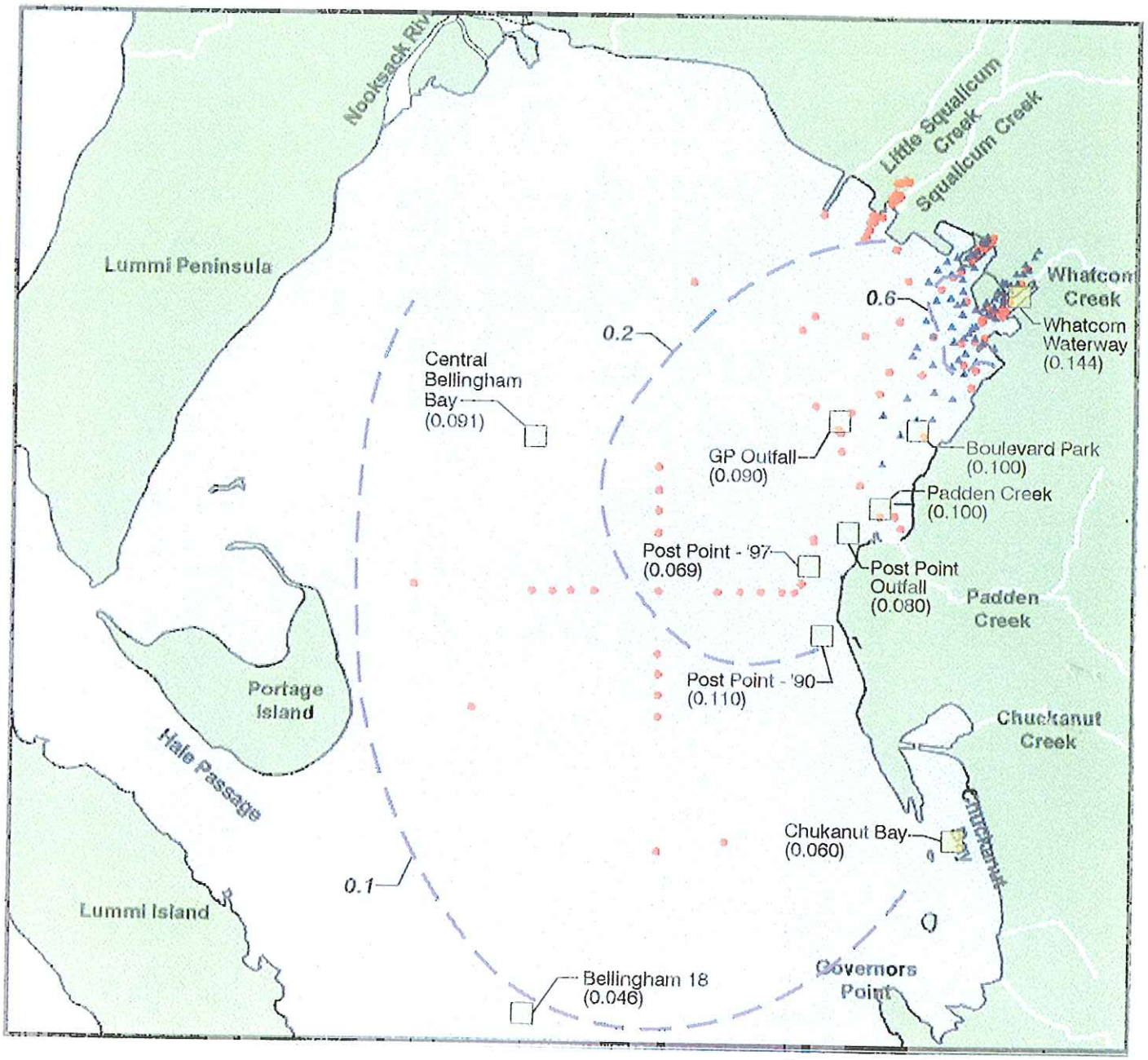
Figure 6-4

Mercury Concentrations in Adult Male Dungeness Crab Muscle Tissue
1990 - 1997



0.1 Sediment Mercury Concentration Contour in mg/kg

Other Sediment Sampling Sites

RI Sediment Sampling Sites

Bellingham 18 Crab Tissue Sampling Location

(0.046) Average Mercury Concentration in mg/kg

Note: Base map prepared from figure developed by Parametrix, Inc. entitled "Sediment Sampling Sites", undated.

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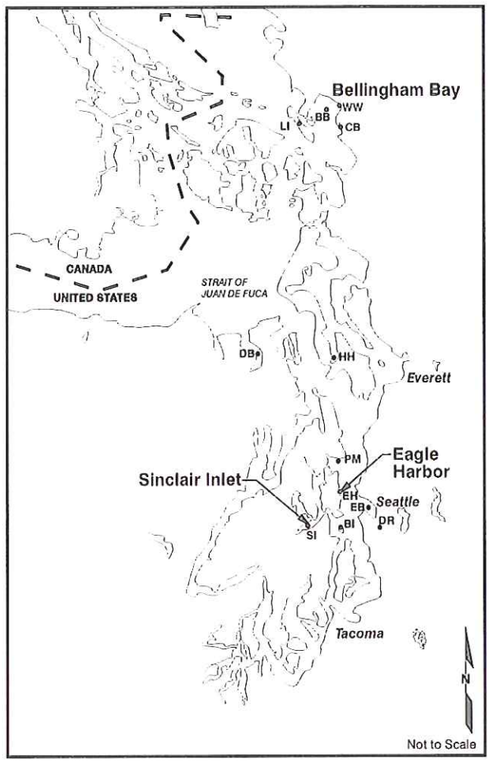
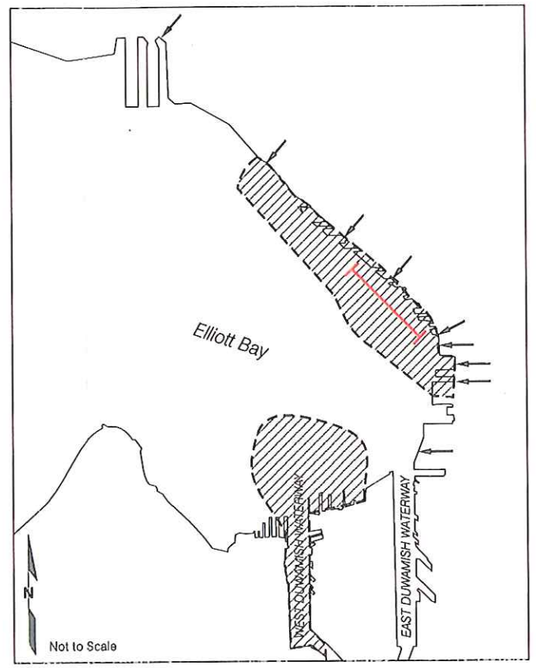
Figure 6-1

Puget Sound Tissue Sample Location Map



BB	Bellingham Bay
BI	Blake Island
DB	Discovery Bay
DR	Duwamish River
EB	Elliott Bay
EH	Eagle Harbor
CB	Chuckenut Bay
HH	Holmes Harbor
LI	Lumni Island
PM	Sinclair Inlet
SI	Whatcom Waterway
WW	

Elliott Bay

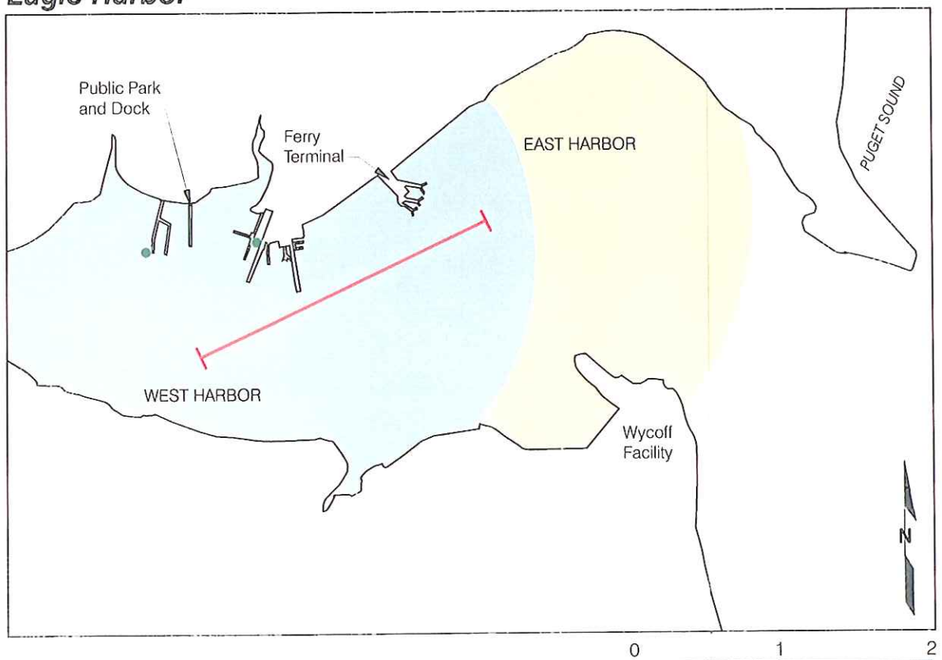


English Sole Trawls (PSAMP, 1996)

Probable Extent of Sediments Exceeding 0.59 mg/kg (Dry Weight) Mercury, based on kriging of available data

Combined Sewer Overflow and Historical Outfall Locations

Eagle Harbor



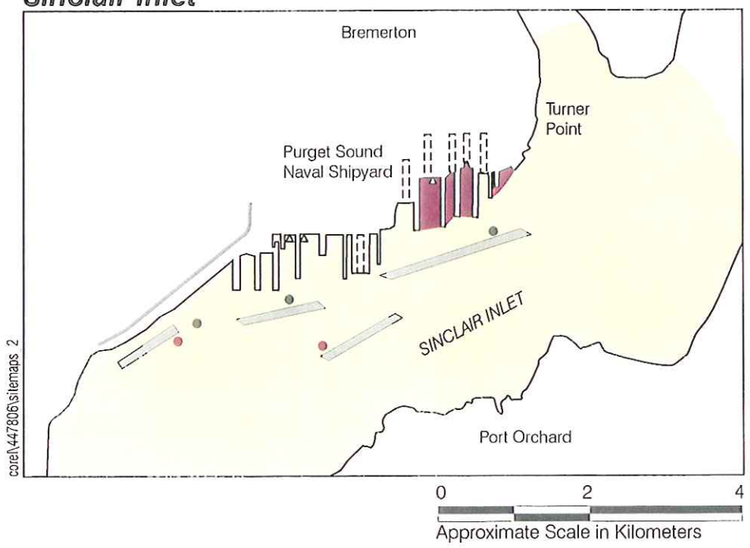
English Sole Trawls (PSAMP, 1996)

Red Rock Crab (CH_2M-Hill, 1991)

Red Rock Crab Home Range

English Sole Home Range

Sinclair Inlet



Blue Mussel (URS, 1996)

Sea Cucumber (URS, 1996)

English Sole Home Range

English Sole Trawls (URS, 1996)

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Figure 6-2

Extent of Sediments Exceeding Bioaccumulation Screening Level

Whatcom Waterway Site

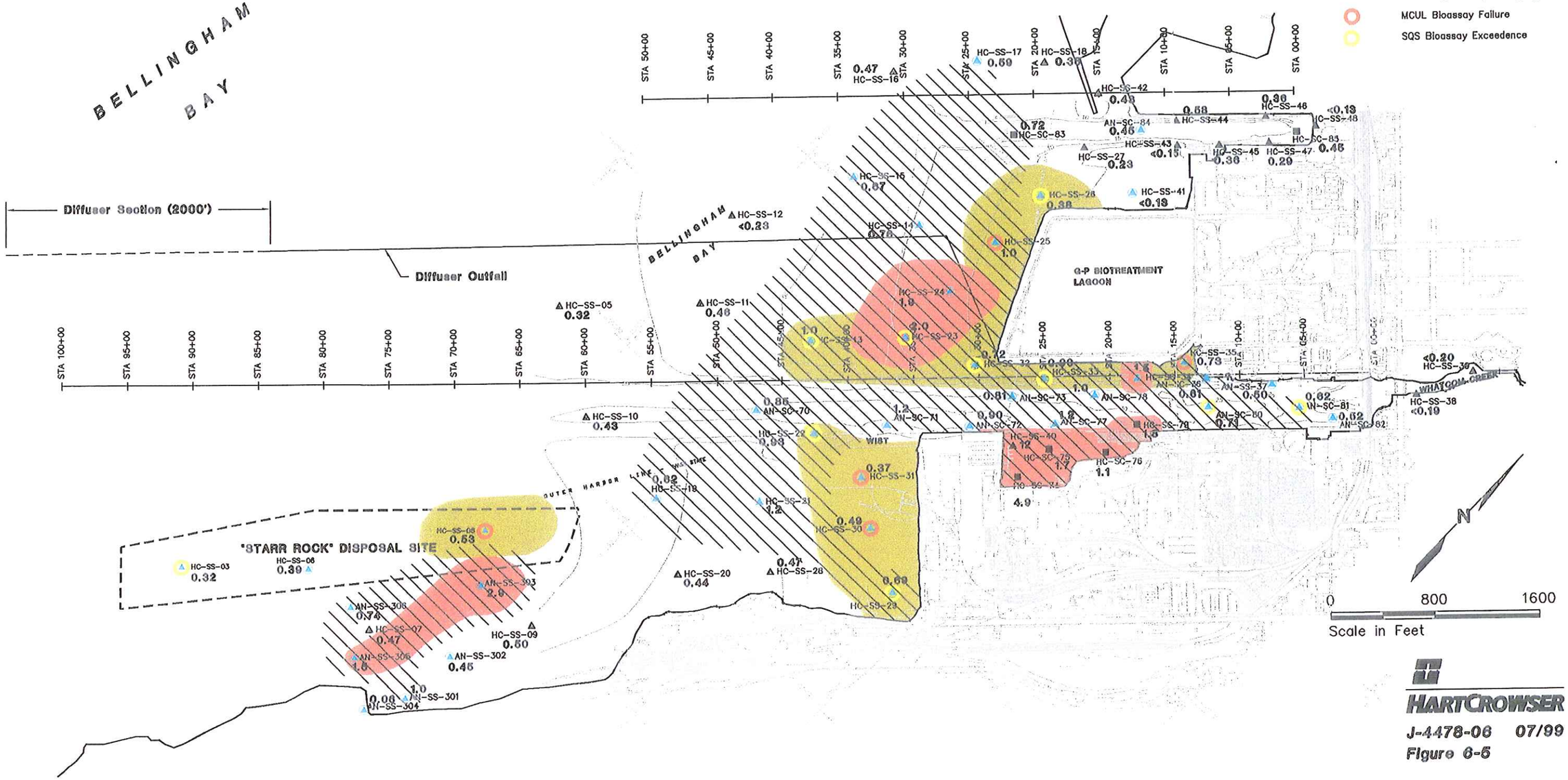

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Figure 6-5

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7.0 NATURAL RESOURCES IN BELLINGHAM BAY

7.1 Local and Regional Habitats

Natural marine habitats in Bellingham Bay include rock, gravel, sand-gravel, sand, and mudflat systems. A detailed shoreline characterization of inner Bellingham Bay is presented on Figure 3-6. Shoreline topography ranges from steep rock faces to sand and mudflats. Within inner Bellingham Bay, the intertidal areas have been modified by soil fill, riprap, bulkheads, and artificial lagoons.

Along the eastern and western shorelines of Bellingham Bay, beaches are typically narrow and steep. Shorelines often consist of rock outcroppings alternating with pockets of sandy and gravelly sediments. Intertidal beaches are generally narrow and composed of sandy to gravelly sediments where bluffs of unconsolidated material front the water. Intertidal sand and mudflats formed by the Nooksack River delta occur at the head of the bay. East and south of the delta, the shoreline is composed of cobble and finer materials.

City and county zoning designations, the City urban growth boundary, and City neighborhood areas within Bellingham Bay are presented on Figure 7-1. Surrounding the project area is a mix of public (Port, City, state, and federal) and private ownership. The City of Bellingham covers an area of about 16,000 acres and includes a population of approximately 58,000. The WW Area is located within and adjacent to the Central Business Districts of Bellingham. Over the last century, the inner waterfront in the project area has been used primarily for industrial and commercial purposes, including sawmills, shipping terminals, pulp, paper, and chemical facilities, cargo docks, boat repair, municipal landfills, bulk oil storage, concrete product manufacturing, fish processing, metal foundry, and municipal storm and sewer drainage. The shoreline of the project area is zoned by the City of Bellingham as Urban Maritime.

7.2 Marine Habitat

The marine habitat within greater Bellingham Bay, including the project area, can be described as diverse and varied in composition (Nelson et al., 1974; CH2M Hill, 1976; Webber, 1978; Shea et al., 1981; Broad et al., 1984; Becker et al., 1989; and Palm, 1995). This diversity has been attributed to variations in oceanographic conditions such as salinity and temperature within the bay, water depth, sediment characteristics, seasonality, land use, and anthropogenic influences. The northern and eastern shorelines of Bellingham Bay serve as fishery nursery areas for juvenile marine fish and shellfish including Starry flounder *(Platichtys stellatus)* and English sole *(Pleuronectes*

vetulus), juvenile Dungeness crab *(Cancer magister)*, and outmigrating juvenile salmon *(Oncorhynchus* sp.) from the area rivers and streams.

7.2.1 Plankton

Phytoplankton. Twenty genera of phytoplankton have been identified in Bellingham Bay (CH2M Hill, 1984). Diatoms comprised of both pelagic and sessile forms dominate the phytoplankton population of the bay. Phytoplankton densities are usually relatively low from January through March and increase rapidly from April to a peak in June. Tollefson (1962) showed that phytoplankton primary productivity within the inner bay was near 1.4 mg chlorophyll/m^3-hr. This productivity rate is lower than values typical for greater Bellingham Bay.

Zooplankton. Zooplankton communities have been found to be diverse within the bay. Fifteen major forms of pelagic zooplankton have been identified within Bellingham Bay (CH2M Hill, 1984). The dominant pelagic zooplankton forms are copepods, copepod nauplii, and tunicates *(Appendicularia)*. Annelids, cladocera, barnacle nauplii, flatworms, and echinoderm larvae are also known to be present. Within the inner bay, zooplankton densities are typically higher at depth (>20 feet) than at the water surface.

Ichthyoplankton. Research conducted by the U.S. DOI (1967) reported a relatively large number of English sole eggs in the surface waters of Bellingham Bay during the peak of the reproductive cycle.

7.2.2 Macrophytes

Eelgrass. The location of existing and historical eelgrass meadows is shown on Figure 7-2. Shoreline surveys extending from the Nooksack River delta to Post Point located eelgrass *(Zostera japonica* and *Zostera marina)* patches and meadows (Palm, 1995; and Thom and Mellum, 1990). Historical data indicate a large eelgrass meadow located on the former delta of Whatcom Creek. The area of this meadow has been significantly reduced as the result of shoreline modification and dredging in the inner bay. Another historical eelgrass meadow *(Z. marina)* is located off of Lummi Island. Presently, colonizing meadows of eelgrass *(Z. japonica)* are located both to the east and west of the Nooksack River delta as shown on Figure 7-2. Area-limited eelgrass meadows are also found along the eastern shoreline of Bellingham Bay near Boulevard Park, Taylor Street Dock, Padden Creek, and at Post Point. Eelgrass patches and small meadows were identified to the west of Little Squalicum Park Reserve, within the Squalicum Creek Waterway, between the I&J Street Waterway and the G-P Aerated Stabilization Basin, and along the eastern shoreline of the bay from north of the Cornwall Avenue Landfill to south of Post Point. The general distribution of eelgrass meadows

and patches within the WW Area is presented on Figure 3-6. More recent detailed maps of eelgrass distribution in inner Bellingham Bay are included in the Bellingham Bay Comprehensive Strategy EIS.

Kelp. Because Bellingham Bay is composed primarily of unconsolidated sediments, there are no significant kelp beds identified within the bay (Sternberg, 1967).

7.2.3 Benthic Invertebrates

Several studies of benthic invertebrates conducted before 1979 reported that polychaetes dominated the benthic assemblage in both abundance and diversity. Bivalves were found to be the second most diverse group (Nelson et al., 1974).

Studies conducted after 1979 found the species composition of the benthic assemblages to be generally similar to those found in earlier surveys of Bellingham Bay, though abundance appeared to have increased. Broad et al. (1984) identified four relatively discrete benthic assemblages in Bellingham Bay. One assemblage, found near the Nooksack River, was dominated by the polychaete *Owenia fusiformis*. A second assemblage was found in the inner bay. It was dominated by the polychaete *Tharyx* sp. A third assemblage was found in the outer section of inner Bellingham Bay. It was also dominated by *Tharyx* sp., but exhibited higher abundance and total biomass than the inner section. A fourth assemblage was found in the outer portions of Bellingham Bay. It was dominated by the bivalve mollusc *Axinopsida serricata*.

PTI (1989) performed a preliminary analysis of the available benthic macroinvertebrate data to evaluate differences in abundance of major taxa within inner Bellingham Bay, relative to mean abundance observed at reference locations in outer Bellingham Bay and Samish Bay. The abundance of amphipods and other crustaceans was significantly lower than that of the reference areas at three sampling locations within and immediately adjacent to the mouth of the Whatcom Waterway (between HC-SS-24 and HC-SS-31; Figure 2-2). Reduced amphipod and other crustacean abundance were also noted at other locations within Bellingham Bay (e.g., near the Post Point and G-P wastewater outfalls). One location near HC-SS-31 (Figure 2-2) had reduced mollusc abundance, and none of the locations in inner Bellingham Bay exhibited reduced polychaete abundance. PTI recommended that additional sediment toxicity bioassays, similar to those described in Section 5.0, should be performed to more fully assess possible toxicity to benthic macroinvertebrates in inner Bellingham Bay, as indicated by their preliminary analysis of the abundance data. The interested reader is referred to the PTI document for a more complete description of the benthic macroinvertebrate analysis.

7.2.4 Shellfish

Several species of shellfish are known to occur within intertidal areas of Bellingham Bay. Webber (1975) observed that the diversity and density of shellfish within the bay vary and are closely related to salinity values. Relatively low shellfish abundance was observed in the western portion of Bellingham Bay influenced by the Nooksack River. Shellfish densities were also relatively low at beaches within Bellingham City limits from Post Point to the northern boundary.

Clam. Clams, including *Macoma* and *Mya* were the most common genera found in inner Bellingham Bay. Clam densities increased in the southern portion of the bay, along the eastern margin.

Oyster. Historically, few Pacific oysters *(Crassostrea gigas)* and no native oysters *(Ostrea lurida)* have been reported in Bellingham Bay (Webber, 1975), presumably because of relatively cold water temperatures, although some populations of moderate density have been observed in the inner bay (Webber, personal communication, 1997). Pacific oysters, an introduced species, are grown commercially and are only abundant where they are cultured.

Crab. A variety of crab species, including Dungeness crab, Red rock crab, and *Crangon* sp. have been observed within the study area (Nelson et al., 1974; Cubbage, 1991). Dungeness crab *(Cancer magister)* are relatively abundant in Bellingham Bay and are the dominant crab species. The distribution of Dungeness crab density in Bellingham Bay is shown on Figure 7-3. The occurrence of Dungeness crab is highest along the shoreline of Post Point and eastern Portage Island. Crab density is also high in the southern-central portion of Bellingham Bay.

Although relatively few data are available on Dungeness crab densities in inner Bellingham Bay, sampling within this area was recently performed by Western Washington University (WWU; funded by G-P). Results of the WWU sampling indicate that the WW Area may contain low to moderate densities of Dungeness crab, relative to abundance within other areas of the bay (see Figure 7-3; Behr, 1998).

Rock crab *(Cancer productus* and *Cancer gracilis)* and Tanner crab *(Chionoecetes bairdi)* are also found within the bay (Dinnel et al., 1988). Rock crab density distribution for Bellingham Bay is shown on Figure 7-4. Highest occurrences of Rock crab are found in the western-central portion of Bellingham Bay and along Post Point. Rock crab density is also high in the central part of Bellingham Bay. Recent WWU sampling results for inner Bellingham Bay indicate relatively moderate abundance of Rock crab within these nearshore areas (Behr, 1998).

Shrimp. Shrimp have been reported in most areas of Bellingham Bay, but are most commonly observed in the deeper portions of the bay. Pandalid species were found to be the dominant shrimp, and all seven species of the pandalid shrimp were found within the bay. The density distribution for Pandalid shrimp in Bellingham Bay is presented on Figure 7-5. Highest occurrences of Pandalid shrimp are known to occur off Post Point and in south-central Bellingham Bay. Bellingham Bay was found to be especially rich in *P. hypsinotus*, *P. danae*, and *P. borealis* (Dinnel et al., 1988). Recent WWU sampling results for inner Bellingham Bay indicate relatively low abundance of Pandalid shrimp in this area (Behr, 1998).

7.2.5 Fish

Comparison of historical and recent fish community surveys in Bellingham Bay indicates a general similarity through time in the composition of dominant species. Of the 80 species common to the major surveys conducted in the bay, 19 species have occurred in all studies and 12 additional species occurred in three of the four studies. Becker et al. (1989) identified the following as important marine fishes in Bellingham Bay: Pacific herring *(Clupea harengus)*, Pacific cod *(Gadus macrocephalus)*, various rockfishes *(Sebastes* sp.), Lingcod *(Ophiodon elongatus)*, Rock sole *(Lepidopsetta bilineata)*, English sole *(Platichtys vetulus)*, and Starry flounder *(Platichthys stellatus)*. Flathead sole *(Hippoglossoides elassodon)*, Butter sole *(Lopsetta isolepis)*, and Longfin smelt *(Spirinchus thaleichthys)* have also been identified as significant species (Dinnel et al., 1988). Reports based on surveys of inner Bellingham Bay indicate a high species richness in the outer Whatcom Waterway (Shea et al., 1981).

Whatcom Creek is known to provide spawning grounds for chum salmon in intermittent stream sections. However, a wide range of salmonids, including coho, chum, chinook, pink, sockeye, steelhead, cutthroat, and Dolly varden have been observed within Whatcom Creek (BBWG, 1998).

Much of the information on juvenile salmon migration routes, schooling and avoidance areas, and length of juvenile residence time is based on historical studies in Bellingham Bay (BBWG, 1998), and support the following conclusions:

- The peak of juvenile salmon migration occurs in May and June.
- A substantial portion of the migration is distributed in offshore waters of Bellingham Bay during May. Schools of migrating juvenile salmon have been observed along all shoreline beach areas including the urbanized portion of Bellingham Bay, but the manner of migration to the shoreline has not been determined.

- Juvenile coho and chinook salmon appear to have different migration habits. Coho appear to remain in the bay for about two weeks longer than chinook, which typically move out of the bay in a 20-day period. Both species have been caught in shallow waters throughout the Bay. Chinook salmon have been caught within the navigable portion of the Whatcom Waterway as well as outside of the channel.

General life history information on these economically important species is described in more detail in BBWG (1998).

7.2.6 Marine Mammals

Harbor seal *(Phoca vitulina richardsi)*, Killer whale *(Orcinus orca)*, Gray whale *(Eschrichtius robustus)*, and Harbor porpoise *(Phocoena phocoena)* have been reported within Bellingham Bay (Shea et al., 1981).

7.2.7 Marine Birds

There are a reported 87 known species of birds known to exist in the vicinity of Bellingham Bay (Shea et al., 1981). Major available bird habitats include: protected harbors, tideflats, estuaries, marshes, undeveloped sandy beaches, rock islands, and man-made structures. Gulls, grebes, cormorants, loons, terns, and murres are the most common birds over the open water areas of the bay.

