Exponent™

Ward Cove Sediment Remediation Project

Detailed Technical Studies Report

Volume I
Remedial Investigation and Feasibility Study

Prepared for

Ketchikan Pulp Company
Ketchikan, Alaska

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Detailed Technical Studies Report Volume I, Remedial Investigation and Feasibility Study

Prepared for

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ACRONYMS AND ABBREVIATIONS

2,3,7,8-TCDD	2,3,7,8-tetrachlorodibenzo-*p*-dioxin
2,3,7,8-TCDF	2,3,7,8-tetrachlorodibenzofuran
ACES	Automated Coastal Engineering