7.3 Fisheries

7.3.1 Commercial Fisheries

Commercial fisheries for Pacific salmon, bottom fish, and shellfish occur in Bellingham Bay. These harvests are usually obtained using gill nets, purse seines, bottom trawls, and crab pots. In addition, the usual and accustomed fishing areas for both the Lummi Nation and Nooksack Tribe encompass all of Bellingham Bay.

Pacific salmon fisheries including chinook, coho, and chum salmon fisheries are the most lucrative fisheries in Bellingham Bay. Although there are no fisheries for pink and sockeye salmon, these species are incidentally caught in the bay. Sockeye salmon are also caught incidentally in the Nooksack River fisheries (CH2M Hill, 1984).

The commercial trawl or drag fishery in Bellingham Bay is composed of three categories based on the final product use of resources: food fish, reduction, and animal food. The primary category has been food fish. The Starry

flounder has been the major species in the trawl and drag fishery (CH2M Hill, 1984).

Currently, the major commercial shellfish fishery in Bellingham Bay is for Dungeness crab. Other species harvested include: native littleneck, butter, manilla, and quohog clams; Pacific oysters; octopus; and Rock crab. Octopus are caught incidentally in the trawl fishery, and Rock crab are caught incidentally in the Dungeness crab fishery (CH2M Hill, 1984).

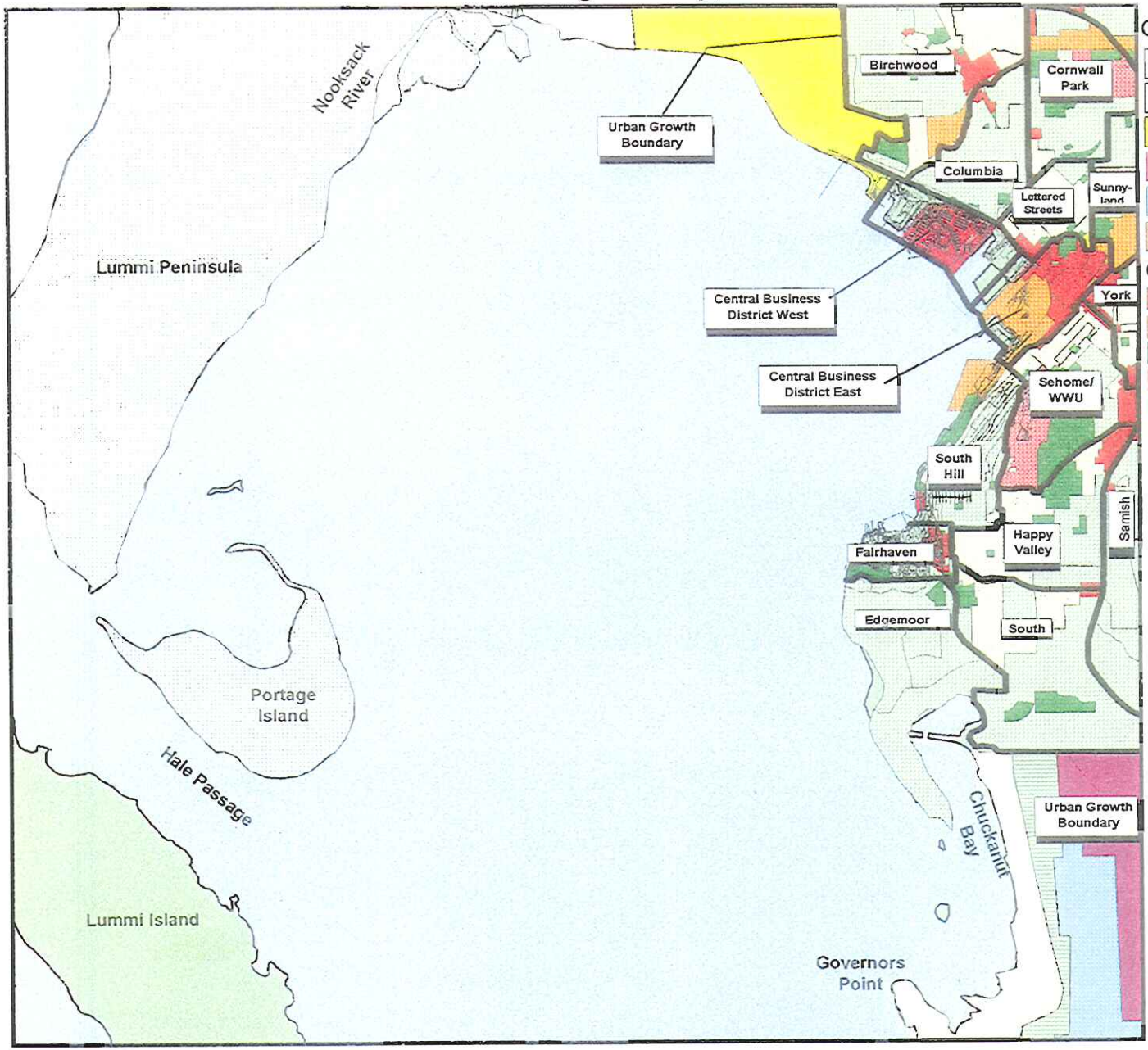
Marine aquaculture activities occur in the area of the Squalicum Harbor Marina, Portage Bay, Hale Passage, Lummi Bay, Samish Bay, and at Point Francis. In addition, the Washington State Department of Health (WSDOH), Office of Shellfish Programs is responsible for classifying actual and potential shellfish growing/harvesting waters in Washington State as approved, conditionally approved, restricted, or prohibited. The primary responsibility of the WSDOH is to ensure that shellfish grown and harvested in the state and consumed are safe for human consumption. They classify waters based on sanitary surveys, which include water quality and shoreline investigations.

Figure 7-6 shows the current status of certification for shellfish harvests in Bellingham Bay. Portions of inner Bellingham Bay, including the project area, are under advisory against shellfish and/or bottom fish harvest, due to fecal coliform and other concerns. The existing advisory against fish and shellfish harvest in inner Bellingham Bay was not specifically based on tissue quality data, but instead relied on available water quality sampling information that indicated a potential for fecal contamination from the water column in the inner bay. WSDOH also decertified a portion of the commercial harvest in the Portage Island area due to high fecal coliform counts in routine samples (BBWG, 1998). WSDOH is considering a proposed restriction to commercial shellfish harvest north of Portage Island and east of Lummi Peninsula. The WSDOH is also considering approving an area for recreational shellfish harvest north of the spit on Portage Island.

7.3.2 Recreational Fisheries

Except for the northwest corner of Bellingham Bay, the bay is considered part of the recreational fishery. Targeted species include: salmon, bottomfish, rockfish, smelt, crab, and clams. Figure 7-6 shows the location of public shellfishing beaches in the Bellingham Bay region. Chinook salmon is the most sought after species by sport fishermen (CH2M Hill, 1984).

Bellingham Bay Demonstration Pilot
City and County Zoning
Bellingham Bay, WA.



County Zoning
- Lummi Reservation
- Suburban Enclaves
- Urban Growth Enclaves
- Rural
- Public/Recreation
- Rural Forestry

City Zoning
- Commercial/Commercial Marine
- Industrial
- Industrial/Commercial
- Institutional
- Public
- Residential/Multi-Family
- Residential/Residential Single-Family
- Birchwood Neighborhood Designations

BELLINGHAM BAY DEMONSTRATION PILOT

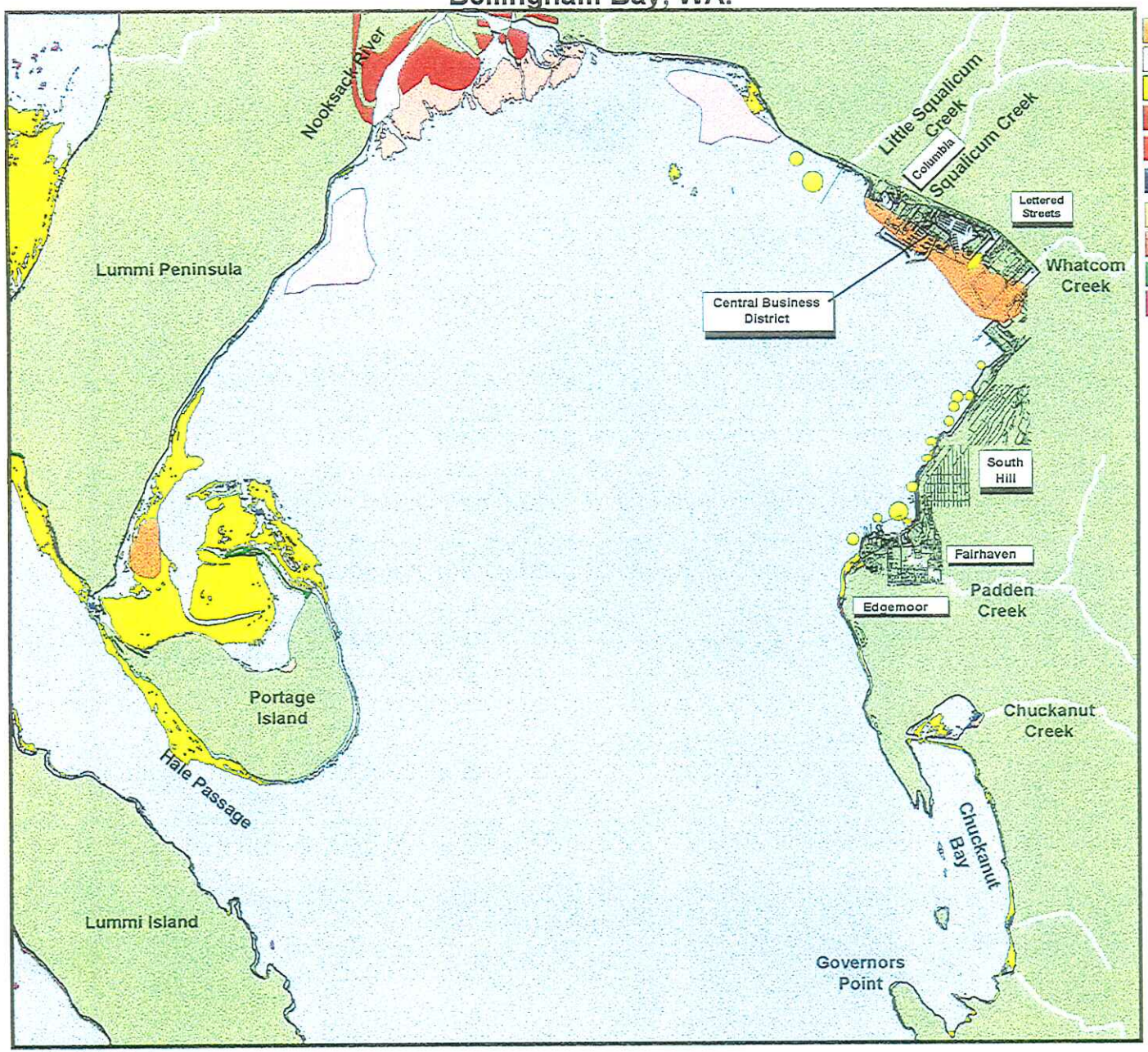
N

5000 0 5000 Feet

June 23, 1998 - Final

J-4478-06 Figure 7-1

Bellingham Bay Demonstration Pilot
Intertidal and Shallow Subtidal Macrofauna, Kelp, and Eelgrass
Bellingham Bay, WA.



- Historical/*Z. marina*
- *Z. japonica*
- *Z. marina*
- 1993 Saltmarsh
- brown algae
- green algae
- mixed algae
- saltmarsh
- spit and berm community
- kelp

BELLINGHAM BAY DEMONSTRATION PILOT

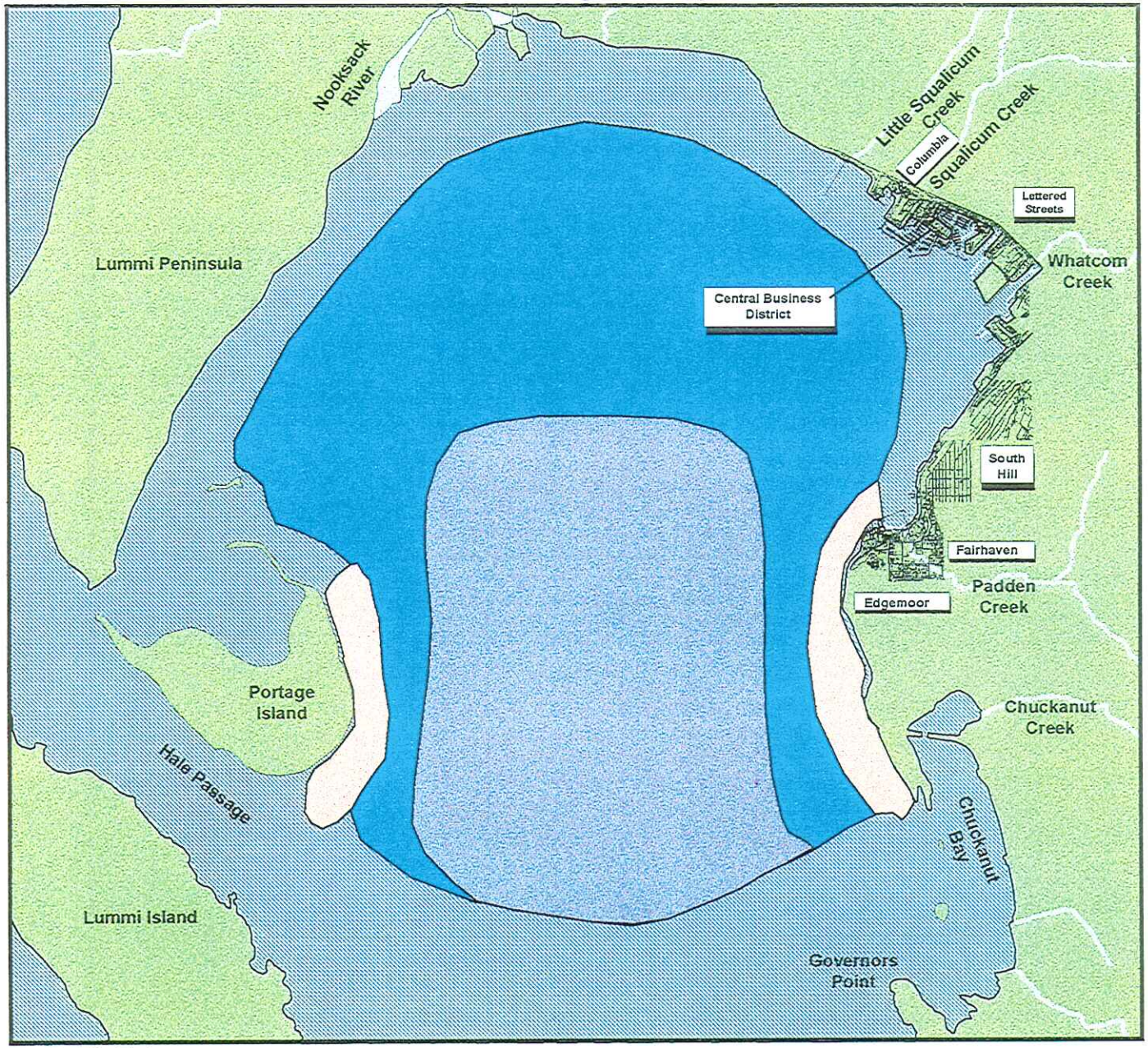
N

5000 0 5000 Feet

June 23, 1998 - Final

J-4478-06 Figure 7-2

Bellingham Bay Demonstration Pilot
Dungeness Crab Densities
Bellingham Bay, WA.



Crab Density (crabs/ha)

- < 50
- 51-150
- >150
- Unknown density

BELLINGHAM BAY DEMONSTRATION PILOT

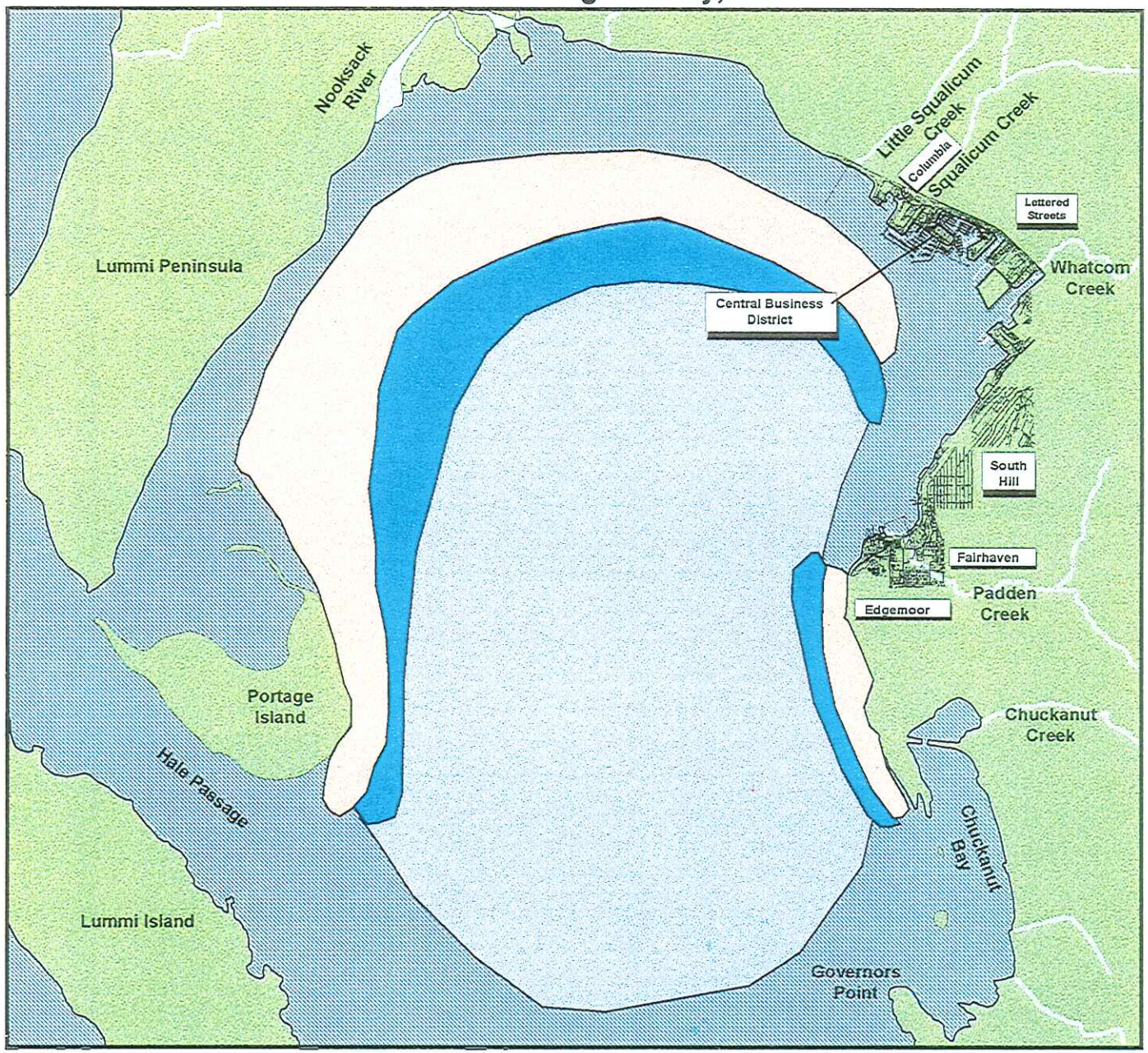
N

5000 0 5000 Feet

June 23, 1998 - Final

J-4478-06 Figure 7-3

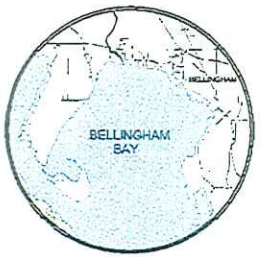
Bellingham Bay Demonstration Pilot
Rock Crab Densities
Bellingham Bay, WA.



Rock Crab Densities

- <50
- 51-100
- >100
- Unknown density

BELLINGHAM BAY DEMONSTRATION PILOT



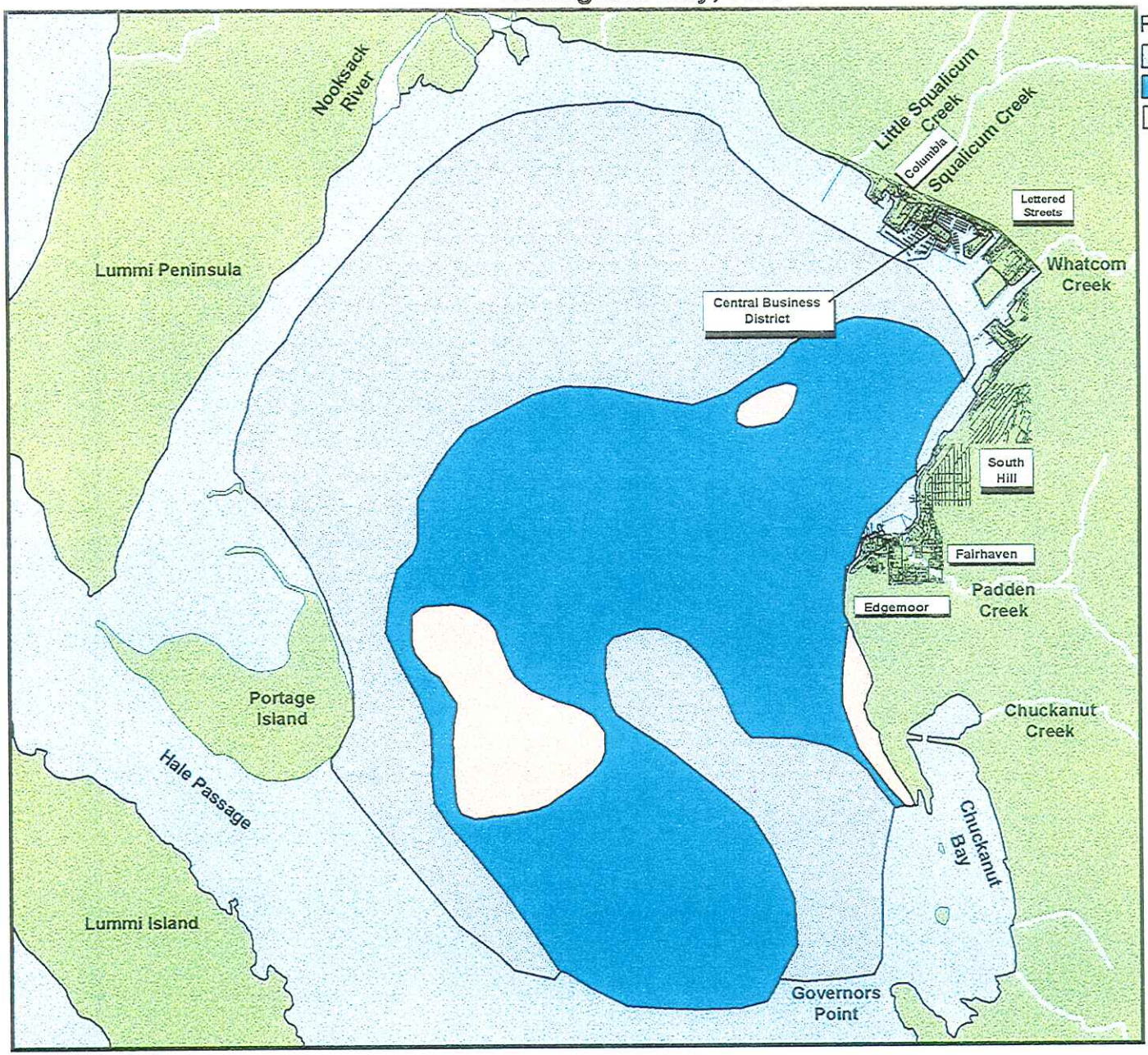
N

5000 0 5000 Feet

June 23, 1998 - Final

J-4478-06 Figure 7-4

Bellingham Bay Demonstration Pilot
Pandalid Shrimp Densities
Bellingham Bay, WA.



Pandalid Shrimp Densities
- <500
- 501-1000
- >1000

BELLINGHAM BAY
DEMONSTRATION PILOT

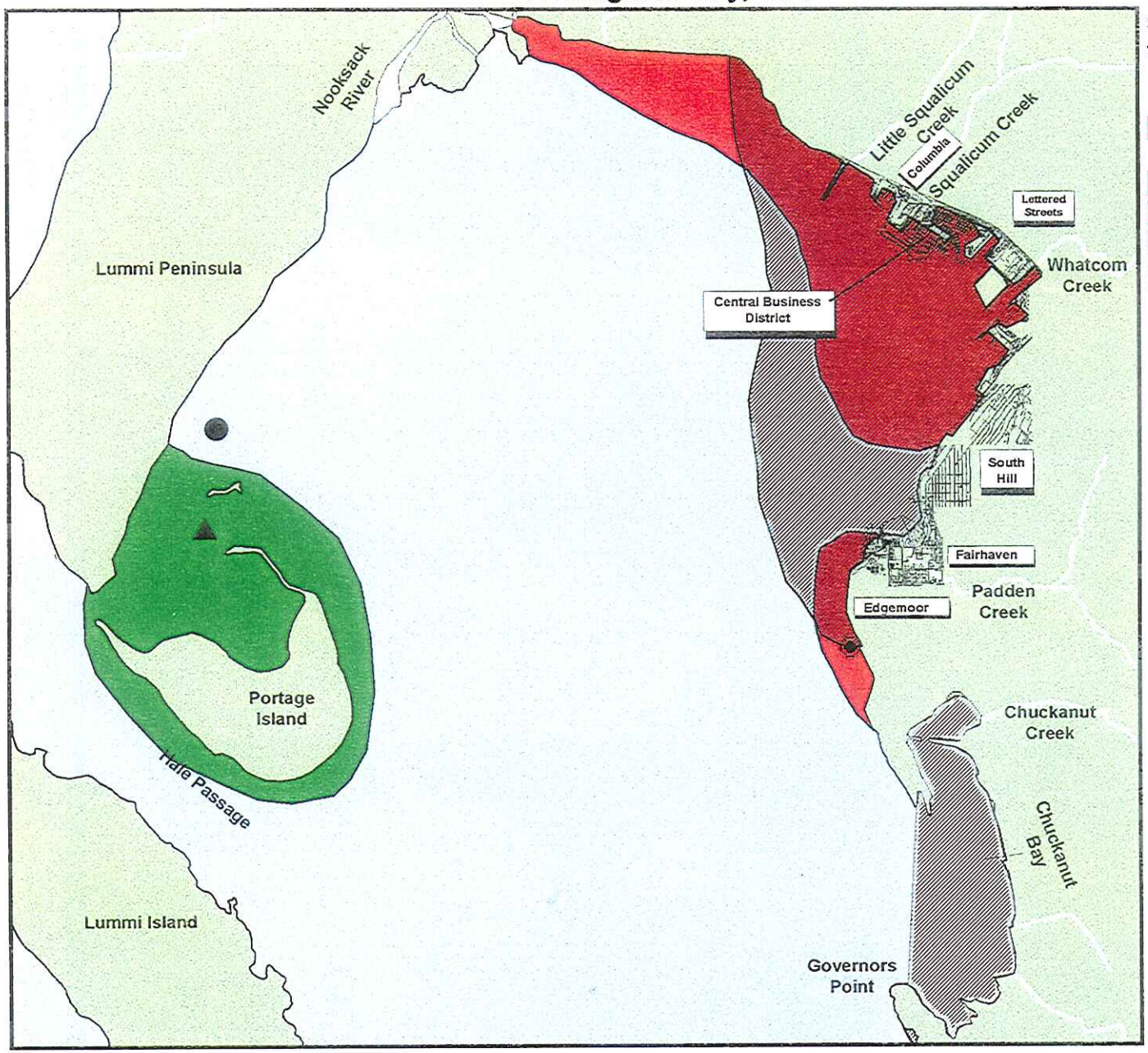
N

5000 0 5000 Feet

June 23, 1998 - Final

J-4478-06 Figure 7-5

Bellingham Bay Demonstration Pilot
Shellfish Certification and Harvest Areas
Bellingham Bay, WA.



Status of Certification for Shellfish Harvests

- Public Shellfish Beach
- Advisory against bottom fish/shellfish harvest
- Certified for commercial harvest
- Overlap
- Uncertifable or decertified
- Closed to Recreational Shellfish Harvest
- Proposed Restricted Commercial Shellfish Harvest Area
- Proposed Approved Recreational Shellfish Harvest Area

BELLINGHAM BAY DEMONSTRATION PILOT

N

5000 0 5000 Feet

June 23, 1998 - Final

J-4478-06 Figure 7-6

8.0 SOURCE CONTROL AND RECONTAMINATION EVALUATION

8.1 Summary

The objective of this source control and recontamination evaluation is to determine whether ongoing contaminant sources are being delivered to Whatcom Waterway and inner Bellingham Bay which could cause constituents to accumulate in surface sediments at concentrations exceeding their SQS values. If ongoing sources are identified, these sources may need to be controlled prior to sediment remediation, otherwise their continued inputs could recontaminate the seabed.

Of the more than 100 analytes analyzed during the RI, only seven were detected in surface sediments at concentrations exceeding SQS chemical criteria. Further, only mercury, phenol, 4-methylphenol, and bis(2-ethylhexyl) phthalate were detected above the SQS in more than one sample, and at concentrations above MCULs. The status of source controls for these primary chemicals of potential concern in the WW Area is summarized below.

8.1.1 Mercury

The available information—including detailed site and source characterization and numeric modeling—indicate that:

- No ongoing, significant sources of mercury are presently discharging to the Whatcom Waterway area;
- Existing discharges from the G-P outfall are not sufficient to result in long-term sediment recontamination in the outfall area to concentrations above the SQS;
- Overall sediment mercury concentrations are decreasing throughout the inner Bellingham Bay area;
- Mercury concentrations detected in sediments throughout the WW Area and near the G-P outfall are likely derived from the reworking of residual accumulations from historical discharges; and
- Sediment natural recovery in the inner bay, including at the G-P outfall site, will be accelerated following the remediation of residual higher concentration "source" areas located within the WW Area, particularly near the G-P log pond.

8.1.2 Phenolics

There appear to be at least two likely sources of phenolic compounds contributing to localized recontamination of sediments in the WW Area:

- Storm water runoff ; and
- *In situ* decomposition of organics and woody material.

Further, both phenol and 4-methylphenol exhibit evidence of significant decay in surface sediments.

The pattern of sediment phenol and 4-methylphenol concentrations in the WW Area matches very closely the distribution of woody debris. All information considered, ongoing discharges from storm drain sources do not appear to represent a significant ongoing source of phenol and 4-methylphenol contamination to sediments. Nevertheless, other phenolic sources may also exist, and contaminant distributions are further complicated by transformation reactions involving addition or removal of methyl groups. Additional studies would be necessary, particularly in the upland drainage basins, before the significance of ongoing phenolic sources could be more fully characterized.

8.1.3 Bis(2-ethylhexyl)phthalate

The areal extent of bis(2-ethylhexyl)phthalate concentrations in the WW Area exceeding SQS or MCUL criteria appears limited and restricted primarily to the nearshore area immediately adjacent to the former Olivine Corporation property on the I&J Street Waterway. The most direct source of bis(2-ethylhexyl) phthalate appears to be upland erosion from adjacent banks and soils of the former Olivine property. Nearshore sediment cleanup could potentially be integrated with future upland cleanup and redevelopment of this site. Local storm water runoff sources may also contribute bis(2-ethylhexyl)phthalate to the waterway.

8.2 Available Source Control Data

The following data were collected or reviewed during the Remedial Investigation to support the identification and evaluation of ongoing sources.

8.2.1 Surface Sediment Data

Chemical concentrations of surface sediments provide evidence of recent contaminant accumulations that may be indicative of ongoing sources. Conversely, chemicals that do not exceed their SQS in surface sediments suggest that present concentrations in source inputs are not high enough to

recontaminate sediments. Surface sediment data are summarized in Section 4.4. Of the more than 100 analytes analyzed, only seven analytes were detected in surface sediments at concentrations exceeding SQS chemical criteria. In order of descending frequency of exceedance relative to SQS chemical criteria, the following analytes were detected in surface sediments within the WW Area:

- Mercury (76% exceeding SQS);
- Phenol (33% exceeding SQS);
- 4-Methylphenol (14% exceeding SQS);
- Bis(2-ethylhexyl) phthalate (4% exceeding SQS; two samples only);
- Acenaphthene (2% exceeding SQS; one sample only);
- Hexachlorobenzene (2% exceeding SQS; one sample only); and
- Benzoic Acid (2% exceeding SQS; one sample only).

For the purposes of this source control evaluation, this list (and related analytes) constitutes the chemicals of potential concern for sediment recontamination. Mercury, phenolics, and bis(2-ethylhexyl)phthalate are evaluated in detail in this section, primarily because these chemicals were detected in more than one sample, and at concentrations above MCULs. Because of their isolated and lower-level enrichments (all below MCULs), acenaphthene, hexachlorobenzene and benzoic acid are considered secondary concerns, and are evaluated in less detail.

8.2.2 Seep, Outfall, and Creek Data

Wet season and dry season samples of discharges from seeps, outfalls, and Whatcom Creek were collected on April 23 and September 26, 1996. Analytical results of these surface water samples were compared to both sediment protection screening criteria and water quality criteria to assess whether current discharges could potentially impact sediments and/or waters of Bellingham Bay.

Sampling locations are shown on Figure 8-1, and physical descriptions of these drainages are presented in the Whatcom Waterway RI/FS Work Plan (Hart Crowser, 1996b). Although twelve potential source sampling locations were originally identified during site reconnaissance, several were not flowing during the sampling periods and could not be sampled. Seven locations were sampled during both wet and dry season events; one location (HC-SW-6) could only be sampled during the wet season. A background seawater location in Bellingham Bay (HC-SW-12) was also sampled. Surface water samples were analyzed for SMS metals (total and dissolved), PAHs, TSS, and field parameters. Metals data from seeps along the Cornwall Avenue Landfill were also incorporated into our evaluation (Landau, 1997).

8.2.3 Sediment Trap Data

Settling particulate matter (SPM) was collected in two sediment traps deployed at locations HC-ST-100 and HC-ST-101, as shown on Figure 2-3. These SPM data, when compared to underlying surface sediment concentrations, provide evidence for recent trends in contaminant inputs to the inner bay. Sediment traps were deployed during three four-month periods spanning a full year of monitoring—October 1996 through January 1997; February 1997 through May 1997; and June 1997 through September 1997. The traps were deployed about 1 meter above the seafloor. The particulate matter collected in the traps was analyzed for mercury, phenolics, TOC, and (first deployment only) grain size. A physical description of settling rates, sedimentation rates, and resuspension rates, as derived from the traps and collocated natural recovery cores, is presented in Section 9.0.

8.2.4 Low-Level Mercury Analysis of Surface Waters

In January 1997, Battelle Marine Sciences Lab analyzed surface water samples in and around Bellingham Bay, using low-level detection limits afforded by EPA Draft Method 1631. These data were primarily used to estimate background mercury concentrations in Bellingham Bay and to develop a site-specific mercury partitioning coefficient for use in a recontamination model of the G-P outfall. Sampling was conducted in accordance with procedures specified in Addendum No. 1 to the Whatcom Waterway Project Plans (Hart Crowser, 1996g). Sampling locations included the Nooksack River and Rosario Strait (freshwater and seawater background locations, respectively), two sites in inner Bellingham Bay (collocated with the two sediment trap locations), and a sample of G-P final effluent just prior to discharge to the deep water outfall. Sampling locations are shown on Figure 8-2. Samples were analyzed for dissolved and particulate mercury (analysis of filtered water and solids retained on the filter, respectively) and TSS.

8.2.5 Georgia-Pacific Effluent Monitoring Data

To assess the recontamination potential of discharges from G-P's deep water outfall, monitoring data for G-P's final effluent were compiled, and annual loading rates were developed for mercury and suspended solids. The last three years of data from 1995 to 1997 were used to develop loading rates; analytical detection limits during these years were improved compared to older monitoring data. These monitoring data are presented in Appendix G.

Average outfall concentrations and loading rates of mercury were developed using one-half detection limit values for those sampling events in which mercury was undetected in the final effluent. Using one-half detection limit values is statistically the least biased substitution method for analyzing data that contain non-detects. Ecology's Toxics Cleanup Program recommends

the use of one-half detection limit values for statistical analysis of environmental data (Ecology, 1992). The uncertainty associated with the statistical treatment of non-detects is evaluated in Sections 8.4.1 and 8.4.5.

8.2.6 Sediment Quality in Upland Drainages

In two recent studies (PTI, 1991 and Cubbage, 1994), sediment samples were collected and analyzed in creeks and storm sewer catch basins draining to Bellingham Bay. These sediments were analyzed for conventional parameters, metals, semivolatile organics, volatile organics, and pesticides/PCBs. These data provide an indication of the quality of particulate material that is being transported in creek drainages and storm water runoff, and that could potentially accumulate in sediments of the inner bay.

8.3 Identification and Characterization of Sources

8.3.1 Chemicals of Concern in Surface Sediments

As summarized in Section 4.0, the primary chemicals of concern in surface sediments of the WW Area are mercury and phenolics (phenol and 4-methylphenol). These chemicals had the highest frequency of occurrence at concentrations above the SQS, and also exhibited some of the highest enrichment ratios (i.e., ratio of sediment concentration to SQS concentration, a measure of relative toxicity). Accordingly, mercury and phenolics were the primary focus of this recontamination evaluation.

The plasticizer compound bis(2-ethylhexyl)phthalate exceeded both SQS and MCUL criteria at a nearshore location (HC-SS-47) on the south bank of the I&J Street Waterway, adjacent to the former Olivine Corporation property between Outfalls HC-SW-10 and HC-SW-11 (Figure 8-1). In addition, similarly elevated bis(2-ethylhexyl)phthalate concentrations have been reported in previous sediment core composites collected in this nearshore area and in adjacent upland areas formerly used for scrap/refuse storage and experimental incineration (Landau, 1994 and HLA, 1996). Bis(2-ethylhexyl)phthalate was detected at a concentration slightly above the SQS in surface sediments (HC-SC-81) sampled at the head of the Whatcom Waterway. No other exceedences of the bis(2-ethylhexyl)phthalate SQS were observed in the WW Area. Based on the available data, the areal extent of bis(2-ethylhexyl)phthalate concentrations exceeding SQS or MCUL criteria appears limited.

Spotty, lower-level enrichments of acenaphthene (HC-SS-47; adjacent to the former Olivine property and coincident with the bis[2-ethylhexyl]phthalate detection discussed above), hexachlorobenzene (HC-SS-34), and benzoic acid (HC-SC-76) were detected in surface sediments of the WW Area. Because of the marginal enrichments of these chemicals (below MCUL and

only 1 to 2.5 times the SQS), and the lack of similar enrichments in adjacent samples, these chemicals are not considered primary source control issues for the WW Area.

No other metals or organic chemicals were detected in surface sediments of the WW Area at concentrations exceeding their respective SQS concentrations.

8.3.2 Chemicals of Concern in Surface Water Discharges

Various surface water discharges to inner Bellingham Bay were compared to *sediment protection screening criteria,* i.e., whole water concentrations which are predicted to prevent recontamination of sediments, and to *water quality criteria* which are protective of marine organisms and human health via fish and shellfish consumption.

The sediment protection screening criteria for surface water, shown in Table 8-1, were developed for SMS metals and polycyclic aromatic hydrocarbons (PAHs) using chemical partitioning coefficients from the literature (Strenge and Peterson, 1989) and from site-specific mercury analyses (see below). The derivation of sediment protection screening criteria for surface waters is explained in greater detail in Section 5.3.6 of the Whatcom Waterway RI/FS Work Plan (Hart Crowser, 1996b). As a conservative screening tool, sediment protection criteria were compared to whole water concentrations, although comparisons with dissolved concentrations are more consistent with partitioning theory.

Water quality criteria are also compiled in Table 8-1. Chronic water quality criteria for metals were derived from state Surface Water Quality Standards, and for PAHs which do not have state standards, criteria are the "Lowest Observed Effects Levels" from EPA's Quality Criteria for Water (1992). Human health criteria were derived from 40-CFR-131 (22Dec92).

Metal concentrations and PAH concentrations of wet season and dry season surface water discharges are summarized in Tables 8-2 and 8-3, respectively.

Low-Level Mercury Analysis. Because standard detection limits are insufficient to allow comparisons with the sediment protection criterion for mercury, low-level mercury analyses using draft EPA Method 1631 were performed on G-P effluent, the Nooksack River, and at other selected locations within and around Bellingham Bay. Particulate, dissolved, and total (whole water) mercury concentrations are summarized in Table 8-4; total mercury concentrations are also reproduced in Table 8-2. To reduce uncertainty, field duplicates were collected and analyzed at every location.

Dissolved mercury concentrations were analyzed directly using field-filtered samples. In contrast, particulate mercury concentrations were estimated by digesting the filtrate, and estimating the mass of sediment on the filtrate from the TSS concentration and the volume of water that passed through the filter. Because of their indirect derivation, exacerbated by TSS errors in low turbidity samples, the particulate mercury concentrations carry a considerable uncertainty. Particulate data are discussed further in Section 8.3.3.

The data also required a blank correction to account for significant amounts of dissolved and particulate mercury in the field blank—a sample of deionized water run through the sampling equipment. Because of the extreme sensitivity of the method, and the high potential for background contamination, blank correction is often required in low-level analytical work. The background seawater sample from Rosario Strait, in particular, carries a high level of uncertainty, because the concentration of mercury in both the dissolved and particulate fractions is less than two times the level of contamination in the field blank.

One of the objectives of performing low-level mercury analysis on both dissolved and particulate fractions was to develop a site-specific partitioning coefficient for mercury in Bellingham Bay. Calculated partitioning coefficients are also presented in Table 8-4. The partitioning coefficients ranged from 300,000 to 7,000,000 L/kg, and averaged 948,000 L/kg (excluding Rosario Strait and the Nooksack River). The site-specific coefficients were used to refine the sediment protection screening criterion for mercury, and were also used as input to the recontamination modeling of the G-P outfall (see Section 8.4).

Metals—Potential Sediment Sources. SMS metals in surface water discharges are compared to screening-level sediment protection criteria in Table 8-2. The sediment protection screening criterion for mercury is so low that it defaults to the practical quantitation limit; using the standard mercury analysis, any detections of mercury would exceed the screening level. A review of G-P effluent monitoring data indicates that the screening criteria were exceeded periodically. The G-P outfall was investigated in greater detail using more detailed contaminant transport models, as described in Section 8.4.

The sediment protection screening criterion for mercury was also exceeded in a water sample obtained during a transient runoff event at location HC-SW-6. Mercury detected in this sample could have been associated with surficial soils or paint/grit particles entrained in runoff from the adjacent BMI property. However, considering the low flow volumes associated with this runoff (2 to 5 gpm) and the relatively low mercury concentration in a sediment sample collected immediately adjacent to the property (0.43 mg/kg in HC-SS-37), this

intermittent discharge does not appear to be a significant source of mercury to waterway sediments.

Sediment protection screening criteria were also exceeded for cadmium at HC-SW-10, silver at HC-SW-4A and HC-SW-6, and zinc at HC-SW-6 and HC-SW-10. However, these metals have not accumulated at concentrations above the SQS in any samples from the WW Area, so these potential sources are not corroborated by sediment chemistry. Discharges of these metals are probably intermittent, and the enrichments (generally less than 2 times the sediment protection criteria) are likely low enough to be diluted below concentrations of concern before the particles settle to the seabed.

Metals—Potential Water Sources. Metals were compared to ecological and human health-based water quality criteria (WQC). Water quality criteria were exceeded for several metals (arsenic, cadmium, copper, lead, mercury, silver, and zinc) and in several discharges (CL-SW-2 and CL-SW-3, HC-SW-1, HC-SW-2, HC-SW-6, HC-SW-7, HC-SW-10, and HC-SW-11). Copper was the criterion most frequently exceeded, due to the relatively low surface water quality standard for this metal. Location HC-SW-10 contained the greatest number of metals enrichments above WQC (arsenic, cadmium, copper, silver, and zinc), and also some of the highest enrichments, although the flow from this outfall is small (0.1 to 5 gpm) and corresponds to a relatively small mass loading. Although exceedences of WQC do not necessarily contribute to recontamination of sediments, they may warrant further investigation due to potential water quality concerns.

PAHs—Potential Sediment Sources. Although PAHs were periodically detected in surface water discharges, their concentrations exceeded sediment protection screening criteria at only one location (HC-SW-11D, the City storm drain at the head of the I&J Street Waterway), and only in the duplicate sample from the dry season sampling event (labeled HC-SW-101). Several of the high molecular weight PAHs (including benzo(a)anthracene, benzo(a)pyrene, benzofluoranthenes, benzo(ghi)perylene, chrysene, and indeno(123-cd)pyrene) exceeded their sediment protection screening criteria, some by a factor of ten or more. Although a single PAH compound exceeded its SQS in a surface sediment from the head of the I&J Street Waterway (acenaphthene in HC-SS-47), the PAH "fingerprint" in the sediment is quite different from that in the storm drain (e.g., acenaphthene was not detected in HC-SW-101). Although PAHs were commonly detected in a recent study of catch basin sediments in Bellingham storm drains, they were not observed at concentrations above the SQS (see Section 8.3.4; Cubbage, 1994). Thus, PAHs do not appear to be significant source control issue for protection of sediments in the WW Area.

PAHs—Potential Water Sources. None of the PAHs exceeded their ambient WQC (or LOELs) for protection of aquatic life. In a single dry weather sample

from municipal outfall HC-SW-11, several of the high molecular weight PAHs exceeded their WQC for protection of human health. The human health-based WQC are derived from an assessment of the potential cancer risk associated with consumption of fish and shellfish that have bioaccumulated PAHs in their tissue (these WQC are thus restricted to the carcinogenic PAHs). However, PAH concentrations in storm water are isolated and intermittent (the enrichments were restricted to a single sample, and were not corroborated in a duplicate sample collected from the same location). Thus, sporadic storm water discharges of PAHs are likely not persistent enough to represent a significant source control concern.

Potential Sources of Phenolics. Although phenols have been identified as a potentially significant source control issue for Bellingham Bay, their importance was not anticipated prior to the RI. As a result, surface water discharges were not analyzed for phenols. However, some limited additional information is available for the G-P outfall, based on the 1993 Class-II inspection of the G-P facility (Ecology, 1994), and from the Cornwall Avenue Landfill seeps, based on Landau (1997).

Phenol was not detected (< 1.2 μg/L), and 4-methylphenol was detected at low concentrations (2.3 μg/L) in a 1993 G-P outfall sample (Ecology, 1994). Similarly low or undetectable concentrations of phenolics were reported in the Cornwall Avenue Landfill seeps (Landau, 1997). These low-level discharges cannot account for the observed distribution of phenolics in the WW Area (see Figure 4-2). Potential phenolic sources to the WW Area are discussed in greater detail below, based on evaluations of storm drain catch basin and settling particulate matter sampling and analysis data.

Potential Sources of Wood Material. As discussed in the Sediment Site and Source Control Documentation Report (BBWG, 1999c) developed by the Pilot Project, no ongoing, significant sources of wood material have been identified within the WW Area that have the potential to recontaminate sediments. Accumulations of bark and associated wood material near the G-P Log Pond and in other areas of the Whatcom Waterway appear to be associated with historical practices. Although relatively limited log rafting operations continue in some areas of the Whatcom Waterway site (e.g., within the Port Log Rafting Area), historically there was much more extensive log rafting throughout inner Bellingham Bay (PTI, 1989). In addition, historical discharges of pulp and other materials from the G-P facility are now controlled by a variety of improved handling, collection, and wastewater treatment processes, all of which are regulated under G-P's existing NPDES permit. Wood material releases to the G-P Log Pond, for example, have been controlled for more than 6 years.

8.3.3 Chemicals of Concern in Settling Particulate Matter

Potential sediment recontamination issues in the WW Area were assessed in more detail by evaluating settling particulate matter (SPM) data available in synoptic analysis of sediment traps, surface sediments, and water column filtrates at selected locations in inner Bellingham Bay (sampling locations HC-XX-100 and HC-XX-101). These SPM samples were analyzed for mercury. In addition, the particulate matter in sediment traps was analyzed for phenols. Analytical results are summarized in Table 8-5.

Mercury. At both sediment trap locations (i.e., HC-ST-100 and HC-ST-101), SPM concentrations of mercury were substantially below the corresponding surficial sediment concentrations, indicating a significant reduction over time in mercury source inputs to the WW Area. The annual average SPM concentration at these sediment trap locations ranged from 0.32 to 0.37 mg/kg, compared with concentrations of 1.1 to 1.5 mg/kg measured in surface sediments directly beneath the traps. Relatively lower SPM concentrations identify a trend of decreasing mercury concentrations in modern sources, compared to higher concentrations in the sediments which include residual mercury from past sources. In other words, newly settled particles are mixed in with residual contaminated sediments (through bioturbation) once they are deposited.

Based on detailed sedimentation analyses summarized in Section 9.0, a large portion of the total SPM collected at HC-ST-100 and HC-ST -101 appears to have been derived from resuspended surface sediments. Nevertheless, even though newly sedimented particles are mixed with resuspended sediments containing residual mercury, concentrations in sediments collected in the sediment traps were, on average, below the SQS, and thus not indicative of an ongoing mercury source control issue in this area. The natural recovery over time of surface sediment mercury concentrations is discussed further in Section 9.0, incorporating the results of sediment core profiling.

Phenolics. Phenol concentrations in SPM are erratic, for example, ranging from < 26 to 12,000 ug/kg in a single sediment trap (HC-ST-100, Table 8-5). Phenol concentrations in sediment trap HC-ST-101 were below the SQS of 420 ug/kg. In contrast, phenol concentrations in sediment trap HC-ST-101 were above the SQS in two of three samples, peaking at 12,000 ug/kg—29 times the SQS, and 10 times the MCUL of 1,200 ug/kg—during the summer deployment. Unlike the enrichments in this sediment trap, phenol concentrations in nearby surface sediments were well below the SQS (i.e., 79 ug/kg at HC-SC-70, see Figure 4-2).

The average 4-methylphenol concentration in SPM samples collected from HC-ST-100 and HC-ST-101 was approximately 37,000 ug/kg, ranging from 3,700 to 140,000 ug/kg. The SPM data exceeded the SQS (or equivalently,

the MCUL of 670 µg/kg), peaking at greater than 200 times the criteria. Though detailed phenolic determinations were not performed on the collocated natural recovery cores at these locations, data collected in the vicinity of the traps nevertheless indicate that 4-methylphenol concentrations in surface sediments are less than 470 ug/kg (Figure 4-2). Further, even the maximum 4-methylphenol concentration detected in surface sediments from inner Bellingham Bay (1,900 ug/kg at HC-SS-09) was well below the average SPM concentration. Thus, resuspension and transport from other areas of the site cannot explain the relatively high 4-methylphenol concentrations observed in the SPM samples.

The SPM data suggest the following:

- Suspended sediments in the WW Area are affected by ongoing sources of both phenol and 4-methylphenol that are not attributable to sediment resuspension. Ongoing upland sources of these chemicals are implied, particularly for 4-methylphenol; and

- Both phenol and 4-methylphenol show evidence of significant decay in surface sediments (i.e., lower surface sediment concentrations compared with SPM), a result which is consistent with the short half life of this compound in aerobic cultures (Callahan et al., 1979). For both chemicals, decay processes apparently have maintained concentrations below the SQS throughout most of the WW Area. Only localized areas closest to wood waste deposits and storm drain outfalls appear to represent areas of concern with respect to phenol and 4-methylphenol (see Figure 4-2). Methylation reactions and transformations also complicate the fate of phenols in the water column and sediments.

Other Constituents. Benzoic acid concentrations were elevated to as high as two times the SQS in SPM samples collected from both sediment traps and throughout the year. However, benzoic acid exceeded its SQS in only one sediment sample collected during the RI, and with a marginal enrichment of 1.2. Although ongoing sources of benzoic acid are indicated by the SPM data, this constituent is evidently rapidly degraded in surface sediments and does not accumulate at concentrations of concern.

8.3.4 Chemicals of Concern in Surface Water Runoff Sediments

Sediments from storm drain catch basins and creeks entering inner Bellingham Bay were analyzed for conventional parameters, metals, semivolatile organics, volatile organics, and pesticides/PCBs in two recent studies (PTI, 1991 and Cubbage, 1994). Selected analytical results for chemicals of concern relative to sediment recontamination are presented in Table 8-6, which also includes a screening-level comparison with SQS criteria. However, such an analysis should be considered a worst-case

evaluation of recontamination, because particles discharged from storm drains will likely be dispersed and mixed with other sources before they settle and deposit in bay sediments.

The Department of Public Works for the City of Bellingham maintains a regular schedule for cleaning out municipal catch basins (Gary Almy, personal communication, 6/18/98). Generally, the City targets a clean out frequency of about once a year, although catch basins with more rapid rates of sediment accumulation may require more frequent cleaning. Between about 1994 and 1996, however, the City suspended clean-out operations because the catch basin sediments, in the form of vactor truck sludge, were subjected to more stringent disposal regulations. By early 1997, the City had re-instated its regular clean-out program. Presently, the City recycles its vactor sludge in the manufacture of cement at Holnam Cement in Seattle.

Mercury. Mercury was detected in storm drains at concentrations up to 0.29 mg/kg, but never at concentrations above the SQS (0.41 mg/kg). Thus, no current or on-going sources of mercury were identified in the upland drainages.

Phenolics. During an initial storm drain and creek sampling/source control study performed in the Bellingham Bay area in 1991, phenol was only marginally enriched above its SQS in 1 of 16 samples, and 4-methylphenol was never enriched above its SQS (PTI, 1991). However, during a follow-up study of the same general area performed in 1993, both phenol and 4-methylphenol were detected at higher concentrations, and often in excess of the SQS and MCUL (Cubbage, 1994). During the 1993 study, phenol exceeded its SQS in 3 of 9 samples, at concentrations as high as 5 times the SQS, although elevated detection limits for some of the samples precludes a more definitive comparison. Concentrations of 4-methylphenol exceeded the SQS in 6 of 9 samples, at concentrations as high as 10 times the SQS and the equivalent MCUL. These data are presented on Figure 8-3.

The available data suggest that storm drains and creeks may represent an on-going source of phenols to sediments in Bellingham Bay. In addition, their occurrence appears to be pervasive in the upland drainages, indicating an association with diffuse runoff rather than a discrete point source. However, the overall distribution of phenol and 4-methylphenol in Bellingham Bay, and particularly within the WW Area, appears more closely associated with the historical deposits of woody debris (see Section 8.3.5). The discrepancy between the absence of phenols in the PTI study and the ubiquity of phenols in the Cubbage study is striking, although the storm drains should have been cleaned out one or more times between the sampling events. The sudden change in phenol concentrations may be caused by one or more of the following:

- Seasonal changes in storm water quality or degree of saturation (i.e., anoxia) of catch basin sediments, which in turn affects the degradation rate of phenolic compounds; and/or

- Length of time since the catch basins had been cleaned out by the Department of Public Works.

An isolated detection of pentachlorophenol was also reported for one catch basin tributary to Little Squalicum Creek (Table 8-6). However, more recent and extensive sampling of this area suggests that exceedences of SQS in this case are restricted to the creek area and do not extend into Bellingham Bay (Ecology and Environment, 1996).

Phthalates. In both the Cubbage and PTI studies, various phthalates—primarily bis(2-ethylhexyl)phthalate and butyl-benzylphthalate—were enriched above SQSs in upland drainages, at concentrations as high as 6 times and 20 times their respective SQSs (Table 8-6). These data suggest that diffuse, on-going sources of phthalates are being discharged in storm water and surface water.

Consistent with a possible storm drain source, bis(2-ethylexyl)phthalate was detected at a concentration slightly above the SQS in surface sediment (HC-SC-81) sampled at the head of the Whatcom Waterway. No other exceedences of the bis(2-ethylhexyl)phthalate SQS were observed in Whatcom Waterway.

As discussed above, bis(2-ethylhexyl)phthalate concentrations exceeded both SQS and MCUL criteria at a nearshore location (HC-SS-47) on the south bank of the I&J Street Waterway, adjacent to the former Olivine Corporation property between Outfalls HC-SW-10 and HC-SW-11 (Figure 8-1). In addition, similarly elevated bis(2-ethylhexyl)phthalate concentrations have been reported in previous sediment core composites collected in this nearshore area and in adjacent upland areas formerly used for scrap/refuse storage and experimental incineration (Landau, 1994 and HLA, 1996). Thus, the most direct source of bis(2-ethylhexyl)phthalate appears to be erosion and runoff from adjacent bank and soils of the former Olivine property. However, based on the available data, the areal extent of bis(2-ethylhexyl)phthalate concentrations exceeding SQS or MCUL criteria appears limited. Nearshore sediment cleanup could potentially be integrated with future upland cleanup and redevelopment of the site.

PAHs. In the Cubbage study, two PAHs were enriched above SQS criteria, and in the PTI study, only isolated and low-level enrichments were reported. These enrichments included dibenz(a,h)anthracene (two samples), indeno(1,2,3-cd)pyrene (two samples), acenaphthene (one sample), and phenanthrene (one sample). These enrichments were less than 2 times their

respective SQS. Although PAHs were detected at concentrations above sediment protection criteria in a discharge from the municipal storm drain at the head of I & J Street Waterway (duplicate of dry weather sample from HC-SW-11), such discharges appear to be intermittent and short-lived (see Section 8.3.2). Thus, PAHs are considered a low priority source control issue for protection of sediment quality.

Other Constituents. Benzoic acid and hexachlorobenzene were marginally above their SQS in only a single sediment sample each, collected during the RI, and are therefore considered low priority source control issues. In upland drainages, benzoic acid was typically detected at concentrations one order of magnitude (factor of ten) less than its SQS; however, a single sample near Post Point (BELL03) was 26 times higher than the SQS. Such an anomaly could potentially cause a localized enrichment in sediments. Although hexachlorobenzene was not detected in either study, detection limits were too high (often 10 times the SQS) to draw definitive conclusions.

8.3.5 Chemicals of Concern – Other Potential Sources

Mercury – Potential Groundwater Source. As part of the chlor/alkali plant RI/FS completed by ENSR (1994), 23 groundwater monitoring wells were installed in upland areas adjacent to the G-P Log Pond. The highest mercury concentration detected in this nearshore area was 9.4 µg/L, exceeding the sediment protection screening criterion (Table 8-1). However, because of the relatively low rate of groundwater flow, the mass loading of mercury represented by these groundwater discharges was at or below approximately 0.005 pound per year. Groundwater does not appear to be a significant source of mercury to the WW Area. Further, substantial source control is indicated in the log pond area, as evidenced by the progressive reduction in surface sediment mercury concentrations over time (based on a comparison of data collected by Bothner, 1973; ENSR, 1994b; and this RI). The need for groundwater controls in the log pond area is evaluated further in the Feasibility Study.

Phenolics – Potential Wood Waste Decomposition Source. As discussed above, the available data suggest that phenol and 4-methylphenol sources to the WW Area are ongoing. Significant enrichments of phenol and 4-methylphenol, averaging 6 times and 29 times the SQS, and peaking at 55 times and 209 times the SQS, respectively, were observed in sediment trap deployments spanning all seasons of the year. These enrichments in the particulate matter of the sediment traps are much greater than the concentrations that have been observed to accumulate in surficial sediments of the inner bay, suggesting that some decomposition of phenols takes place in the aerobic environment at the seafloor (see Section 8.3.3).

In addition to storm water runoff sources discussed above, another possible source of phenolics is the decomposition of woody material associated with WW Area sediments. Degradation products including 4-methylphenol appear to accumulate in anaerobic sediment environments. The phenol enrichments in surface sediments appear to be correlated with woody deposits, as evidenced by a comparison with Figure 3-9. In particular, phenolic compound enrichments in the G-P log pond and between the WIST pier and the R.G. Haley site are coincident with visual estimates of woody material exceeding 20 percent by volume in sediments. Both phenol and 4-methylphenol are known degradation products of lignin (Hodson et al., 1983; Hatcher et al., 1988), and accumulations of these compounds in regional sediments are frequently associated with woody debris deposits (EPA, 1989; PTI, 1998). The pattern of sediment phenol and 4-methylphenol concentrations in Bellingham Bay matches very closely the distribution of woody debris.

In addition, the fate of phenols in the aquatic environment appears to be complicated by methylation reactions, in response to changes in the redox state of the sediments. Phenol enrichments are largely restricted to surface sediments, whereas enrichments of methylated phenols are more prevalent in the subsurface.

Phthalates and PAHs – Potential Upland Erosion Source. As discussed above, bis(2-ethylhexyl)phthalate concentrations exceeded both SQS and MCUL criteria at a nearshore location (HC-SS-47) on the south bank of the I&J Street Waterway, adjacent to the former Olivine Corporation property between Outfalls HC-SW-10 and HC-SW-11 (Figure 8-1). In addition, similarly elevated bis(2-ethylhexyl)phthalate concentrations have been reported in previous sediment core composites collected in this nearshore area and in adjacent upland areas formerly used for scrap/refuse storage and experimental incineration (Landau, 1994 and HLA, 1996). Thus, the most direct source of bis(2-ethylhexyl)phthalate appears to be erosion and runoff from adjacent banks and soils of the former Olivine property. Erosion from this adjacent waterfront property may also explain a detection of acenaphthene in this area. Nevertheless, based on the available data, the areal extent of bis(2-ethylhexyl) phthalate and acenaphthene concentrations exceeding SQS or MCUL criteria appears limited. Nearshore sediment cleanup could potentially be integrated with future upland cleanup and redevelopment of the site.

8.4 Recontamination Evaluation of the G-P Outfall

The recontamination evaluation of the G-P outfall was structured as a series of screening steps. The screening steps progressed from simple calculations, which required very conservative assumptions (and likely overestimate sediment quality impacts), to more complex contaminant transport analyses, which provide a more accurate description of the physical and chemical

processes of sedimentation. The screening evaluation included the following steps, as specified in Addendum No. 1 to the Whatcom Waterway RI/FS Project Plans:

- Step 1—TSS-Normalize Effluent Monitoring Data;
- Step 2—Evaluate Outfall Mixing in a Near-Field Model; and
- Step 3—Update Ecology's Far-Field Model (WASP) Using RI Data.

The methods and results of the screening steps are described in more detail below.

8.4.1 TSS-Normalized Effluent Monitoring Data

In the first step of the recontamination evaluation, we developed average loads of mercury and suspended solids (TSS) in G-P's final effluent based on monitoring data from 1995 through 1997; these monitoring data are provided in Appendix G. Discharges through the G-P outfall ranged from 15 to 57 MGD, and averaged 36.5 MGD. TSS concentrations over the two-year period ranged from about 26 to 197 mg/L, and averaged 96 mg/L. Multiplying the average discharge rate by the average TSS concentration results in an average daily solids load of approximately 13,000 kg/day.

Mercury concentrations over the 1995 to 1997 period (156 samples through October 1997) ranged from less than 0.2 to 4.1 μg/L. Using one-half detection limit values for non-detected data (following Ecology, 1992), the average mercury concentration is 0.24 μg/L (the geometric average concentration is 0.15 μg/L). This corresponds to an average daily loading rate of 0.033 kg/day mercury (or a geometric loading rate of 0.021 kg/day). Other statistical treatments of non-detected data are also possible. Substituting zero or the full value of the detection limit for non-detected values results in average mercury concentrations of 0.17 and 0.31 μg/L, respectively, or average daily loads of 0.024 and 0.043 kg/day. The uncertainty in model predictions resulting from these various methods of statistical treatment is assessed below in Section 8.4.5.

A conservative estimate of particulate mercury concentrations in the final effluent was derived by TSS-normalizing the mercury concentrations (i.e., "worst-case" assumption that all mercury is bound to suspended solids). TSS-normalized mercury concentrations calculated in this manner ranged from 1.6 to 2.5 mg/kg (using geometric and arithmetic averages, respectively). However, this analysis does not account for mixing and dispersion in the water column and bioturbation in the sediments before the particles are incorporated into the seabed. These processes were investigated in subsequent screening steps, as described below.

8.4.2 Evaluation of Outfall Mixing in a Near-Field Model

In the second step of the recontamination evaluation, mixing of G-P effluent with ambient seawater in the near-field dilution zone surrounding the outfall was considered. G-P's regulatory mixing zone is approximately 500 by 2,500 feet in size, equal to the length of the 2,000-foot outfall plus a 250-foot mixing zone on all sides. Hydrodynamic studies of the outfall plume indicate that an average dilution ratio of 200 to 1 occurs within this mixing zone (i.e., 1 part effluent is mixed with 199 parts ambient seawater) (Ebasco, 1994). The majority of this mixing is complete within about 16 feet of the outfall ports (the "zone of initial dilution" where the most vigorous turbulence takes place).

G-P Effluent Composition. As derived in the preceding section, the G-P effluent averages 96 mg/L TSS, and 0.24 µg/L mercury.

Total Suspended Solids (TSS) in Ambient Seawater. The average TSS concentration in ambient seawater was based on recent water column studies conducted by WWU (Colyer, 1998). Three locations were sampled between the WIST pier and Squalicum Harbor, just up-current from the outfall, as shown on Figure 8-4. TSS samples were collected and analyzed across multiple water depths and tidal phases during sampling events in June 1996, August 1996, and January 1997. Salinity and temperature profiles collected concurrently indicated that the low-density surface water layer (lower salinity and higher temperature waters influenced by runoff from the Nooksack River and Whatcom Creek) was typically about 2 meters thick (Collias et al., 1966, observed that the surface water layer fluctuated seasonally between 2 and 5 meters in thickness). The average TSS concentrations in surface waters and deep waters were not significantly different; both averaged approximately 10 mg/L.

Mercury in Ambient Seawater. The average mercury concentration in ambient seawater was based on low-level mercury analyses collected in January 1997 (Table 8-4). Total mercury concentrations in background locations ranged from an average of 0.46 ng/L in Rosario Strait to 4.0 ng/L in the Nooksack River. The much higher mercury concentration in the Nooksack River is largely the result of the much higher suspended solids content in the river (32 mg/L) compared to Rosario Strait (<1 mg/L).

The mercury concentrations in the Nooksack River and Rosario Strait were mixed together in appropriate proportions to estimate the ambient seawater composition in the vicinity of the G-P outfall. The most conservative estimate is to assume that seawater near the outfall is composed of 100 percent Nooksack River water, with the particulate fraction adjusted to match the local TSS concentration of 10 mg/L (i.e., assuming that some of the suspended solids in transport in the river settled out on the seafloor when the river

entered the bay). This approach yields an ambient mercury concentration in Bellingham Bay of 3.5 ng/L.

Other Nooksack-to-Rosario mixing ratios can be estimated on the basis of solids or salt balance determinations. To preserve solids balance and produce a TSS concentration of 10 mg/L in the vicinity of the outfall requires that waters from Rosario Strait (0.7 mg/L) and the Nooksack River (32 mg/L) be mixed together at a ratio of 2.3-to-1. This yields an ambient mercury concentration of 1.5 ng/L. To preserve salt balance and produce a salinity of 26 ppt in the vicinity of the outfall requires that water from Rosario Strait (30 ppt) and the Nooksack River (1 ppt) be mixed together at a ratio of 6-to-1. This yields an ambient mercury concentration of 1.1 ng/L.

We selected the most conservative (i.e., highest) ambient mercury concentration (3.5 ng/L) to use as the baseline assumption in the recontamination modeling which follows. The sensitivity of modeling results to the estimated range of ambient mercury concentrations (3.5, 1.5, and 1.1 ng/L) is evaluated in the Section 8.4.5 below.

Mercury Partitioning Coefficient. The chemical partitioning coefficient describes how mercury is distributed between water and suspended particles, and whether it occurs primarily in dissolved form or bound to solids. The partitioning coefficient is simply the ratio of the particulate concentration on suspended solids to the dissolved concentration in water, assuming that the water and solids are in equilibrium. Mercury partitioning in the vicinity of the outfall plume is influenced by the chemistry of both the effluent and receiving water. Therefore, an average of the partitioning coefficients estimated for the G-P effluent (HC-SW-99) and for the inner bay samples (HC-SW-100 and HC-SW-101) was used to model chemical distribution processes in the outfall plume.

The mean of the partitioning coefficients for samples HC-SW-99, HC-SW-100, and HC-SW-101 is approximately 948,000 L/kg (i.e., 1×10^6 L/kg; see Table 8-4). This value is at the upper end of mercury partitioning coefficients reported in the literature (EPA, 1996a), and therefore provides a conservative (worst-case) prediction of sediment recontamination potential. The range of the calculated partitioning coefficients is high, spanning more than an order of magnitude. This high variability is mostly attributed to uncertainties associated with the sampling, analysis, and calculation of particulate mercury concentrations, especially in low-TSS waters (see **Low-Level Mercury Analysis** in Section 8.3.2 for further discussion).

Mixing Calculations. Plume concentrations were estimated using a standard dilution equation (Fischer et al., 1979 and Ecology, 1996b):

$$C_p = (C_e - C_a) / DF + C_a$$

where:

C_p is the concentration in the mixed outfall plume;
C_e is the concentration in the effluent as it exits the pipe;
C_a is the ambient concentration; and
DF is the calculated dilution factor.
Note: Concentrations may be expressed in ng/L or any other consistent units.

Mixing G-P effluent water (average TSS = 96 mg/L; average Hg = 240 ng/L) and ambient receiving water (average TSS = 10 mg/L; average Hg = 3.5 ng/L) at a dilution ratio of 200 to 1 results in a TSS concentration of 10.4 mg/L and a whole water mercury concentration of 4.7 ng/L in the plume.

The equivalent sediment concentration in equilibrium with the whole water mercury concentration in the plume can be calculated from the following equation:

$$C_{ss} = (C_{ww} / 1{,}000{,}000) / [(TSS / 1{,}000{,}000) + (1/K_d)]$$

where:

C_{ss} is the suspended sediment mercury concentration in mg/kg;
C_{ww} is the whole water mercury concentration in ng/L;
TSS is the total suspended solids concentration in mg/L; and
K_d is the mercury partitioning coefficient in L/kg.

Partitioning the total mercury between water and suspended sediment using a K_d of 1,000,000 L/kg results in an estimated particulate mercury concentration of 0.41 mg/kg—the concentration that could potentially become incorporated into the sediments adjacent to the outfall, but without consideration of ambient water and sediment transport processes. This conservative value equaled the SQS of 0.41 mg/kg. More detailed assessments of water and sediment transport were performed to further assess recontamination potential, as described below.

8.4.3 Updated Far-Field (WASP) Model

The final step in the recontamination evaluation involved the construction of a 3-dimensional, far-field contaminant transport model using the EPA computer code WASP (Water Quality Analysis Simulation Program, version 5; Ambrose et al., 1993). The WASP model performs mass balance calculations for water, solids, and mercury, and evaluates mass transfers between the water column and the sediments. The model includes mixing and chemical partitioning

processes, as in the previous screening step, but also incorporates currents, tidal dispersion, sedimentation, and resuspension.

The footprint of the model developed for the outfall and inner Bellingham Bay consisted of 16 surface sediment segments (segments 33 through 48), as shown on Figure 8-4. Each sediment segment was 975 meters long by 488 meters wide by 12 centimeters deep. Overlying the sediment segments were 16 surface water segments (segments 1 through 16) and 16 deep water segments (segments 17 through 32) to describe water column processes.

The model construction was originally set up by Sharon R. Brown of the Washington State Department of Ecology (Ecology, 1996a; see also PTI, 1992). However, the model was updated and revised wherever possible to include new site-specific data acquired during the RI and in related oceanographic studies by WWU (Colyer, 1998). Model input parameters are summarized in Table 8-7. Model input and output files are provided in Appendix H.

The major revisions to the Ecology model included the following:

- **Boundary TSS Concentration.** The boundary condition for TSS was based on recent water column studies conducted by Colyer (1998, Figure 8-4). The average TSS concentrations in surface waters (freshwater influence) and deep waters both averaged 10 mg/L.

- **Boundary Mercury Concentration.** The background concentration for mercury was established using the recently acquired, low-level mercury data. The particulate mercury concentration in the Nooksack River was adjusted to the local TSS concentration of 10 mg/L in the vicinity of the outfall. As described above, the total (whole water) background mercury concentration was estimated at 3.5 ng/L. This background or boundary condition does not include inputs from sediment resuspension of residual mercury contamination in the inner bay that has accumulated from historical discharges.

- **Initial Mercury Concentrations.** Specification of initial mercury concentrations was necessary to evaluate natural recovery in the dynamic version of the model. Average mercury concentrations were calculated for each model segment using the RI sediment quality database. This database includes data collected during the RI and recent (i.e., within the last five years) sediment quality data in peripheral areas where RI coverage is lacking. Surface sediment mercury concentrations are posted on Figure 4-1, and also on Figure 8-4. The existing mercury concentration in outfall segment No. 38 averages 0.59 mg/kg. The initial mercury concentration in areas removed from the outfall and the industrial

waterfront is 0.23 mg/kg. This concentration was also assigned to the deeper water segments that had no existing surface sediment data.

- **Mercury Partitioning Coefficient.** Estimates of mercury partitioning coefficients were derived from the low-level mercury study (Table 8-1). The geometric mean value of partitioning coefficients in the G-P effluent (HC-SW-99) and in the inner bay (HC-SW-100 and HC-SW-101) was used to predict chemical distribution processes in the outfall plume. The value of this average partitioning coefficient is approximately 1,000,000 L/kg. This coefficient is at the upper end of reported literature values (summarized in EPA's Mercury Report to Congress, 1996 draft), and therefore provides a conservative estimate of recontamination potential.

- **G-P Effluent Discharge Rate, Solids and Mercury Loading Rates.** The flow and composition of the G-P effluent were updated to include recent monitoring data from 1995 through 1997. The average discharge rate was 36.5 MGD (1.6 m^3/sec), and the average solids load was approximately 13,000 kg/day. A conservative estimate of the mercury loading from the G-P effluent discharge was input to the WASP model to address analytical uncertainties. The assumed average mercury loading of 0.043 kg/day was calculated by conservatively substituting the detection limit (typically 0.2 µg/L) for non-detected values. Use of zero or one-half detection limit substitution values would result in lower loading estimates of 0.024 and 0.033 kg/day, respectively (see Section 8.4.5).

- **More Conservative Transport Parameters.** The dispersion coefficients, originally derived for Bellingham Bay by Collias et al. (1966), were reduced to be protective of the more sheltered environment in the inner part of the bay. The coefficients were reduced to one-third of their original value. These lower dispersion coefficients result in predictions of a more concentrated plume near the outfall, and therefore result in higher (i.e., more conservative) predictions of sediment concentrations. The recent work of Colyer (1998) generally confirmed the current directions used previously by Ecology—net longshore drift in a southwesterly direction in the vicinity of the outfall (see Figure 3-3), although intermittent reversals occur during the tidal cycle.

- **Modification of Sedimentation and Resuspension Fluxes.** The structure of the sedimentation fluxes was significantly altered from the original WASP file. A forty-ninth segment was added below the sixteen surface sediment layers to allow sediment to flux through the surface sediments to a "burial" segment below. The sediment segments were also changed to "constant volume" mode (parameter IBEDV in Record 1 of "Volumes"); this is the preferred mode that allows the user to achieve mass balance of sediment fluxes and bulk densities. Finally, a resuspension flux was added to the model to simulate sediment exchange

across the seabed. The dry bulk density of the sediments (0.5 gm/cm^3), and the rates of particle settling (8.2 cm/yr), sedimentation (1.6 cm/yr), and by difference, resuspension (6.6 cm/yr, or 80 percent) were incorporated from dated cores (HC-NR-100, HC-NR-101, and HC-NR-102) and sediment traps (HC-ST-100 and HC-ST-101)(see Section 9.0).

8.4.4 WASP Model Results

Steady-State Model Results. The WASP model was configured to run in both dynamic and steady-state modes. The steady-state model predicts the distribution of surface sediment concentrations that will ultimately equilibrate with source inputs (the G-P outfall in this case). This model provides an assessment of whether the outfall could recontaminate bay sediments, and whether sediment quality will improve or degrade with time. The steady-state version of WASP was coded by James Martin at ASCI in Athens, Georgia, one of the model developers. For computational efficiency, the steady-state version was used to setup the baseline scenario, and to assess model sensitivity and uncertainty which requires varying model input parameters over an observed or statistically defined range of values (see Section 8.4.5 below).

The results of the steady-state WASP model prediction are shown on Figure 8-4; the input and output files are provided in Appendix H. The maximum predicted mercury concentration in the diffuser segment (No. 38) is 0.36 mg/kg. Thus, sediments in the vicinity of the G-P outfall are not predicted to accumulate at concentrations above the SQS (0.41 mg/kg).

Note that predicted sediment mercury concentrations around the perimeter of the model grid are on the order of 0.32 mg/kg. This concentration corresponds to the boundary condition supplied to the model. A whole water mercury concentration of 3.5 ng/L, partitioned onto a solids concentration of 10 mg/L using a distribution coefficient of 1,000,000 L/kg, results in this background concentration on suspended particles and sediments. Steady-state mercury concentrations in the outfall segment are therefore about 0.04 mg/kg (roughly 12 percent) higher than assumed background.

Results from the steady-state modeling, as described below, indicated that outfall discharges would not cause mercury recontamination of the sediments. Accordingly, the dynamic model was used to determine how long it would take the sediments to recover below the SQS, and to more fully assess natural recovery trends in sediments adjacent to the outfall.

Dynamic Model Results. The results of the dynamic WASP model prediction are summarized in Table 8-8 and on Figure 8-5; the input file is provided in Appendix H. The dynamic model shows the rate of approach toward the equilibrium sediment concentration (i.e., the rate of natural recovery).

The dynamic model results indicate that sediments within the outfall area are predicted to recover relatively quickly, decreasing to below the mercury SQS in only a few years. The model prediction represents a post-remediation condition, since it does not incorporate inputs from the resuspension of residual, historical mercury contamination in the inner bay that may be drifting down-current to the outfall site. The present rate of natural recovery, prior to removal or capping of this residual contamination, will therefore be slower than predicted by the model. Additional natural recovery modeling is presented in the feasibility study (FS).

8.4.5 Model Uncertainty Analysis

Some amount of uncertainty is associated with any predictive modeling effort. However, the modeling uncertainty was minimized by incorporating as much site-specific information on physical and chemical transport processes as possible. This site-specific information includes settling and sedimentation rates, background mercury and TSS concentrations, mercury partitioning coefficients, and effluent flow rates and concentrations. These parameters have been averaged from multiple measurements and therefore contain a reasonable degree of statistical certainty and probability.

An uncertainty analysis was performed to determine which input parameters contributed most to modeling errors, and to assess the magnitude of potential modeling errors. This analysis provides a level of confidence in the prediction that the G-P outfall is not a source of mercury recontamination to the bay. Results of the model uncertainty analysis are presented in Table 8-9.

Numerous model input parameters—including outfall loading rates of mercury and suspended solids, advection, dispersion, mercury partitioning coefficient, and ambient mercury concentrations—were varied over a probable range of values to assess their effect on predicted sediment mercury concentrations in model segment No. 38, located within the outfall area. Where sufficient statistical data were available, as for solids loading and sedimentation rates, the value range was developed using 95% confidence limits on the mean. For mercury loading from the outfall, the probable range of values was developed using different statistical treatments for analyzing this "censored" data set (i.e., loadings were developed using different substitution methods for non-detected values, including zero, half-U, and full-U substitution). The probable range of ambient (background) mercury concentrations was developed by examining the potential proportions of freshwater from the Nooksack River and seawater from Rosario Strait which satisfy the solids and salt balance (see also **Mercury in Ambient Seawater** in Section 8.4.2). Finally, advection, dispersion, and the mercury partitioning coefficient were varied by an order of magnitude because such variability is typical for these parameters.

The results of the uncertainty analysis indicate that the model prediction of no recontamination is very robust. Over the probable range of values for all critical input parameters, mercury concentrations in outfall segment No. 38 did not exceed the SQS in any simulation. The highest concentrations resulted from model simulations that assumed (1) extremely low dispersion coefficients at 10 percent of the values reported for Bellingham Bay (predicted mercury = 0.39 mg/kg), and (2) an extremely high mercury partitioning coefficient at three times the geometric mean value (predicted mercury = 0.38 mg/kg). There is a low probability that these input parameters would take on even more extreme values (i.e., lower dispersion or higher partitioning).

One of the most sensitive input parameters is the ambient mercury concentration in Bellingham Bay. However, as described above, the model used the most conservative value for this parameter. Assuming more likely values for ambient water concentrations would result in lower predicted sediment concentrations in the vicinity of the outfalls.

Table 8-1 - Surface Water Screening Levels

Chemical	Marine Sediment Quality Standard Chapter 173-204 WAC	Literature K_d (metals) or K_{oc} (organics)	Sediment Protection Screening Criteria in µg/L (a)	Marine Chronic/ Human Health Water Quality Criteria in ug/L	Laboratory Quantitation Limit in µg/L	Combined Surface Water Screening Level in µg/L(d)
Metals	in mg/kg					
Arsenic	57	21.5 (b)	2,651	36	5	36
Cadmium	5.1	567 (b)	9	8.0 (f)	0.2	8
Chromium	260	360 (b)	722	50	10	50
Copper	390	336 (b)	1,161	2.5 (f)	2	2.5
Lead	450	1,830 (b)	246	5.8 (f)	3	5.8
Mercury	0.41	1,000,000 (h)	0.0004	0.025	0.2	0.2
Silver	6.1	40 (b)	152	1.2 (f)	0.2	1.2
Zinc	410	1,460 (b)	281	77 (f)	10	77
PAHs	in mg/kg - OC					
Naphthalene	99	940 (c)	105	2,350 (e)	0.1	105
Acenaphthylene	66	2,500 (c)	26	300 (e)	0.2	26
Acenaphthene	16	4,600 (c)	3.5	710 (e)	0.1	3.5
Fluorene	23	7,300 (c)	3.2	14,000 (d)	0.02	3.2
Phenanthrene	100	14,000 (c)	7.1	4.6 (e)	0.01	4.6
Anthracene	220	14,000 (c)	16	110,000 (d)	0.01	16
2-Methylnaphthalene	38	4,500 (c)	8.4	300 (e)	0.1	8.4
Fluoranthene	160	38,000 (c)	4.2	370 (d)	0.05	4.2
Pyrene	1,000	38,000 (c)	26	11,000 (d)	0.01	26
Benzo(a)anthracene	110	1,400,000 (c)	0.08	0.03 (d)	0.01	0.03
Chrysene	110	200,000 (c)	0.55	0.03 (d)	0.01	0.03
Total benzofluoranthenes	230	550,000 (c)	0.42	0.03 (d)	0.02	0.03
Benzo(a)pyrene	99	5,500,000 (c)	0.02	0.03 (d)	0.01	0.02
Indeno(1,2,3-cd)pyrene	34	1,600,000 (c)	0.02	0.03 (d)	0.01	0.02
Dibenzo(a,h)anthracene	12	3,300,000 (c)	0.004	0.03 (d)	0.02	0.02 (g)
Benzo(g,h,i)perylene	31	1,600,000 (c)	0.02	300 (e)	0.02	0.02

NOTES:

a. Sediment Protection Criteria, based on partitioning theory, is the sediment quality chemical criteria divided by the partition coefficient (K_d or K_{oc}) and converted to µg/L.
b. Maximum protection coefficient (K_d) value reported in Table 4.1 of Strenge and Peterson (1989).
c. Organic carbon-normalized partition coefficient (Koc) from Table A.6 of Strenge and Peterson (1989).
d. Human health criteria as set forth in 40 CFR 131 (22-Dec-92).
e. Insufficient data to develop criteria. Value presented is the Lowest Observed Effect Level from EPA's Quality Criteria for Water 1992.
f. Dissolved Marine Chronic Water Quality Criteria.
g. Where the reporting limit is greater than the sediment protection criteria or the water quality criteria, the reporting limit is the action level.
h. Updated to a site-specific partitioning coefficient for Bellingham Bay.

Table 8-2 - Summary of Metals Results for Potential Sources in the Whatcom Waterway Area

Source	Sample ID	Season	Approx. Flow in gpm	Salinity in ppt	TSS in mg/L	Total Arsenic in ug/L	Dissolved Arsenic in ug/L	Total Cadmium in ug/L	Dissolved Cadmium in ug/L	Total Copper in ug/L	Dissolved Copper in ug/L	Total Lead in ug/L	Dissolved Lead in ug/L	Total Mercury in ug/L	Dissolved Mercury in ug/L	Total Silver in ug/L	Dissolved Silver in ug/L	Total Zinc in ug/L	Dissolved Zinc in ug/L
Ambient Water Quality Criterion (Chronic)			N/A	N/A	N/A	36	N/A	N/A	8.0	N/A	2.5	N/A	5.8	0.025	N/A	N/A	1.2	N/A	77
Sediment Protection Screening Level (see Table 8-1)			N/A	N/A	N/A	2,600	N/A	9.0	N/A	1,200	N/A	250	N/A	0.0004	N/A	150	N/A	280	N/A
Background Seawater - Rosario Strait	BC-101	Wet	N/A	-	0.7	-	-	-	-	-	-	-	-	0.00046	0.00006	-	-	-	-
Bkgd. Freshwater - Nooksack River	BC-100	Wet	NM	-	32	-	-	-	-	-	-	-	-	0.00404	0.00323	-	-	-	-
Inner Bellingham Bay Seawater	HC-SW-12	Wet	N/A	23.0	27	5 U	5 U	0.2 U	0.2 U	1 U	1 U	3 U	3 U	0.2 U	0.2 U	0.2 UJ	0.2 UJ	10 U	10 U
	HC-SW-12	Dry	N/A	15.0	19	5 U	5 U	0.4 U	0.2 U	1 U	1.8 J	15 U	15 U	0.2 U	0.2 U	0.2 U	0.4 U	10 U	10 U
	HC-SW-100	Wet	N/A	-	0.8	-	-	-	-	-	-	-	-	0.00110	0.00021	-	-	-	-
	HC-SW-101	Wet	N/A	-	2.3	-	-	-	-	-	-	-	-	0.00089	0.00038	-	-	-	-
A. Cornwall Avenue Landfill Seeps	CL-SW-3 (a)	Dry	NM	1.0	-	1 U	< 1 (b)	2 U	< 2 (b)	105	< 105 (b)	53	< 53 (b)	0.1 U	< 0.1 (b)	3 U	< 3 (b)	135	< 135 (b)
	CL-SW-2 (a)	Dry	NM	16.0	-	1	< 1 (b)	2 U	< 2 (b)	8	< 8 (b)	15	< 15 (b)	0.1 U	< 0.1 (b)	3 U	< 3 (b)	24	< 24 (b)
	CL-SW-1 (a)	Dry	NM	1.1	-	1	< 1 (b)	2 U	< 2 (b)	2 U	< 2 (b)	2	< 2 (b)	0.1 U	< 0.1 (b)	3 U	< 3 (b)	8	< 8 (b)
B. R.G. Haley Outfall	HC-SW-1	Wet	480	0.00	32	5 U	5 U	0.2 U	0.2 U	60	33	3 U	3 U	0.2 U	0.2 U	0.2 UJ	0.2 UJ	31	22
	HC-SW-1	Dry	120	0.01	10 U	5 U	5 U	0.2 U	0.2 U	8	4	3 U	3 U	0.2 U	0.2 U	0.2 U	0.2 U	10	17 J
C. R.G. Haley - City Storm Outfall	HC-SW-2	Wet	24	0.11	10 U	5 U	5 U	0.2 U	0.2 U	3.6	2.9	3 U	3 U	0.2 U	0.2 U	0.2 UJ	0.2 UJ	72	49
	HC-SW-2	Dry	4	0.41	10	5 U	5 U	0.2 U	0.2 U	1 U	1 U	3 U	3 U	0.2 U	0.2 U	0.2 U	0.2 U	21	12
D. Whatcom Creek:																			
Near Marine Heritage Center Pk.	HC-SW-4B	Wet	143,000	0.00	47	5 U	5 U	0.2 U	0.2 U	3	2	3 U	3 U	0.2 U	0.2 U	0.2 UJ	0.2 UJ	22	10 U
	HC-SW-4B	Dry	9,800	0.00	10 U	5 U	5 U	0.2 U	0.2 U	2.9	1.9	9.2	3 U	0.2 U	0.2 U	0.22	0.2 U	13	18 J
At Holly Street Bridge	HC-SW-4A	Wet	143,000	0.04	75	5 U	5 U	0.2 U	0.2 U	11	1.8	16	3 U	0.2 U	0.2 U	360 J	0.2 J	49	49
	HC-SW-4A	Dry	9,800	0.33	10 U	5 U	5 U	0.2 U	0.2 U	1.5	1 U	3 U	3 U	0.2 U	0.2 U	0.2 U	0.2 U	10 U	10 U
E. BMI Runoff	HC-SW-6	Wet	2 - 5	0.00	290	15	5 U	1.0	0.2 U	1,100	6.7	59	3 U	0.2	0.2 U	210 J	0.2 UJ	370	10 U
F. "C" Street - City Storm/CSO Outfall	HC-SW-7	Wet	800	0.00	38	5 U	5 U	0.2 U	0.2 U	9.8	3.3	3.3	3 U	0.2 U	0.2 U	0.2 UJ	0.2 UJ	38	12
	HC-SW-7	Dry	115	6.5	11	5 U	5 U	0.90	0.91 J	140	34	14	4.3	0.2 U	0.2 U	0.32	0.2 U	68	43
G. G-P Treatment Lagoon Outfall	'93 Class II	Wet	24,200	2.0	81	1.7 J	-	2.0	-	21 J	-	2.0 J	-	0.29 J	-	0.5 UJ	-	50	-
	'95-96 G-P	Avg.	25,300	-	96		-		-		-		-	0.29 J	-		-		-
	SW-99	Wet	NM	-	98	-	-	-	-	-	-	-	-	0.0840	0.0021	-	-	-	-
H. Bornstein Seafoods Drain Outfall	HC-SW-10	Wet	0.1	1.5	110	350	540 J	19	32 J	1,000	1,600 J	3 U	3 U	0.2 U	0.2 U	15 J	28 J	610	830 J
	HC-SW-10	Dry	1 - 5	2.5	240	64	61	3.8	2.8	130	110	3 U	3 U	0.2 U	0.2 U	1.0	0.92	550	320
I. "I&J" Street - City Storm Outfall	HC-SW-11	Wet	2	0.00	10 U	5 U	5 U	0.2 U	0.2 U	5.1	3.3	3 U	3 U	0.2 U	0.2 U	0.2 UJ	0.2 UJ	25	20
	HC-SW-11	Dry	7	2.5	10 U	5 U	5 U	0.2 U	0.2 U	1.1	1 U	3 U	3 U	0.2 U	0.2 U	0.2 U	0.2 U	16	10 U

NOTES:

(a) From Landau (1997).

(b) Landau (1997) did not determine dissolved metal concentrations; therefore, dissolved water quality criteria were compared to total metal concentrations.

"-" = Not Analyzed; N/A = Not Applicable; NM = Not Measured

U = Not detected at detection limit shown; J = Estimated value; UJ = Estimated detection limit

Table 8-3 - Summary of PAH Results for Potential Sources in Whatcom Waterway Area

Low Molecular Weight PAHs in ug/L

Sample ID	2-methyl-naphthalene	ace-naphthene	ace-naphthylene	anthracene	fluorene	naphthalene	phenan-threne
Wtr Qual. Criteria[3]:	300	710	300	110000	14000	2350	4.6
Sedt. Prot. Criteria[3]:	8.4	3.5	26	16	3.2	105	7.1
HC-SW-1D	0.12 U	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.012 U
HC-SW-1W	0.15	0.19	0.24 U	0.012 U	0.13	0.12 U	0.062
HC-SW-2D	0.12 UE	0.12 UE	0.24 UE	0.012 UE	0.024 UE	0.12 UE	0.012 UE
HC-SW-2W	0.12 U	0.12 U	0.24 U	0.012 U	0.04	0.12 U	0.025
HC-SW-4A-D	0.12 U	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.012 U
HC-SW-4A-W	0.12 U	0.12 U	0.25 U	0.012 U	0.025 U	0.12 U	0.012 U
HC-SW-4B-D	0.12 UE	0.12 UE	0.24 UE	0.012 UE	0.024 UE	0.12 UE	0.012 UE
HC-SW-4B-W	0.12 U	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.012 U
HC-SW-6W	0.13	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.18
HC-SW-7D	0.12 UE	0.12 UE	0.24 UE	0.012 UE	0.024 UE	0.12 UE	0.023 E
HC-SW-7W	0.13 U	0.13 U	0.26 U	0.013 U	0.026 U	0.13 U	0.017
HC-SW-100[1]	0.13 U	0.13 U	0.25 U	0.013 U	0.025 U	0.13 U	0.017
HC-SW-10D	0.12 UE	0.12 UE	0.24 UE	0.012 UE	0.024 UE	0.12 UE	0.046 E
HC-SW-10W	0.12 UE	0.12 UE	0.24 UE	0.012 U	0.024 UE	0.12 UE	0.03 E
HC-SW-11D	0.12 U	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.012 U
HC-SW-101[2]	0.12 U	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.049
HC-SW-11W	0.12 U	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.012 U
HC-SW-12D	0.12 U	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.012 U
HC-SW-12W	0.12 U	0.12 U	0.25 U	0.012 U	0.025 U	0.12 U	0.012 U

Notes:
(1) Field duplilcate of HC-SW-7W
(2) Field duplicate of HC-SW-11D
(3) See Table 8-1 for development of water quality and sediment protection criteria.
U = Not detected at detection limit indicated.
UE = Estimated value/detection limit.

Table 8-3 - Summary of PAH Results for Potential Sources in Whatcom Waterway Area (con't)

High Molecular Weight PAHs in ug/L

Sample ID	fluoranthene	pyrene	benzo(ghi)-perylene	benzo(a)-anthracene	benzo(a)-pyrene	total benzo-fluoranthenes	chrysene	dibenz(ah)-anthracene	indeno-(123-cd)pyrene
Wtr Qual. Criteria[3]:	370	11000	300	0.03	0.03	0.03	0.03	0.03	0.03
Sedt. Prot. Criteria[3]:	4.2	26	0.02	0.08	0.02	0.42	0.55	0.02	0.02
HC-SW-1D	0.024 U	0.044	0.024 U	0.012 U	0.012 U	0.024 U	0.012 U	0.024 U	0.012 U
HC-SW-1W	0.15	0.091	0.024 U	0.012 U	0.012 U	0.024 UE	0.012 U	0.024 U	0.012 U
HC-SW-2D	0.024 UE	0.012 UE	0.024 UE	0.012 UE	0.012 UE	0.024 UE	0.012 UE	0.024 UE	0.012 UE
HC-SW-2W	0.13	0.084	0.024 U	0.021	0.018	0.024 U	0.019	0.024 U	0.012 U
HC-SW-4A-D	0.024 U	0.012 U	0.024 U	0.012 U	0.012 U	0.024 U	0.012 U	0.024 U	0.012 U
HC-SW-4A-W	0.025 U	0.013	0.025 U	0.012 U	0.012 U	0.025 UE	0.012 U	0.025 U	0.012 U
HC-SW-4B-D	0.024 UE	0.012 UE	0.024 UE	0.012 UE	0.012 UE	0.024 UE	0.012 UE	0.024 UE	0.012 UE
HC-SW-4B-W	0.024 U	0.013	0.024 U	0.012 U	0.012 U	0.024 U	0.012 U	0.024 U	0.012 U
HC-SW-6W	0.3 U	0.17	0.024 U	0.012 U	0.016	0.024 U	0.039	0.024 U	0.012 U
HC-SW-7D	0.029 UE	0.036 E	0.024 UE	0.012 UE	0.012 UE	0.024 UE	0.012 UE	0.024 UE	0.012 UE
HC-SW-7W	0.03 U	0.018	0.026 U	0.013 U	0.013 U	0.026 U	0.013 U	0.026 U	0.013 U
HC-SW-100[1]	0.035 U	0.013 U	0.025 U	0.013 U	0.013 U	0.025 U	0.013 U	0.025 U	0.013 U
HC-SW-10D	0.063	0.012 UE	0.024 UE	0.012 UE	0.012 UE	0.024 UE	0.012 UE	0.024 UE	0.012 UE
HC-SW-10W	0.024 UE	0.012 UE	0.024 UE	0.012 UE	0.012 UE	0.024 UE	0.012 UE	0.024 UE	0.012 UE
HC-SW-11D	0.031	0.017	0.024 U	0.012 U	0.012 U	0.024 U	0.012 U	0.024 U	0.012 U
HC-SW-101[2]	1.2	0.95	0.17	0.74	0.43	1.18	1.2	0.024 U	0.21
HC-SW-11W	0.077	0.055	0.024 U	0.012 U	0.019 0	0.024 U	0.012 U	0.024 U	0.012 U
HC-SW-12D	0.024 U	0.012 U	0.024 U	0.012 U	0.012 U	0.024 U	0.012 U	0.024 U	0.012 U
HC-SW-12W	0.025 U	0.012 U	0.025 U	0.012 U	0.012 U	0.025 U	0.012 U	0.025 U	0.012 U

Notes:
(1) Field duplilcate of HC-SW-7W
(2) Field duplicate of HC-SW-11D
(3) See Table 8-1 for development of water quality and sediment protection criteria.
U = Not detected at detection limit indicated.
UE = Estimated value/detection limit.

Table 8-4 - Summary of Low-Level Mercury Results and Partitioning Coefficients

Uncorrected Data

Sample ID	Location	TSS mg/L	Dissolved Mercury ng/L	Particulate Mercury-total[1] ng	Particulate Solids-total[1] mg	Particulate Mercury mg/kg	Total Mercury ng/L	Part. Coef. (K_d) L/kg
HC-BC-101R1	Rosario Strait	0.67	0.30	0.61	0.60	1.03	0.99	3.42E+06
HC-BC-101R2		0.67	0.26	0.73	0.59	1.24	1.09	4.86E+06
HC-BC-100-R1	Nooksack	32.00	3.90	0.92	22.97	0.0399	5.18	1.02E+04
HC-BC-100-R2		32.00	2.99	0.73	16.65	0.0436	4.38	1.46E+04
HC-SW-100-R1	Whatcom	0.75	0.51	1.29	0.68	1.91	1.94	3.77E+06
HC-SW-100-R2		0.75	0.34	0.84	0.62	1.37	1.37	4.05E+06
HC-SW-101-R1	Whatcom	2.30	0.55	0.49	2.05	0.24	1.10	4.41E+05
HC-SW-101-R2		2.30	0.65	1.15	2.33	0.49	1.78	7.57E+05
HC-SW-99-R1	Outfall	98.00	2.07	7.11	7.87	0.90	90.61	4.36E+05
HC-SW-99-R2		98.00	2.62	5.87	6.86	0.86	86.43	3.26E+05
HC-FB-03	Field Blank		0.22	0.32	0.89	0.36		1.63E+06
							Average Kd[2]:	9.48E+05

Field-Blank Corrected Data

Sample ID	Location	TSS mg/L	Dissolved Mercury ng/L	Particulate Mercury-total[1] ng	Particulate Solids-total[1] mg	Particulate Mercury mg/kg	Total Mercury ng/L	Part. Coef. (K_d) L/kg
HC-BC-101R1	Rosario Strait	0.67	0.08	0.30	0.60	0.50	0.41	6.13E+06
HC-BC-101R2		0.67	0.04	0.42	0.59	0.71	0.51	1.91E+07
HC-BC-100-R1	Nooksack	32.00	3.68	0.60	22.97	0.03	4.52	7.10E+03
HC-BC-100-R2		32.00	2.77	0.41	16.65	0.02	3.56	8.85E+03
HC-SW-100-R1	Whatcom	0.75	0.29	0.97	0.68	1.44	1.37	5.01E+06
HC-SW-100-R2		0.75	0.12	0.53	0.62	0.86	0.76	7.14E+06
HC-SW-101-R1	Whatcom	2.30	0.33	0.18	2.05	0.09	0.53	2.63E+05
HC-SW-101-R2		2.30	0.43	0.83	2.33	0.36	1.25	8.27E+05
HC-SW-99-R1	Oufall	98.00	1.85	6.79	7.87	0.86	86.44	4.66E+05
HC-SW-99-R2		98.00	2.40	5.55	6.86	0.81	81.68	3.37E+05
							Average K_d[2]:	1.03E+06

(1) Total mass retained on filter.
(2) Includes both replicates of HC-SW-99, HC-SW-100, and HC-SW-101

Table 8-5 - Summary of Chemical Results for Settling Particulate Matter

Sample ID	Sampling Date	Sampling Type	in mg/kg	Concentration in ug/kg						
			mercury	phenol	2-methyl-phenol	4-methyl-phenol	2,4-dimethyl phenol	penta-chloro-phenol	benzoic acid	benzyl alcohol
HC-ST-100	2/2/97	Sedt Trap	0.50	26 U	11 E	3,700	26 U	11 E	880	15 E
HC-ST-100	5/20/97	Sedt Trap	0.35	490	6.8 E	27,000	370 U	4.9 E	560 E	47 E
HC-ST-100	9/26/97	Sedt Trap	0.20 U	12,000	51 J	140,000	32 U	60 U	1,500	550 U
HC-SW-100	1/28/97	Seawater Part[1]	1.15 E	-	-	-	-	-	-	-
HC-NR-100	9/17/96	Sediment	1.10 E	-	-	-	-	-	-	-
HC-ST-101	2/2/97	Sedt Trap	0.64	28 U	30 U	12,000	2.8 E	7.5 E	1,300	48 U
HC-ST-101	5/20/97	Sedt Trap	n/r[2]	n/r[2]	n/r[2]	n/r[2]	n/r[2]	n/r[2]	n/r[2]	n/r[2]
HC-ST-101	9/26/97	Sedt Trap	0.21 U	310	4.8 J	2,300	1.9 J	3.9 J	820	13 J
HC-SW-101	1/28/97	Seawater Part[1]	0.22 E	-	-	-	-	-	-	-
HC-NR-101	9/17/96	Sediment	1.50	-	-	-	-	-	-	-
		SQS:	0.41	420	63	670	29	360	650	57
		MCUL:	0.59	1,200	63	670	29	690	650	73

Notes:

(1) Particulate concentration estimated from TSS concentration, volume of seawater filtered, and estimated mass retained on filter.

(2) Samples were not recovered (n/r) because sediment trap had tipped over sometime during the deployment period.

"-" Not Analyzed

U = Not detected at detection limit indicated.

E = Estimated concentration.

Table 8-6 - Selected Chemical Results for Sediments Collected from Upland Creeks and Storm Drains

	mercury in mg/kg	phenol in ug/kg	4-methyl-phenol in ug/kg	penta-chloro-phenol in ug/kg	bis(2ethyl)-hexyl)-phthalate in ug/kg (OC)	butyl-benzyl-phthalate in ug/kg (OC)	dimethyl-phthalate in ug/kg (OC)
SQS	0.41	420	670	360	47	4.9	53
MCUL	0.59	1,200	670	690	78	64	53
Source Tracing Study - Phase 1 (PTI, 1991)							
BELL01	0.03	5 E	5 U	14 E	40	2.3	3.3
BELL02	0.09 U	140 U	72 U	360 U	38 E	2.1 U	6.5
BELL03	0.28 U	630 E	270 U	*1,300 U*	51	23.0	3.1 U
BELL06	0.05 U	6 E	20	130 U	77 E	3.3 E	5.2
BELL08	*4.00 U*	*8,300 U*	*4,200 U*	*21,000 U*	290 E	100.0 E	18.0 U
BELL09	*1.30 U*	140 U	68 U	340 U	9 E	1.8 U	1.8 U
BELL13	*7.80 U*	120 U	140	220 U	31	4.9 E	5.9
BELL14	0.06 U	100 U	51 U	260 U	62	*5.7 U*	5.7 U
BELL16	0.08 U	330 U	160 U	*520 U*	19	3.1 U	3.1 U
WHAT01	*0.45 U*	120 U	62 U	310 U	29	2.1 E	0.4 UE
WHAT02	0.05 U	10 U	12	51 U	50	2.0 U	2.0 U
WHAT03	0.13 U	68 U	34 U	170 U	35	7.8 E	0.5 E
LINC01	0.05 U	4 E	6 U	8 UE	130	1.2 U	0.4 E
SQAL01	0.04 U	28	4 E	52 U	10	0.6	0.4 E
SQAL02	0.07 U	150 U	75 U	*370 U*	140	14.0	13.0 U
BAKR01	0.07 U	4 E	3 E	53 U	11	1.5	0.3 E
Source Tracing Study - Phase 2 (Cubbage, 1994)							
WHAT1	0.15	*590 U*	3,800	49 E	*63 U*	*17 U*	17 U
WHAT2	0.05 E	*520 U*	1,700	19 E	*65 U*	*11 U*	11 U
BELL30	0.20	1,100	*800 U*	9 U	*57 U*	*57 U*	*57 U*
BELL31	0.29	940	*700 U*	8 U	*50 U*	*50 U*	50 U
BELL091	0.04 E	510 E	7,000	4 E	12 U	*12 U*	12 U
BELL092	0.25	*450 U*	6,100	9 E	7 U	85	7 U
BELL093	0.10	360 U	560	12 U	16 U	8 E	16 U
BELL131	0.05 E	*2,300 U*	*2,300 U*	12 U	*105 U*	*105 U*	*105 U*
BELL132	0.16	510 E	4,700	20 E	9 U	14	9 U
BELL161	0.03 E	410 U	67 E	REJ	*85 U*	*37 U*	37 U
BELL162	0.03 E	2,100	3,900	790	13 U	110	200

Notes:

Concentration exceeds SQS criterion

Concentration exceeds MCUL criterion

U = Not detected at detection limit indicated.

E = Estimated concentration.

Table 8-7 - WASP Model Input Parameters

Parameter	Value	Units	Reference
Geometry			
Segment Size	975 x 488	m	Ecology, 1996
Surface Water Depth	5	m	Ecology, 1996
Deep Water Depth	4 to 22	m	Ecology, 1996
Surface Sediment Depth	12	cm	Ecology, 1996
Physical Transport			
Net Advection-Surface	0.065	m/sec	Ecology, 1996
Net Advection-Deep	0.032	m/sec	Ecology, 1996
Dispersion	17 to 100	m^2/sec	33% of Ecology values
Effluent Characteristics			
Flow	1.6	m^3/sec	95/97 G-P Monitoring Data
TSS	13,000	kg/day	95/97 G-P Monitoring Data
Mercury	0.043	kg/day	95/97 G-P Monitoring Data
Sedimentation			
Sedimentation Rate	1.6	cm/yr	RI "NR" Cores
Sediment Density	0.5	g/cm^3	RI "NR" Cores
Settling Rate	8.2	cm/yr	RI Sediment Traps
Resuspension	80	percent	Calculated
Boundary Conditions			
Ambient TSS	10	mg/L	Colyer (1998)
Ambient Mercury	0.0035	ug/L	RI Low-Level Hg Sampling
Chemical Parameters			
K_d	1,000,000	L/kg	RI Low-Level Hg Sampling
Initial Sediment Concentrations			
Segment 33	0.64	mg/kg	RI Database
Segment 34	0.41	mg/kg	RI Database
Segment 35	0.23	mg/kg	RI Database
Segment 36	0.23	mg/kg	RI Database
Segment 37	0.25	mg/kg	RI Database
Segment 38	0.59	mg/kg	RI Database
Segment 39	0.23	mg/kg	RI Database
Segment 40	0.23	mg/kg	RI Database
Segment 41	0.23	mg/kg	RI Database
Segment 42	0.39	mg/kg	RI Database
Segments 43-48	0.23	mg/kg	RI Database

Table 8-8 - Ten-Year WASP Model Predictions

Predicted Mercury Concentration in mg/kg

	Model Segment Number															
Year	33	34	35	36	37	38	39	40	41	42	43	44	45	46	47	48
0	0.64	0.41	0.23	0.23	0.25	0.59	0.23	0.23	0.23	0.39	0.23	0.23	0.23	0.23	0.23	0.23
1	0.48	0.37	0.28	0.28	0.29	0.47	0.29	0.28	0.28	0.36	0.28	0.28	0.27	0.28	0.28	0.28
2	0.41	0.35	0.30	0.30	0.31	0.41	0.31	0.30	0.30	0.35	0.30	0.30	0.30	0.30	0.30	0.30
3	0.37	0.34	0.32	0.31	0.32	0.39	0.32	0.31	0.31	0.34	0.32	0.31	0.31	0.31	0.31	0.31
4	0.35	0.34	0.32	0.32	0.32	0.37	0.33	0.32	0.32	0.34	0.32	0.32	0.31	0.32	0.32	0.31
5	0.33	0.34	0.32	0.32	0.32	0.37	0.33	0.32	0.32	0.33	0.33	0.32	0.32	0.32	0.32	0.32
6	0.33	0.33	0.33	0.32	0.33	0.36	0.33	0.32	0.32	0.33	0.33	0.32	0.32	0.32	0.32	0.32
7	0.32	0.33	0.33	0.32	0.33	0.36	0.33	0.32	0.32	0.33	0.33	0.32	0.32	0.32	0.32	0.32
8	0.32	0.33	0.33	0.32	0.33	0.36	0.33	0.33	0.32	0.33	0.33	0.32	0.32	0.32	0.32	0.32
9	0.32	0.33	0.33	0.32	0.33	0.36	0.33	0.33	0.32	0.33	0.33	0.32	0.32	0.32	0.32	0.32
10	0.32	0.33	0.33	0.32	0.33	0.36	0.34	0.33	0.32	0.33	0.33	0.32	0.32	0.32	0.32	0.32
SS	0.32	0.33	0.33	0.32	0.33	0.36	0.34	0.33	0.32	0.33	0.33	0.32	0.32	0.32	0.32	0.32

Mercury concentration above SQS (0.41 mg/kg)

SS = Steady state.

Table 8-9 - WASP Model Uncertainty Analysis

Uncertainty Parameter	Units	Predicted Steady-State Mercury Concentration in Outfall Segment 38 (mg/kg)			Basis of Range	Uncertainty
Mercury Load in Outfall	kg/day	0.03	0.04	0.05	0-U, 1/2-U, and Full-U	Low
		0.35	0.36	0.37		
TSS Load in Outfall	kg/day	13,030	13,280	13,530	95% confidence limits	Low
		0.36	0.36	0.36		
Advection	m/sec	0.1x Ecology	0.33x Ecology	1x Ecology	order of magnitude	Low
		0.37	0.37	0.36		
Dispersion	m2/sec	0.1x Ecology	0.33x Ecology	1x Ecology	order of magnitude	Moderate
		0.39	0.36	0.34		
Sedimentation Rate	cm/yr	1.4	1.6	1.8	95% confidence limits	None[1]
		0.36	0.36	0.36		
Ambient Mercury Conc.	ug/L	0.0011	0.0015	0.0035	14%, 30%, 100% Nooksack	High
		0.14	0.18	0.36		
Partitioning Coefficient	L/kg	300,000	1,000,000	3,000,000	order of magnitude	Moderate
		0.30	0.36	0.38		

Notes:

Baseline Condition

[1] Sedimentation rate does not affect the steady-state prediction, only the recovery rate.

Table 8-2 - Summary of Metals Results

Source	Sample ID	Dissolved Silver in ug/L	Total Zinc in ug/L	Dissolved Zinc in ug/L
Ambient Water Quality Criterion (Chronic)		1.2	N/A	77
Sediment Protection Screening Level (see Table 8-1)		N/A	280	N/A
Background Seawater - Rosario Strait	BC-101	.	.	.
Bkgd. Freshwater - Nooksack River	BC-100	.	.	.
Inner Bellingham Bay Seawater	HC-SW-12	0.2 UJ	10 U	10 U
	HC-SW-12	0.4 U	10 U	10 U
	HC-SW-100	.	.	.
	HC-SW-101	.	.	.
A. Cornwall Avenue Landfill Seeps	CL-SW-3 (a)	< 3 (b)	135	< 135 (b)
	CL-SW-2 (a)	< 3 (b)	24	< 24 (b)
	CL-SW-1 (a)	< 3 (b)	8	< 8 (b)
B. R.G. Haley Outfall	HC-SW-1	0.2 UJ	31	22
	HC-SW-1	0.2 U	10	17 J
C. R.G. Haley - City Storm Outfall	HC-SW-2	0.2 UJ	72	49
	HC-SW-2	0.2 U	21	12
D. Whatcom Creek:				
Near Marine Heritage Center Pk.	HC-SW-4B	0.2 UJ	22	10 U
	HC-SW-4B	0.2 U	13	18 J
At Holly Street Bridge	HC-SW-4A	0.2 J	49	49
	HC-SW-4A	0.2 U	10 U	10 U
E. BMI Runoff	HC-SW-6	0.2 UJ	370	10 U
F. "C" Street - City Storm/CSO Outfall	HC-SW-7	0.2 UJ	38	12
	HC-SW-7	0.2 U	68	43
G. G-P Treatment Lagoon Outfall	'93 Class II	.	50	.
	'95-96 G-P	.		.
	SW-99	.	.	.
H. Bornstein Seafoods Drain Outfall	HC-SW-10	28 J	610	830 J
	HC-SW-10	0.92	550	320
I. "I&J" Street - City Storm Outfall	HC-SW-11	0.2 UJ	25	20
	HC-SW-11	0.2 U	16	10 U

NOTES:

(a) From Landau (1997).

(b) Landau (1997) did not determine dissolved metal conce

"-" = Not Analyzed; N/A = Not Applicable; NM = Not Meast

U = Not detected at detection limit shown; J = Estimated val

Table 8-3 - Summary of PAH Results for Potential Sources in Whatcom Waterway Area

Low Molecular Weight PAHs in ug/L

Sample ID	2-methyl-naphthalene	ace-naphthene	ace-naphthylene	anthracene	fluorene	naphthalene	phenan-threne
Wtr Qual. Criteria[3]:	300	710	300	110000	14000	2350	4.6
Sedt. Prot. Criteria[3]:	8.4	3.5	26	16	3.2	105	7.1
HC-SW-1D	0.12 U	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.012 U
HC-SW-1W	0.15	0.19	0.24 U	0.012 U	0.13	0.12 U	0.062
HC-SW-2D	0.12 UE	0.12 UE	0.24 UE	0.012 UE	0.024 UE	0.12 UE	0.012 UE
HC-SW-2W	0.12 U	0.12 U	0.24 U	0.012 U	0.04	0.12 U	0.025
HC-SW-4A-D	0.12 U	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.012 U
HC-SW-4A-W	0.12 U	0.12 U	0.25 U	0.012 U	0.025 U	0.12 U	0.012 U
HC-SW-4B-D	0.12 UE	0.12 UE	0.24 UE	0.012 UE	0.024 UE	0.12 UE	0.012 UE
HC-SW-4B-W	0.12 U	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.012 U
HC-SW-6W	0.13	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.18
HC-SW-7D	0.12 UE	0.12 UE	0.24 UE	0.012 UE	0.024 UE	0.12 UE	0.023 E
HC-SW-7W	0.13 U	0.13 U	0.26 U	0.013 U	0.026 U	0.13 U	0.017
HC-SW-100[1]	0.13 U	0.13 U	0.25 U	0.013 U	0.025 U	0.13 U	0.017
HC-SW-10D	0.12 UE	0.12 UE	0.24 UE	0.012 UE	0.024 UE	0.12 UE	0.046 E
HC-SW-10W	0.12 UE	0.12 UE	0.24 UE	0.012 U	0.024 UE	0.12 UE	0.03 E
HC-SW-11D	0.12 U	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.012 U
HC-SW-101[2]	0.12 U	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.049
HC-SW-11W	0.12 U	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.012 U
HC-SW-12D	0.12 U	0.12 U	0.24 U	0.012 U	0.024 U	0.12 U	0.012 U
HC-SW-12W	0.12 U	0.12 U	0.25 U	0.012 U	0.025 U	0.12 U	0.012 U

Notes:
(1) Field duplilcate of HC-SW-7W
(2) Field duplicate of HC-SW-11D
(3) See Table 8-1 for development of water quality and sediment protection criteria.
U = Not detected at detection limit indicated.
UE = Estimated value/detection limit.

Table 8-3 - Summary of PAH Results for Potential Sources in Whatcom Waterway Area (con't)

High Molecular Weight PAHs in ug/L

Sample ID	fluoranthene	pyrene	benzo(ghi)-perylene	benzo(a)-anthracene	benzo(a)-pyrene	total benzo-fluoranthenes	chrysene	dibenz(ah)-anthracene	indeno-(123-cd)pyrene
Wtr Qual. Criteria[3]:	370	11000	300	0.03	0.03	0.03	0.03	0.03	0.03
Sedt. Prot. Criteria[3]:	4.2	26	0.02	0.08	0.02	0.42	0.55	0.02	0.02
HC-SW-1D	0.024 U	0.044	0.024 U	0.012 U	0.012 U	0.024 U	0.012 U	0.024 U	0.012 U
HC-SW-1W	0.15	0.091	0.024 U	0.012 U	0.012 U	0.024 UE	0.012 U	0.024 U	0.012 U
HC-SW-2D	0.024 UE	0.012 UE	0.024 UE	0.012 UE	0.012 UE	0.024 UE	0.012 UE	0.024 UE	0.012 UE
HC-SW-2W	0.13	0.084	0.024 U	0.021	0.018	0.024 U	0.019	0.024 U	0.012 U
HC-SW-4A-D	0.024 U	0.012 U	0.024 U	0.012 U	0.012 U	0.024 U	0.012 U	0.024 U	0.012 U
HC-SW-4A-W	0.025 U	0.013	0.025 U	0.012 U	0.012 U	0.025 UE	0.012 U	0.025 U	0.012 U
HC-SW-4B-D	0.024 UE	0.012 UE	0.024 UE	0.012 UE	0.012 UE	0.024 UE	0.012 UE	0.024 UE	0.012 UE
HC-SW-4B-W	0.024 U	0.013	0.024 U	0.012 U	0.012 U	0.024 U	0.012 U	0.024 U	0.012 U
HC-SW-6W	0.3 U	0.17	0.024 U	0.012 U	0.016	0.024 U	0.039	0.024 U	0.012 U
HC-SW-7D	0.029 UE	0.036 E	0.024 UE	0.012 UE	0.012 UE	0.024 UE	0.012 UE	0.024 UE	0.012 UE
HC-SW-7W	0.03 U	0.018	0.026 U	0.013 U	0.013 U	0.026 U	0.013 U	0.026 U	0.013 U
HC-SW-100[1]	0.035 U	0.013 U	0.025 U	0.013 U	0.013 U	0.025 U	0.013 U	0.025 U	0.013 U
HC-SW-10D	0.063	0.012 UE	0.024 UE	0.012 UE	0.012 UE	0.024 UE	0.012 UE	0.024 UE	0.012 UE
HC-SW-10W	0.024 UE	0.012 UE	0.024 UE	0.012 UE	0.012 UE	0.024 UE	0.012 UE	0.024 UE	0.012 UE
HC-SW-11D	0.031	0.017	0.024 U	0.012 U	0.012 U	0.024 U	0.012 U	0.024 U	0.012 U
HC-SW-101[2]	1.2	0.95	0.17	0.74	0.43	1.18	1.2	0.024 U	0.21
HC-SW-11W	0.077	0.055	0.024 U	0.012 U	0.019 0	0.024 U	0.012 U	0.024 U	0.012 U
HC-SW-12D	0.024 U	0.012 U	0.024 U	0.012 U	0.012 U	0.024 U	0.012 U	0.024 U	0.012 U
HC-SW-12W	0.025 U	0.012 U	0.025 U	0.012 U	0.012 U	0.025 U	0.012 U	0.025 U	0.012 U

Notes:
(1) Field duplilcate of HC-SW-7W
(2) Field duplicate of HC-SW-11D
(3) See Table 8-1 for development of water quality and sediment protection criteria.
U = Not detected at detection limit indicated.
UE = Estimated value/detection limit.

Table 8-4 - Summary of Low-Level Mercury Results and Partitioning Coefficients

Uncorrected Data

Sample ID	Location	TSS mg/L	Dissolved Mercury ng/L	Particulate Mercury-total[1] ng	Particulate Solids-total[1] mg	Particulate Mercury mg/kg	Total Mercury ng/L	Part. Coef. (K_d) L/kg
HC-BC-101R1	Rosario Strait	0.67	0.30	0.61	0.60	1.03	0.99	3.42E+06
HC-BC-101R2		0.67	0.26	0.73	0.59	1.24	1.09	4.86E+06
HC-BC-100-R1	Nooksack	32.00	3.90	0.92	22.97	0.0399	5.18	1.02E+04
HC-BC-100-R2		32.00	2.99	0.73	16.65	0.0436	4.38	1.46E+04
HC-SW-100-R1	Whatcom	0.75	0.51	1.29	0.68	1.91	1.94	3.77E+06
HC-SW-100-R2		0.75	0.34	0.84	0.62	1.37	1.37	4.05E+06
HC-SW-101-R1	Whatcom	2.30	0.55	0.49	2.05	0.24	1.10	4.41E+05
HC-SW-101-R2		2.30	0.65	1.15	2.33	0.49	1.78	7.57E+05
HC-SW-99-R1	Outfall	98.00	2.07	7.11	7.87	0.90	90.61	4.36E+05
HC-SW-99-R2		98.00	2.62	5.87	6.86	0.86	86.43	3.26E+05
HC-FB-03	Field Blank		0.22	0.32	0.89	0.36		1.63E+06
							Average Kd[2]:	9.48E+05

Field-Blank Corrected Data

Sample ID	Location	TSS mg/L	Dissolved Mercury ng/L	Particulate Mercury-total[1] ng	Particulate Solids-total[1] mg	Particulate Mercury mg/kg	Total Mercury ng/L	Part. Coef. (K_d) L/kg
HC-BC-101R1	Rosario Strait	0.67	0.08	0.30	0.60	0.50	0.41	6.13E+06
HC-BC-101R2		0.67	0.04	0.42	0.59	0.71	0.51	1.91E+07
HC-BC-100-R1	Nooksack	32.00	3.68	0.60	22.97	0.03	4.52	7.10E+03
HC-BC-100-R2		32.00	2.77	0.41	16.65	0.02	3.56	8.85E+03
HC-SW-100-R1	Whatcom	0.75	0.29	0.97	0.68	1.44	1.37	5.01E+06
HC-SW-100-R2		0.75	0.12	0.53	0.62	0.86	0.76	7.14E+06
HC-SW-101-R1	Whatcom	2.30	0.33	0.18	2.05	0.09	0.53	2.63E+05
HC-SW-101-R2		2.30	0.43	0.83	2.33	0.36	1.25	8.27E+05
HC-SW-99-R1	Oufall	98.00	1.85	6.79	7.87	0.86	86.44	4.66E+05
HC-SW-99-R2		98.00	2.40	5.55	6.86	0.81	81.68	3.37E+05
							Average K_d[2]:	1.03E+06

(1) Total mass retained on filter.
(2) Includes both replicates of HC-SW-99, HC-SW-100, and HC-SW-101

Table 8-5 - Summary of Chemical Results for Settling Particulate Matter

Sample ID	Sampling Date	Sampling Type	in mg/kg	Concentration in ug/kg						
			mercury	phenol	2-methyl-phenol	4-methyl-phenol	2,4-dimethyl phenol	penta-chloro-phenol	benzoic acid	benzyl alcohol
HC-ST-100	2/2/97	Sedt Trap	0.50	26 U	11 E	3,700	26 U	11 E	880	15 E
HC-ST-100	5/20/97	Sedt Trap	0.35	490	6.8 E	27,000	370 U	4.9 E	560 E	47 E
HC-ST-100	9/26/97	Sedt Trap	0.20 U	12,000	51 J	140,000	32 U	60 U	1,500	550 U
HC-SW-100	1/28/97	Seawater Part[1]	1.15 E	-	-	-	-	-	-	-
HC-NR-100	9/17/96	Sediment	1.10 E	-	-	-	-	-	-	-
HC-ST-101	2/2/97	Sedt Trap	0.64	28 U	30 U	12,000	2.8 E	7.5 E	1,300	48 U
HC-ST-101	5/20/97	Sedt Trap	n/r[2]	n/r[2]	n/r[2]	n/r[2]	n/r[2]	n/r[2]	n/r[2]	n/r[2]
HC-ST-101	9/26/97	Sedt Trap	0.21 U	310	4.8 J	2,300	1.9 J	3.9 J	820	13 J
HC-SW-101	1/28/97	Seawater Part[1]	0.22 E	-	-	-	-	-	-	-
HC-NR-101	9/17/96	Sediment	1.50	-	-	-	-	-	-	-
		SQS:	0.41	420	63	670	29	360	650	57
		MCUL:	0.59	1,200	63	670	29	690	650	73

Notes:

(1) Particulate concentration estimated from TSS concentration, volume of seawater filtered, and estimated mass retained on filter.

(2) Samples were not recovered (n/r) because sediment trap had tipped over sometime during the deployment period.

"-" Not Analyzed

U = Not detected at detection limit indicated.

E = Estimated concentration.

Table 8-6 - Selected Chemical Results for Sediments Collected from Upland Creeks and Storm Drains

	mercury in mg/kg	phenol in ug/kg	4-methyl-phenol in ug/kg	penta-chloro-phenol in ug/kg	bis(2ethyl)-hexyl)-phthalate in ug/kg (OC)	butyl-benzyl-phthalate in ug/kg (OC)	dimethyl-phthalate in ug/kg (OC)
SQS	0.41	420	670	360	47	4.9	53
MCUL	0.59	1,200	670	690	78	64	53
Source Tracing Study - Phase 1 (PTI, 1991)							
BELL01	0.03	5 E	5 U	14 E	40	2.3	3.3
BELL02	0.09 U	140 U	72 U	360 U	38 E	2.1 U	6.5
BELL03	0.28 U	630 E	270 U	*1,300 U*	51	23.0	3.1 U
BELL06	0.05 U	6 E	20	130 U	77 E	3.3 E	5.2
BELL08	*4.00 U*	*8,300 U*	*4,200 U*	*21,000 U*	290 E	100.0 E	18.0 U
BELL09	*1.30 U*	140 U	68 U	340 U	9 E	1.8 U	1.8 U
BELL13	*7.80 U*	120 U	140	220 U	31	4.9 E	5.9
BELL14	0.06 U	100 U	51 U	260 U	62	*5.7 U*	5.7 U
BELL16	0.08 U	330 U	160 U	*520 U*	19	3.1 U	3.1 U
WHAT01	*0.45 U*	120 U	62 U	310 U	29	2.1 E	0.4 UE
WHAT02	0.05 U	10 U	12	51 U	50	2.0 U	2.0 U
WHAT03	0.13 U	68 U	34 U	170 U	35	7.8 E	0.5 E
LINC01	0.05 U	4 E	6 U	8 UE	130	1.2 U	0.4 E
SQAL01	0.04 U	28	4 E	52 U	10	0.6	0.4 E
SQAL02	0.07 U	150 U	75 U	*370 U*	140	14.0	13.0 U
BAKR01	0.07 U	4 E	3 E	53 U	11	1.5	0.3 E
Source Tracing Study - Phase 2 (Cubbage, 1994)							
WHAT1	0.15	*590 U*	3,800	49 E	*63 U*	*17 U*	17 U
WHAT2	0.05 E	*520 U*	1,700	19 E	*65 U*	*11 U*	11 U
BELL30	0.20	1,100	*800 U*	9 U	*57 U*	*57 U*	*57 U*
BELL31	0.29	940	*700 U*	8 U	*50 U*	*50 U*	50 U
BELL091	0.04 E	510 E	7,000	4 E	12 U	*12 U*	12 U
BELL092	0.25	*450 U*	6,100	9 E	7 U	85	7 U
BELL093	0.10	360 U	560	12 U	16 U	8 E	16 U
BELL131	0.05 E	*2,300 U*	*2,300 U*	12 U	*105 U*	*105 U*	*105 U*
BELL132	0.16	510 E	4,700	20 E	9 U	14	9 U
BELL161	0.03 E	410 U	67 E	REJ	*85 U*	*37 U*	37 U
BELL162	0.03 E	2,100	3,900	790	13 U	110	200

Notes:

Concentration exceeds SQS criterion

Concentration exceeds MCUL criterion

U = Not detected at detection limit indicated.

E = Estimated concentration.

Table 8-7 - WASP Model Input Parameters

Parameter	Value	Units	Reference
Geometry			
Segment Size	975 x 488	m	Ecology, 1996
Surface Water Depth	5	m	Ecology, 1996
Deep Water Depth	4 to 22	m	Ecology, 1996
Surface Sediment Depth	12	cm	Ecology, 1996
Physical Transport			
Net Advection-Surface	0.065	m/sec	Ecology, 1996
Net Advection-Deep	0.032	m/sec	Ecology, 1996
Dispersion	17 to 100	m^2/sec	33% of Ecology values
Effluent Characteristics			
Flow	1.6	m^3/sec	95/97 G-P Monitoring Data
TSS	13,000	kg/day	95/97 G-P Monitoring Data
Mercury	0.043	kg/day	95/97 G-P Monitoring Data
Sedimentation			
Sedimentation Rate	1.6	cm/yr	RI "NR" Cores
Sediment Density	0.5	g/cm^3	RI "NR" Cores
Settling Rate	8.2	cm/yr	RI Sediment Traps
Resuspension	80	percent	Calculated
Boundary Conditions			
Ambient TSS	10	mg/L	Colyer (1998)
Ambient Mercury	0.0035	ug/L	RI Low-Level Hg Sampling
Chemical Parameters			
K_d	1,000,000	L/kg	RI Low-Level Hg Sampling
Initial Sediment Concentrations			
Segment 33	0.64	mg/kg	RI Database
Segment 34	0.41	mg/kg	RI Database
Segment 35	0.23	mg/kg	RI Database
Segment 36	0.23	mg/kg	RI Database
Segment 37	0.25	mg/kg	RI Database
Segment 38	0.59	mg/kg	RI Database
Segment 39	0.23	mg/kg	RI Database
Segment 40	0.23	mg/kg	RI Database
Segment 41	0.23	mg/kg	RI Database
Segment 42	0.39	mg/kg	RI Database
Segments 43-48	0.23	mg/kg	RI Database

Table 8-8 - Ten-Year WASP Model Predictions

Predicted Mercury Concentration in mg/kg

	Model Segment Number															
Year	33	34	35	36	37	38	39	40	41	42	43	44	45	46	47	48
0	0.64	0.41	0.23	0.23	0.25	0.59	0.23	0.23	0.23	0.39	0.23	0.23	0.23	0.23	0.23	0.23
1	0.48	0.37	0.28	0.28	0.29	0.47	0.29	0.28	0.28	0.36	0.28	0.28	0.27	0.28	0.28	0.28
2	0.41	0.35	0.30	0.30	0.31	0.41	0.31	0.30	0.30	0.35	0.30	0.30	0.30	0.30	0.30	0.30
3	0.37	0.34	0.32	0.31	0.32	0.39	0.32	0.31	0.31	0.34	0.32	0.31	0.31	0.31	0.31	0.31
4	0.35	0.34	0.32	0.32	0.32	0.37	0.33	0.32	0.32	0.34	0.32	0.32	0.31	0.32	0.32	0.31
5	0.33	0.34	0.32	0.32	0.32	0.37	0.33	0.32	0.32	0.33	0.33	0.32	0.32	0.32	0.32	0.32
6	0.33	0.33	0.33	0.32	0.33	0.36	0.33	0.32	0.32	0.33	0.33	0.32	0.32	0.32	0.32	0.32
7	0.32	0.33	0.33	0.32	0.33	0.36	0.33	0.32	0.32	0.33	0.33	0.32	0.32	0.32	0.32	0.32
8	0.32	0.33	0.33	0.32	0.33	0.36	0.33	0.33	0.32	0.33	0.33	0.32	0.32	0.32	0.32	0.32
9	0.32	0.33	0.33	0.32	0.33	0.36	0.33	0.33	0.32	0.33	0.33	0.32	0.32	0.32	0.32	0.32
10	0.32	0.33	0.33	0.32	0.33	0.36	0.34	0.33	0.32	0.33	0.33	0.32	0.32	0.32	0.32	0.32
SS	0.32	0.33	0.33	0.32	0.33	0.36	0.34	0.33	0.32	0.33	0.33	0.32	0.32	0.32	0.32	0.32

Mercury concentration above SQS (0.41 mg/kg)

SS = Steady state.

Table 8-9 - WASP Model Uncertainty Analysis

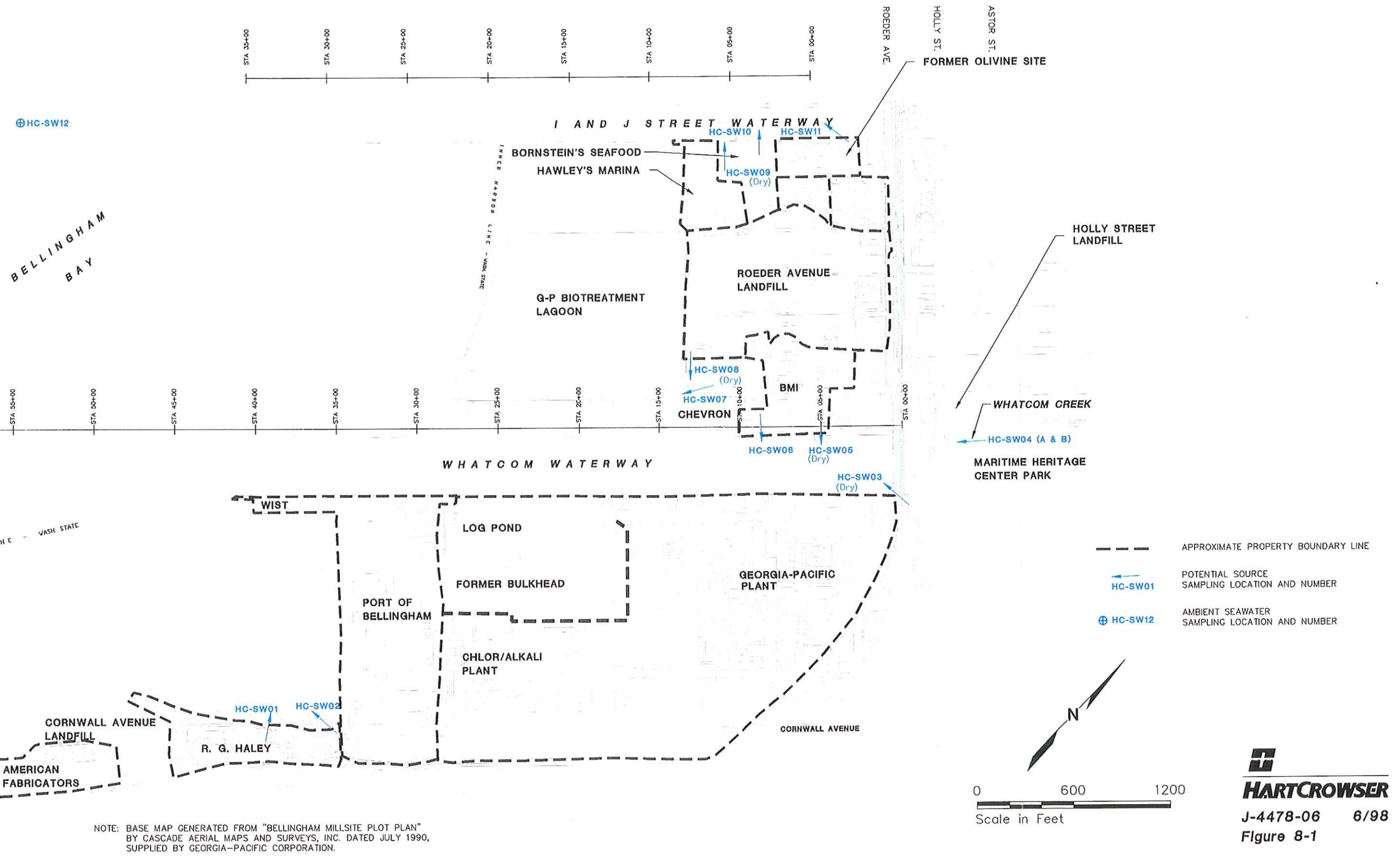
Uncertainty Parameter	Units	Predicted Steady-State Mercury Concentration in Outfall Segment 38 (mg/kg)			Basis of Range	Uncertainty
Mercury Load in Outfall	kg/day	0.03	0.04	0.05	0-U, 1/2-U, and Full-U	Low
		0.35	0.36	0.37		
TSS Load in Outfall	kg/day	13,030	13,280	13,530	95% confidence limits	Low
		0.36	0.36	0.36		
Advection	m/sec	0.1x Ecology	0.33x Ecology	1x Ecology	order of magnitude	Low
		0.37	0.37	0.36		
Dispersion	m2/sec	0.1x Ecology	0.33x Ecology	1x Ecology	order of magnitude	Moderate
		0.39	0.36	0.34		
Sedimentation Rate	cm/yr	1.4	1.6	1.8	95% confidence limits	None[1]
		0.36	0.36	0.36		
Ambient Mercury Conc.	ug/L	0.0011	0.0015	0.0035	14%, 30%, 100% Nooksack	High
		0.14	0.18	0.36		
Partitioning Coefficient	L/kg	300,000	1,000,000	3,000,000	order of magnitude	Moderate
		0.30	0.36	0.38		

Notes:

Baseline Condition

[1] Sedimentation rate does not affect the steady-state prediction, only the recovery rate.

Source Sampling Location Plan
Whatcom Waterway Area



NOTE: BASE MAP GENERATED FROM "BELLINGHAM MILLSITE PLOT PLAN" BY CASCADE AERIAL MAPS AND SURVEYS, INC. DATED JULY 1990, SUPPLIED BY GEORGIA-PACIFIC CORPORATION.

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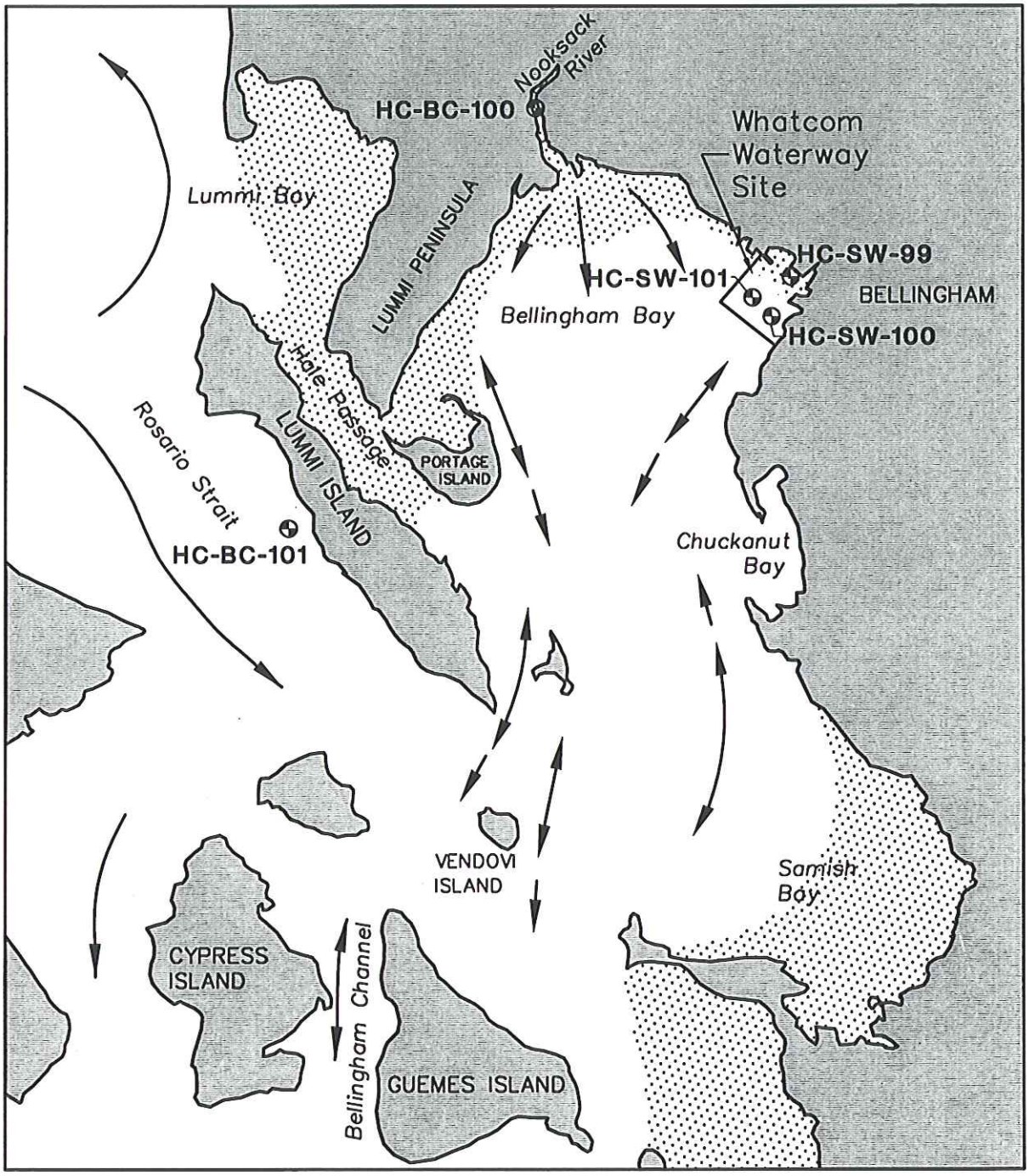
Figure 8-1

Low-Level Mercury Sampling Location Map



Shoal Area (< 20 Feet MLLW)

HC-SW-99 Low-Level Mercury Sampling Location and Number

Net Current Drift Directions Inferred from Puget Sound Environmental Atlas (1987) and Collias (1966)

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Figure 8-2

CVD 6/29/9L -15000 woodstk.pcp
44780628

Phenol Concentrations in Municipal Catch Basins

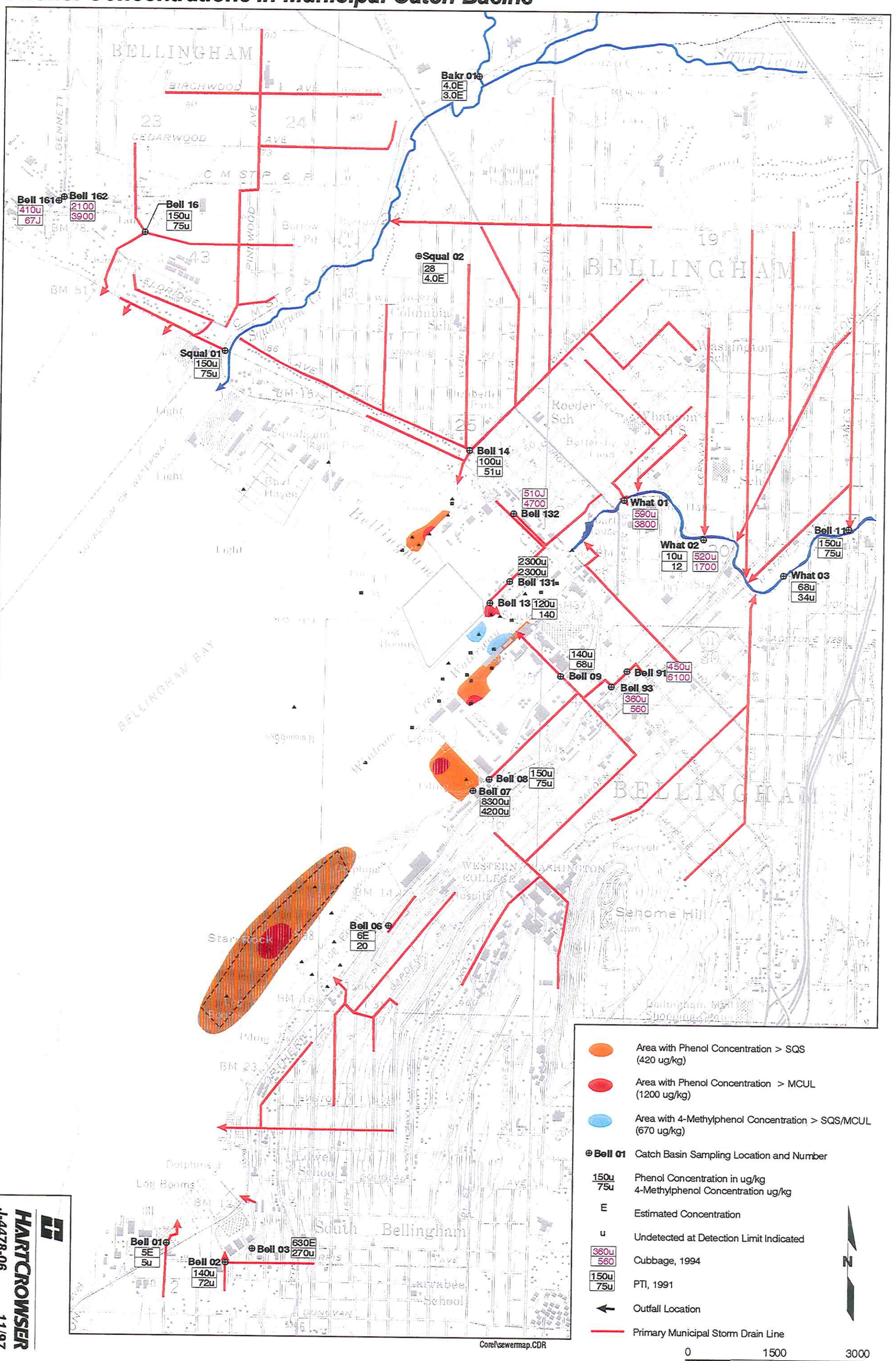

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Figure 8-3

10. FEASIBILITY STUDY REPORT ORGANIZATION

The Feasibility Study (FS) builds upon the results of the RI presented in Volume I of this report. The FS is intended to provide sufficient data, analysis, and engineering evaluations to enable Ecology to select a cleanup action alternative that is protective of human health and the environment.

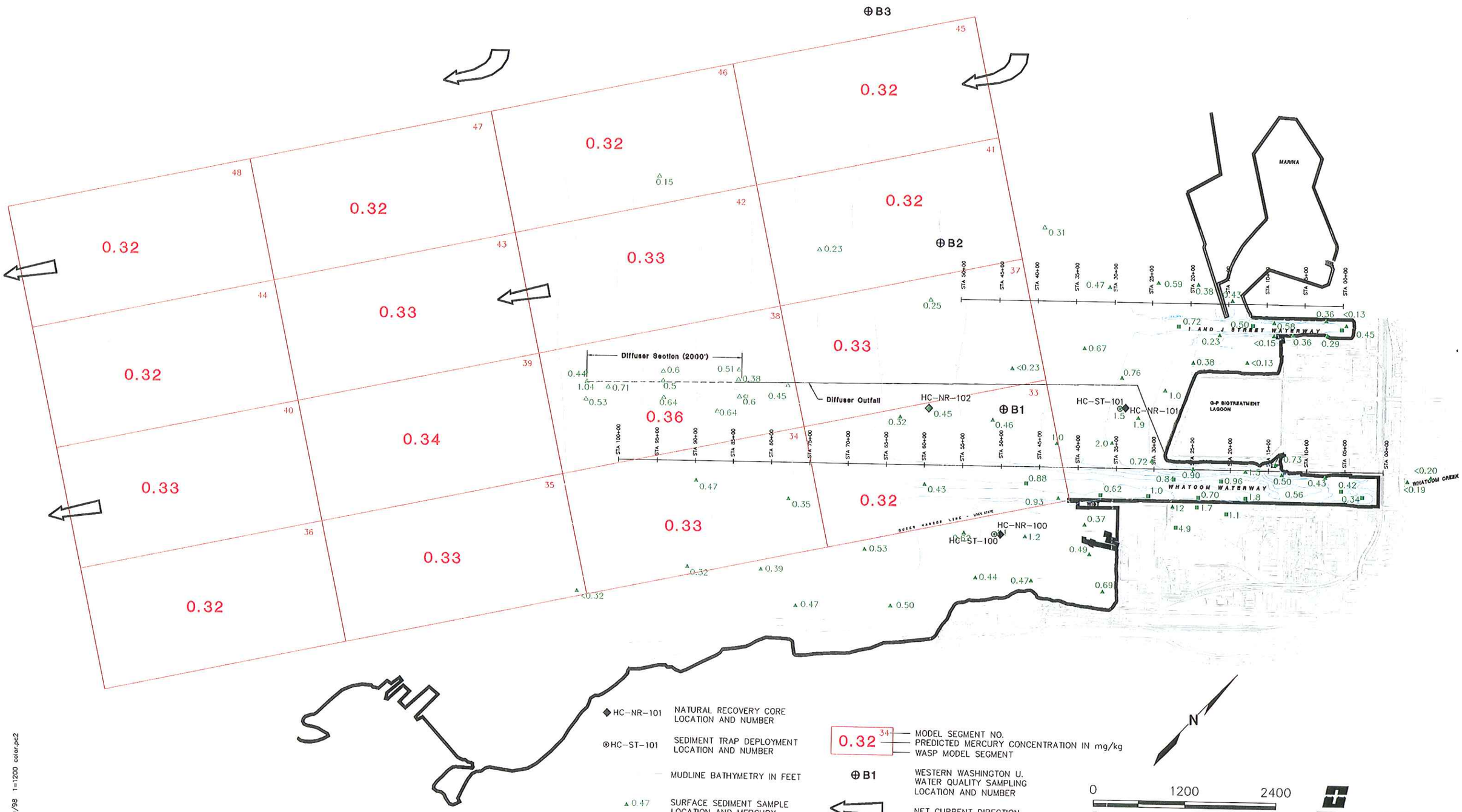
There are five sections of the FS Report following this introduction section, as summarized below:

- **SECTION 11 – Sediment Cleanup Requirements.** This section provides a review of the available data, including summary assessments of SMS cleanup criteria comparisons, source control, and potentially applicable laws, to provide a framework of appropriate sediment cleanup requirements for the WW Area.
- **SECTION 12 – Establishment of Site Sediment Units.** This section reviews the RI data presented in Volume I and establishes site sediment units (SSU) based on unique physical, chemical, biological, and navigational/land use characteristics.
- **SECTION 13 – Identification and Assembly of Cleanup Technologies.** This section identifies and screens potential cleanup technologies, applies them to the appropriate SSUs identified in Section 12, and assembles potential remedial action alternatives.
- **SECTION 14 – Detailed Evaluation of Cleanup Action Alternatives.** This section evaluates the different remedial action alternatives identified in Section 13 based on SMS evaluation criteria as generally described in the Sediment Cleanup Standards User Manual and the MTCA Cleanup Standards Regulation.
- **SECTION 15 - References.** This section presents references used in the development of the RI/FS.

Volume III of the report contains technical appendices for both the RI and FS reports. Appendices A-J provide additional information supporting the RI, and Appendices K-N provide additional information supporting this FS.

Appendix K provides additional information on the natural recovery modeling used to evaluate alternatives. Appendix L summarizes the Puget Sound Dredged Disposal Analysis (PSDDA) evaluation completed as part of the FS, following Addendum No. 2 to the RI/FS Project Plans (Hart Crowser, 1997b). Appendix M summarizes the sequential batch leaching test data completed as part of the FS, also following Addendum No. 2 to the RI/FS Project Plans. Appendix N presents FS-level cost estimates and costing assumptions for each of the remedial action alternatives evaluated in Section 14.

WASP Model Configuration and Predicted Sediment Mercury Concentrations



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44780530

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Figure 8-4

Predicted Sediment Mercury Concentrations in Model Segment 38

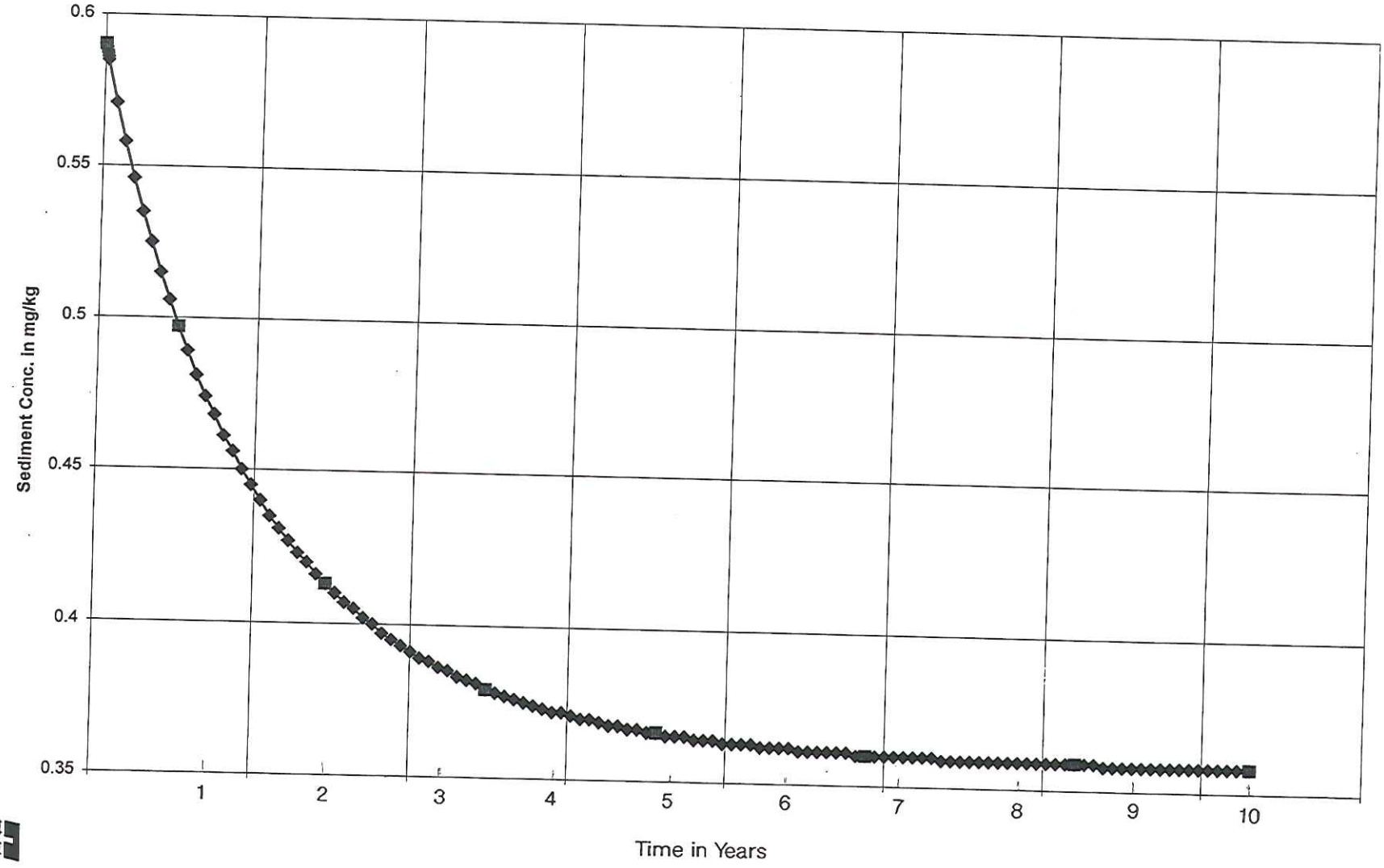

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Figure 8-5

9.0 SEDIMENT NATURAL RECOVERY EVALUATION

This section presents information on sediment natural recovery processes within inner Bellingham Bay. Previous work (Bothner, 1973; Bothner et al., 1980; and Officer and Lynch, 1989) established that surface sediment mercury concentrations in Bellingham Bay have decreased through time since the early 1970s, following improved wastewater treatment at the G-P facility.

This section presents the results of radioisotope and chemical profiles in natural recovery cores, and sediment trap deployments over a one-year period. Settling, sedimentation, and resuspension rates are calculated, and natural recovery rates are estimated from historical mercury profiles. Additional natural recovery modeling, including an expanded spatial analysis and simulation of remediation scenarios, is presented in the Feasibility Study.

9.1 Natural Recovery Core and Sediment Trap Data

As part of the RI/FS sampling effort, three natural recovery cores (HC-NR-100, HC-NR-101, and HC-NR-102) were collected and two sediment traps (HC-ST-100 and HC-ST-101) were deployed and sampled within the study area (Figure 2-2). Sediment traps HC-ST-100 and HC-ST-101 were collocated with natural recovery cores HC-NR-100 and HC-NR-101, respectively; the traps were deployed for three periods, each approximately four months in duration.

The natural recovery cores were sectioned in approximately 2-centimeter increments as described in the approved Sampling and Analysis Plan (SAP; Hart Crowser, 1996e). Selected subsamples were submitted for isotopic analysis of lead-210 (Pb-210), and cesium-137 (Cs-137), and chemical analysis of total mercury, and total solids. Detailed descriptions of the sediment stratigraphy, and compaction-corrected sampling intervals for the natural recovery cores are presented in Appendix A. Validated analytical results for the radioisotope and chemical analyses are presented in Appendix B. Data from the natural recovery cores were used to estimate the net sedimentation rates in the study area, and to evaluate mercury concentration trends through time.

Sediment traps were deployed, retrieved, and sampled as described in Section 8.1. Settled particulate matter (SPM) that had accumulated in the traps was analyzed for total mercury, phenols, TOC, and total solids. Data from the sediment trap study were used to estimate gross sedimentation rates, and to characterize the chemical and physical properties of SPM in the study area. In addition, comparison of gross sedimentation rates in sediment

traps with net sedimentation rates in collocated, radio-dated cores provides an estimate of resuspension rates.

9.2 Sedimentation Rates

Estimated rates of gross and net sedimentation, which are required to support natural recovery and recontamination evaluations, are discussed in this section. The gross sedimentation rate (settling rate) was estimated from sediment trap data and provides a measurement of the flux of suspended solids through the water column. The net sedimentation rate was estimated from sediment cores dated with radioisotopes (Cs-137 or Pb-210) or chemical tracers which can be correlated with specific historical events (i.e., mercury in Bellingham Bay). Net sedimentation describes the rate at which sediments are permanently incorporated into the seabed. The difference between gross sedimentation rates and net sedimentation rates provides information on the rate at which bottom sediments are resuspended to the overlying water column where they may be subject to horizontal advection or resettling. A detailed description of the dating techniques applied to the natural recovery cores is presented in Appendix A.

Sediment in the natural recovery cores has been subjected to both coring-induced compaction (an artifact of the sampling process) and burial-induced compaction (the natural consolidation of sediments). The effect of sampling-induced compaction was removed from the data, and actual sampling depths were reconstructed based on the ratio of core penetration to core recovery (see Appendix A).

Sedimentation rates are often presented in mass-based accumulation units (g/cm^2-yr) to implicitly account for burial-induced compaction and porosity reduction with depth in the sediment. However, the density gradients in the natural recovery cores are slight (see Section 3.6); therefore, sedimentation rate calculations were performed using length-based units (cm/yr) without introducing significant errors. Length-based units were preferred for the following reasons: (1) the point of compliance for biological effects is defined on the basis of length, not mass, and is typically assumed to be the depth of the biological mixing zone (approximately 10 cm); and (2) length-based sedimentation rates are simpler, more intuitive, and more easily compared to geologic events in the sediment stratigraphy.

9.2.1 Net Sedimentation Rates

Lead-210. Depth profiles of Pb-210 are presented on Figure 9-1. Net sedimentation rates can be calculated from Pb-210 activity based on a model of constant and uniform sediment accumulation (Battelle, 1995). Sediment accumulation rates, however, are affected by seasonal variations in sedimentation resulting from river discharges, vessel traffic, and biological

activities, as well as long-term variations resulting from changing land use patterns in the watersheds. Therefore, the interpretation of Pb-210 profiles is often subject to model assumption violations, particularly in shallow urban waterways such as inner Bellingham Bay (see Appendix D). Non-uniform sedimentation probably accounts for much of the observed scatter in the profiles, although radioisotope counting errors also contribute to the uncertainty.

The supported Pb-210 activity for the natural recovery cores was estimated to be 0.75 disintegrations per minute per gram (dpm/g). This estimate is based on the range of published, supported Pb-210 values (0.5 to 1 dpm/g) typical for Puget Sound sediments (Battelle, 1995). We used an estimated value for the supported Pb-210 activity because a baseline Pb-210 value could not be established with certainty in the lower sections of the cores; Pb-210 deeper analyses would have been required. The estimated supported value of 0.75 dpm/g is believed to be representative of Bellingham Bay conditions.

The net sedimentation rate was calculated from the slope of natural logarithm of excess Pb-210 activity versus depth below the mixing layer (See Appendix D). The slope was statistically determined using linear regression techniques. The estimated Pb-210 sedimentation rates ranged from 1.4 to 2.07 centimeters per year (cm/yr) (Table 9-1). These rates are generally consistent with sedimentation rates estimated using Cs-137 or mercury, as described below.

Cesium-137. Cesium-137 has entered the oceans over the last 50 years as the result of nuclear weapons testing. The peak in Cs-137 profiles is believed to reflect the major global input of Cs-137 to the earth's atmosphere during the period of active bomb testing, and is correlated with a date of 1962. An additional index depth is the point where Cs-137 concentrations begin to increase sharply from a background or non-detectable concentrations to measurable concentrations. This point can be time labeled because Cs-137 is anthropogenic in origin and no background concentrations occurred in sediments prior to the nuclear weapon testing. The depth representing the onset of the introduction of Cs-137 to the sediments is correlated with 1950. Profiles of Cs-137 in the natural recovery cores are presented on Figure 9-2.

The sedimentation rates calculated from the Cs-137 profiles using both of the time indices (i.e., the onset and the peak of atmospheric fallout) are presented in Table 9-1. The estimated sedimentation rates were generally consistent between the natural recovery cores and ranged from 1.52 to 1.99 cm/yr based on the introduction of Cs-137 activity, and from 1.43 to 1.52 cm/yr based on the peak of Cs-137 activity. These sedimentation rates are generally consistent with the estimates derived using Pb-210 or mercury profiles. Modern sedimentation rates appear to be relatively stable, based on

consistency across different datums, and thus are appropriate for use in future projections.

Mercury. Selected subsamples from each natural recovery core were analyzed for total mercury. Mercury was selected as a chemical tracer because it is a primary constituent of concern in Bellingham Bay and the period of maximum discharge to the bay is well-documented. Maximum discharges of mercury to Bellingham Bay occurred between 1965 and 1970 (Bothner et al., 1980). Year 1970 was used as the datum in our analysis to correspond with the maximum sediment mercury concentrations in natural recovery cores, allowing a few years of lag time for mercury to flux through the water column and become incorporated in the sediment. Profiles of sediment mercury concentrations are presented on Figure 9-3.

Estimated sedimentation rates based on the mercury profiles and our knowledge of historical mercury discharges are presented in Table 9-1. Estimated sedimentation rates are generally consistent among the natural recovery cores and range from 1.54 to 1.98 cm/yr. These sedimentation rates are also consistent with estimates based on radioisotopic dating methods.

Depth to Dredge Horizons. Sedimentation rates in the inner bay and waterways can also be estimated by considering the date of the deepest dredging event and the depth to that dredging horizon (i.e., depth to native sediments). In the inner bay, sedimentation rates calculated from the thickness of the post-dredge layer compare well with sedimentation rates calculated using radioisotopic methods. Channel deepening of the outer part of the Whatcom Waterway to elevation -35 feet MLLW occurred in 1969. Since that time, approximately 1 to 3 feet of recent sediments have accumulated. Over a 27-year period (RI surveys were conducted in 1996), this thickness of sediment corresponds to sedimentation rates between 1.1 and 3.4 cm/yr, in good agreement with isotopically determined rates between 1.5 and 1.8 cm/yr.

The primary deepening event within the Whatcom Waterway occurred in 1960. Since that time, about 2 to 10 feet of sediment (5 feet on average) has accumulated, corresponding to sedimentation rates of 1.7 to 8.5 cm/yr (4.2 cm/yr on average). These rates should be considered minimum estimates because they do not account for material that was removed during a maintenance dredging event in 1974. In the I & J Street Waterway, about 3 to 6 feet of sediment has accumulated since the waterway was deepened in 1966, corresponding to sedimentation rates between 3.1 and 6.1 cm/yr. These also represent minimum estimates because a partial removal of material occurred during maintenance dredging in 1992.

Summary of Net Sedimentation Rates. Average net sedimentation rates were calculated using the mean of the four estimation methods: (1) onset of

Cs-137 activity, (2) peak of Cs-137 activity, (3) Pb-210 decay, and (4) peak mercury concentration, as summarized in Table 9-1. Average sedimentation rates calculated for inner Bellingham Bay are generally consistent among the three natural recovery cores and range from 1.52 cm/yr at HC-NR-100 and HC-NR-101 to 1.77 cm/yr at HC-NR-102. Because the uncertainty in the sedimentation rate estimates is about ±0.5 cm/yr, based on the variability between estimation methods, the slightly higher rate at HC-NR-102 is probably not significant.

Averaging sedimentation rates across several dating methods provides a degree of corroboration, and also reduces the uncertainty of the rate estimate. Each of the isotopic dating methods contains a higher degree of uncertainty when considered independently. Much of this uncertainty probably derives from deviations in the assumption of constant and uniform sedimentation over time. Uncertainty is introduced by variable sedimentation rates, variable grain size and sediment density, analytical and radionuclide counting errors, and disturbance of the seafloor by propwash, anchor drag, or construction events. Such problems are common in shallow marine environments. The profiles in core HC-NR-101 are especially difficult to interpret (Figures 9-1 through 9-3), as this area may have been disturbed by the construction of G-P's biotreatment lagoon or outfall diffuser line.

9.2.2 Gross Sedimentation Rate

Gross sedimentation, or particle settling rate, is the sum of the net sedimentation and sediment resuspension. Gross sedimentation rates were determined by measuring the flux of particulate matter into sediment traps deployed about one meter above the seabed. Gross sedimentation rates are often higher than net sedimentation rates, because only a fraction of the particles settling through the water column are permanently incorporated into the seabed.

As part of the RI sampling program, two sediment traps (HC-ST-100 and HC-ST-101) were deployed in inner Bellingham Bay for three periods, each of approximately four months duration (see Figure 2-2). The entire deployment period spanned from October 1996 to September 1997; however, sediment trap HC-ST-101 tipped over during the second deployment period and no sample was recovered. A more complete description of sediment trap deployment, recovery, and sample handling procedures is presented in Appendix A. Grain size analyses of settling particulate matter (SPM) are also presented in Appendix A. Validated chemical analytical results of SPM are presented in Appendix B.

Particle mass accumulation rates were generally consistent between the two sediment trap locations. Mass accumulation rates ranged from 3.69 to 9.59 g/cm^2-yr, and from 3.55 to 9.16 g/cm^2-yr at locations HC-ST-100 and HC-ST-

101, respectively. Surface sediment dry densities in collocated natural recovery cores were used to convert from mass-based accumulation units (g/cm^2-yr) to length-based units (cm/yr). The dry density of surface sediments at the sediment trap locations is 0.47 g/cm^3 at HC-NR-100 and 0.42 g/cm^3 at HC-NR-101. Thus, estimated gross sedimentation rates ranged from 7.85 to 20.4 cm/yr, and from 8.45 to 21.8 cm/yr at locations HC-ST-100 and HC-ST-101, respectively (Table 9-2).

As evidenced by the seasonal data, gross sedimentation rates varied by almost a factor of three between the fall/winter and summer deployment periods. Higher settling rates in summer may be caused by a more direct influence from Nooksack River runoff, which is carried to the site in clockwise, fair-weather circulation patterns that are more typical of summer months. Settling of suspended sediments from the turbid river plume is apparently enhanced during this time period. During winter months, prevailing counter-clockwise circulation patterns deflect the river plume toward Lummi Peninsula and away from the site, resulting in lower settling rates.

9.2.3 Resuspension Rate

Resuspension rates were estimated by the difference between gross sedimentation rates measured in sediment traps and net sedimentation rates measured in dated cores [Resuspension = (Gross SR - Net SR)/Gross SR] (Baker et al., 1991). Resuspension describes the continuous exchange of sediments between the seabed and water column. Gross and net sedimentation rates in paired sediment trap and natural recovery core locations are summarized in Table 9-2. The average of the net sedimentation rates estimated using the four different dating techniques was used in the resuspension rate calculations. Resuspension rates ranged from 81 to 93 percent throughout the year, averaging about 90 percent at both locations.

9.2.4 Mixed Layer Thickness

Mixing within the sediment column is a result of bioturbation and tidal or propeller-induced currents. The thickness of the surface mixed layer was interpreted from plots of the natural logarithm of excess Pb-210 activity with depth. The depth at which the Pb-210 activity indicates steady-state decay behavior (constant decrease with depth in the log activity) corresponds to the bottom of the mixed layer; within the mixed layer, Pb-210 activity is theoretically constant. In these cores, however, Pb-210 activity in the mixed layer is erratic, and may be complicated by propwash, anchor drag, construction events, and other bottom disturbances. Based on the Pb-210 profiles (Figure 9-1), the base of the mixed-layer is estimated to be 24 cm in core HC-NR-100, 15 cm in core HC-NR-101, and 11 cm in core HC-NR-102. Although the Pb-210 profiles contain a large degree of scatter, these values

are in general agreement with studies conducted in other Puget Sound embayments (Battelle, 1995).

9.3 Sediment Natural Recovery

Mercury. Depth profiles of total mercury concentration in the natural recovery cores show a significant decrease in sediment concentrations since the peak discharge period of 1965 to 1970 (Figure 9-3). These data agree with previously published data on sediment mercury trends in Bellingham Bay (Bothner, 1973; Bothner et al., 1980; and Officer and Lynch, 1989) and indicate that the surface sediment mercury concentrations are naturally declining. This recovery is driven in part by the significant reduction in source inputs beginning in 1970. Following source controls, sediment concentrations have been reduced by natural processes including: burial of contaminated sediment with cleaner sediment; mixing of cleaner surface sediments with deeper sediments by burrowing organisms and bottom currents; and exchange of sediments with the overlying water column through resuspension.

Sediment mercury concentrations during the period following source control are presented on Figure 9-4. These plots show the decline in mercury concentrations since the peak discharge period that ended in 1970. Future reductions in total mercury concentration were projected forward over a ten-year period by fitting an exponential decay curve to the core profiles during the recovery period. A summary of this analysis is presented in Table 9-3.

Based on this exponential decay model, we estimate that mercury concentrations in the surface sediments of inner Bellingham Bay will decrease by an additional 30 to 40 percent over the next ten years. However, these recovery projections are based on present-day contaminant inputs to the inner bay, which include inputs from the resuspension of elevated concentrations of mercury in WW Area sediments. Remedial dredging or capping of sediments in Whatcom Waterway would reduce the mercury concentration in resuspended particulate matter that is circulating in the inner bay, and therefore accelerate the recovery process. Estimated sedimentation rates in the inner bay indicate that mercury and other contaminants are subject to relatively rapid burial (approximately 16 cm over a ten-year period) and thus isolation from the overlying water column and marine organisms.

Similar reductions in mercury concentration can be discerned by evaluating trends in surface sediment quality over time, i.e., comparing previously collected surface sediment data (see Hart Crowser, 1996b, Figure 4-2) with the data collected during the RI. Compared to trend analysis within a single core, however, trend analysis of historical surface sediment quality contains many more uncertainties that confound interpretations, and make the analysis

qualitative but not quantitative. Such errors include (1) spatial heterogeneity in physical and chemical properties of sediments; (2) inter-laboratory variability between investigations; and (3) differences in sampling methods between investigations, in particular, differences in sampling depth. Although reductions in mercury concentrations can be observed in many areas, the amount of reduction that has occurred cannot be precisely defined.

This preliminary evaluation shows significant reductions in mercury concentrations over the last several decades and good potential for continued future improvements in sediment quality. Sediment natural recovery is further considered in the Feasibility Study (FS) as one component of an overall cleanup plan. Additional natural recovery modeling may be performed in the FS as necessary to support the evaluation of remedial alternatives.

Table 9-1 Summary of Estimated Sedimentation Rates

	Sedimentation Rate in cm/yr				
Natural Recovery Core Number	Pb-210 Decay	Onset of Cs-137 (1950)	Peak of Cs-137 (1962)	Peak of Mercury (1970)	Average Sedimentation Rate
HC-NR-100	1.40	1.69	1.43	1.54	1.52
HC-NR-101	1.06	1.99	1.41	1.61	1.52
HC-NR-102	2.07	1.52	1.52	1.98	1.77
				Inner Bay Average Sedimentation Rate	1.60

Table 9-2 Summary of Sedimentation and Resuspension Rates

	Net Sedt. Rate in cm/yr	Gross Sedt. Rate in cm/yr	Resuspension Rate
HC-NR-100/HC-ST-100			
Oct-96 to Jan-97	1.52	7.85	81%
Feb-97 to May-97	1.52	9.00	83%
May-97 to Sep-97	1.52	20.40	93%
Average(1)	1.52	14.13	89%
HC-NR-101/HC-ST-101			
Oct-96 to Jan-97	1.52	8.45	82%
Feb-97 to May-97	1.52	N/R	N/R
May-97 to Sep-97	1.52	21.80	93%
Average(1)	1.52	15.13	90%

Notes:

N/R = Sediment trap was tipped over and sample was not recovered.

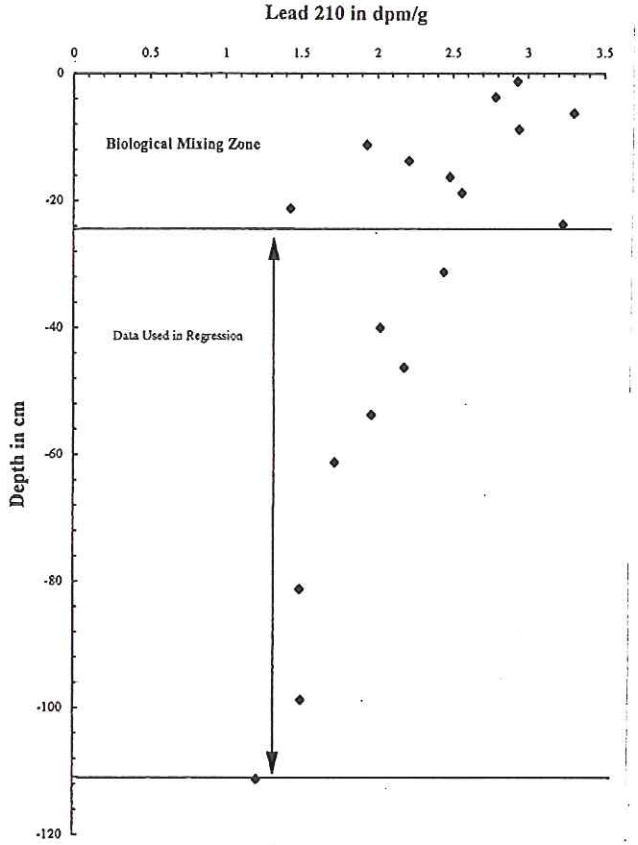
(1) Includes average of first and third deployments.

Table 9-3 Ten-Year Recovery Projections of Mercury Concentrations in Inner Bellingham Bay

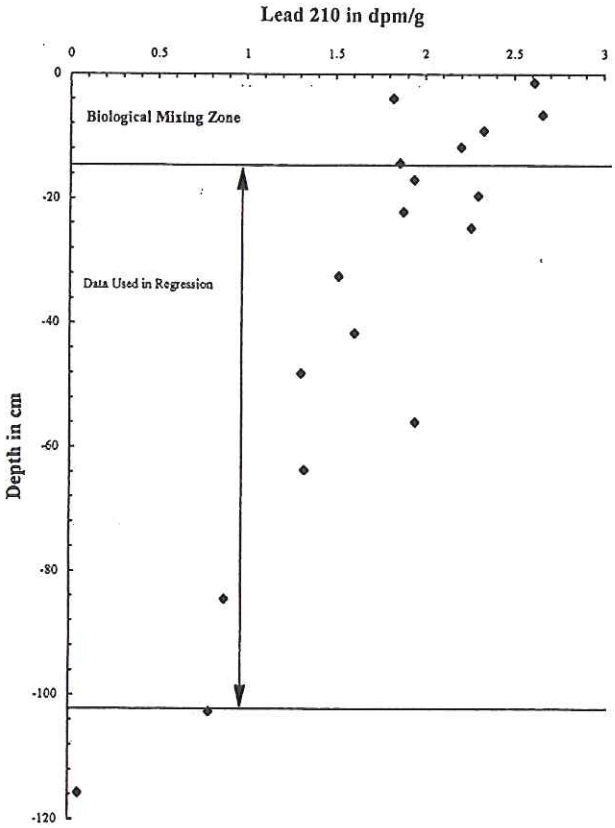
						Sediment Mercury Concentration in mg/kg	
Natural Recovery Core Number	Maximum Sample Interval Used in Regression in cm	Number of Samples Used in Regression	Regression R^2 Value	Standard Error	Average Net Sedimentation Rate in cm/yr	Year 1995	Year 2005 (+/- 2 Standard Error)
HC-NR-100	-44.2	11	0.59	0.059	1.52	1.3	0.80 (+/- 0.12)
HC-NR-101	-41.6	12	0.68	0.075	1.52	1.7	1.12 (+/- 0.15)
HC-NR-102	-45.6	13	0.81	0.057	1.77	0.34	0.23 (+/- 0.11)

Lead–210 Core Profiles

HC-NR-100



HC-NR-101



HC–NR–102

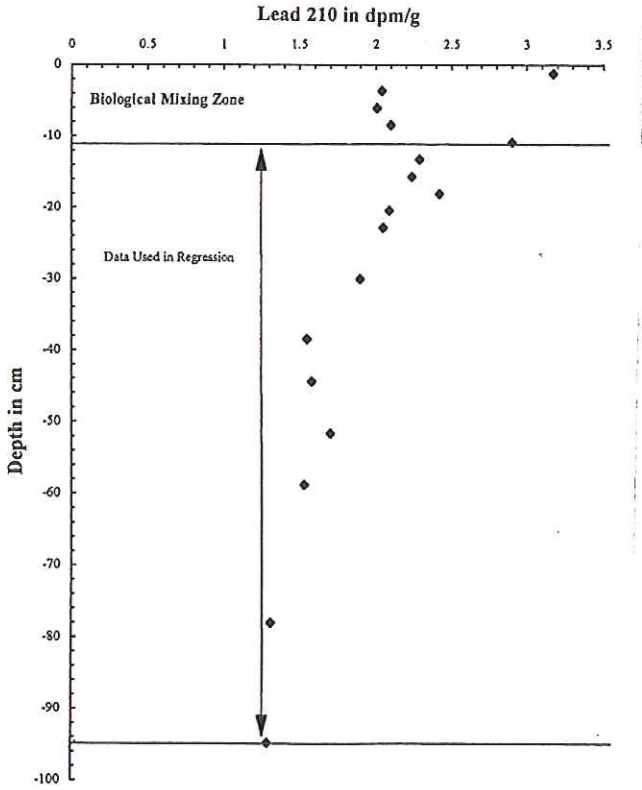


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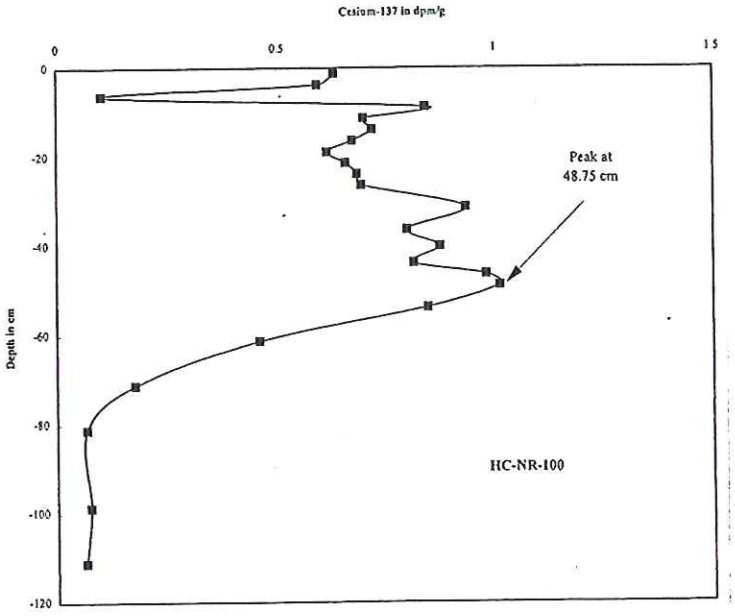
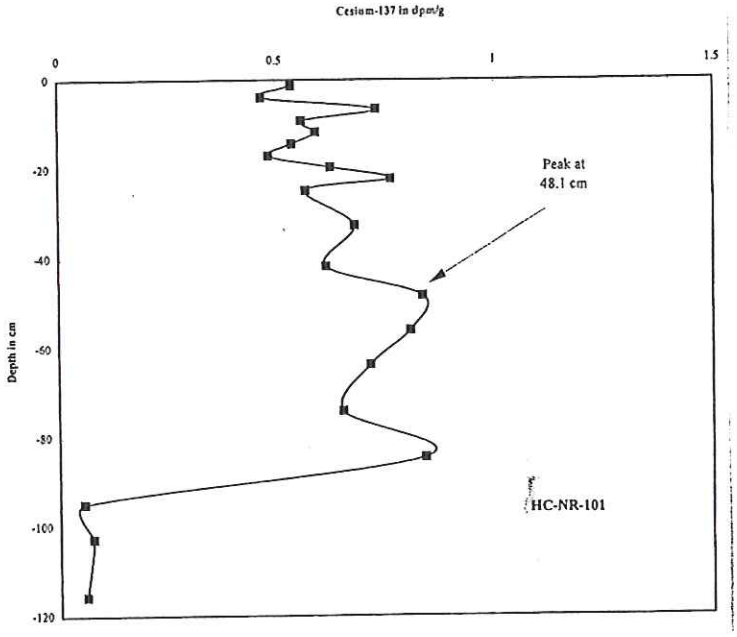
Figure 9–1

Cesium-137 Core Profiles

HC-NR-100



HC-NR-101



HC-NR-102

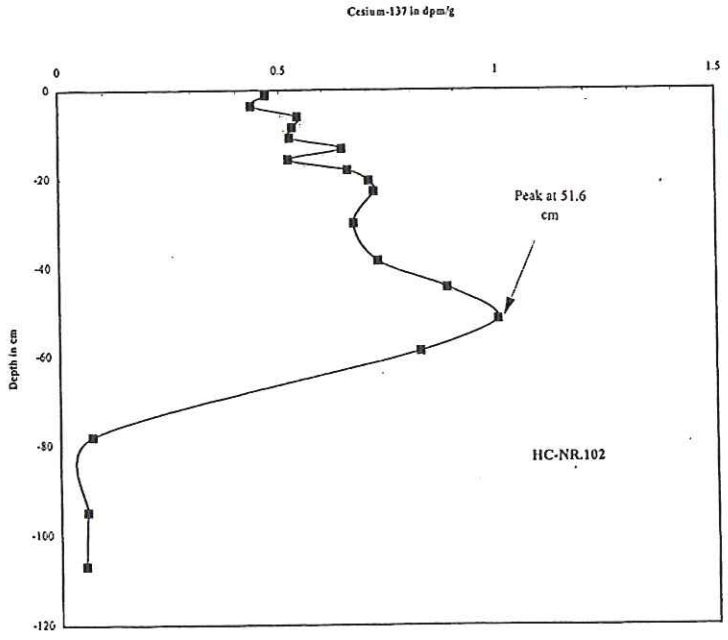


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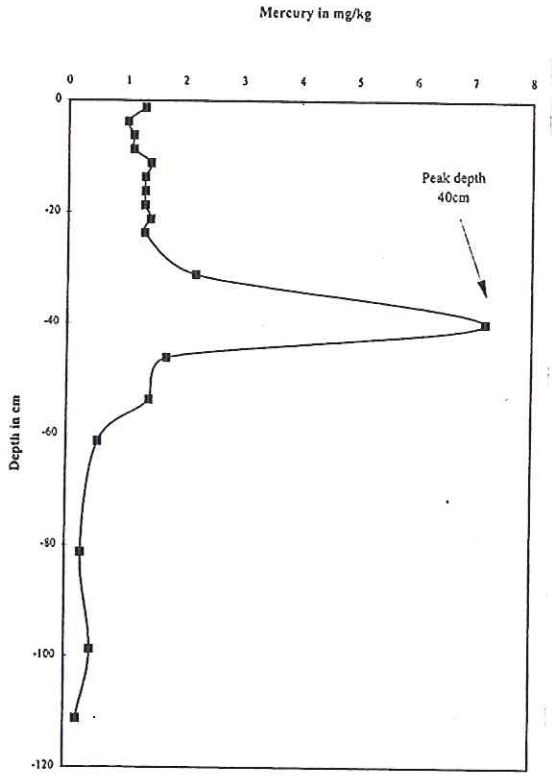
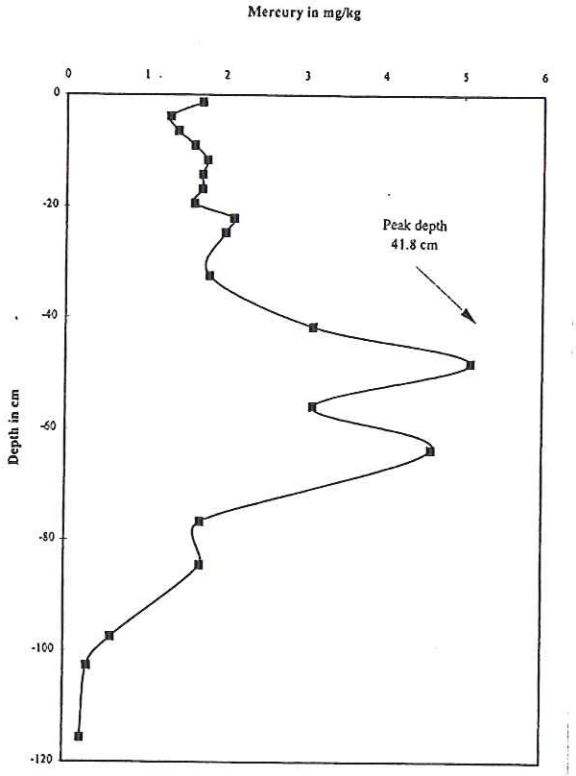
Figure 9-2

Mercury Concentration Core Profiles

HC-NR-100



HC-NR-101



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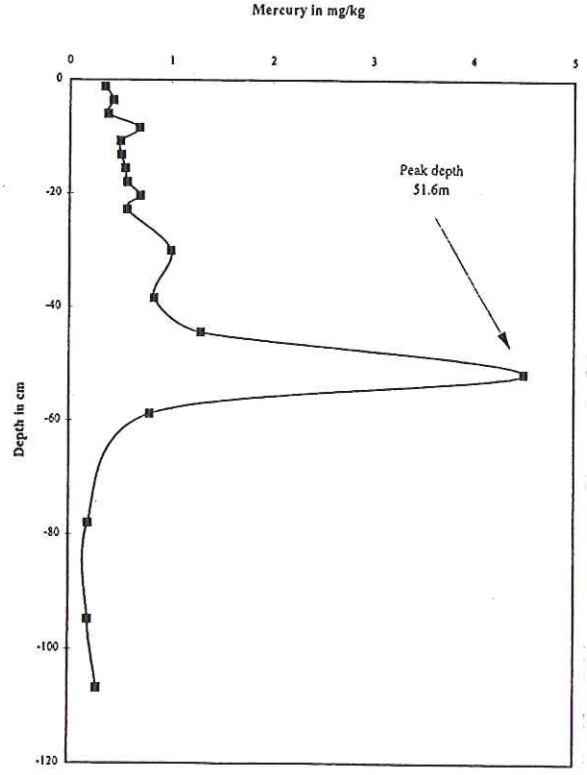

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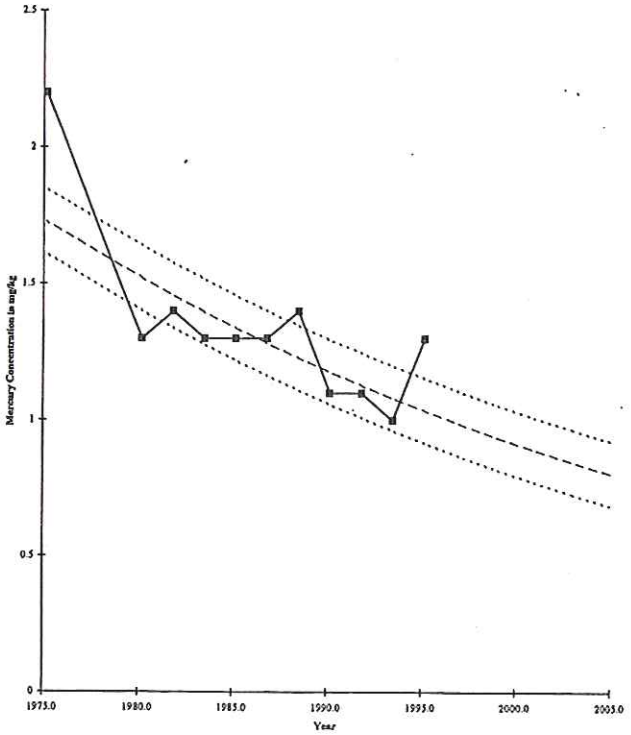
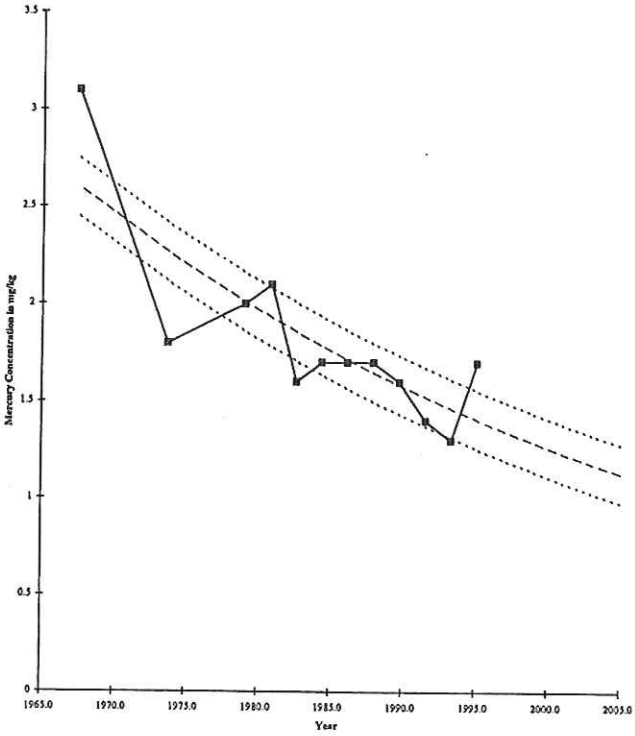
Figure 9-3

Projected Mercury Concentration Trends
Inner Bellingham Bay

HC-NR-100



HC-NR-101



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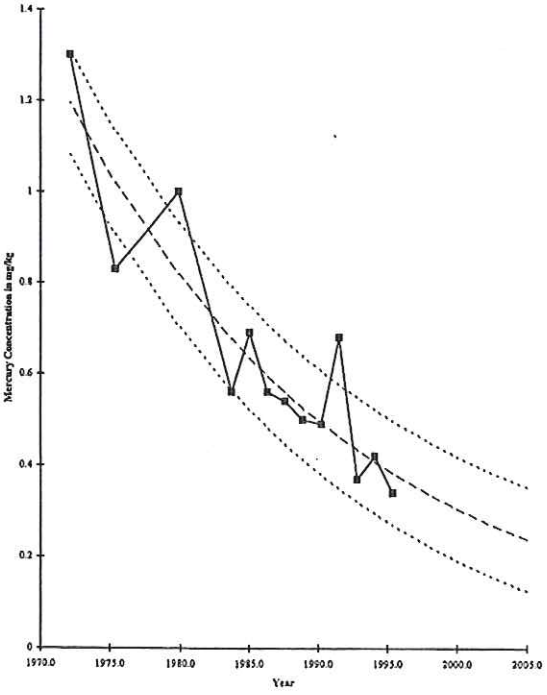

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Figure 9–4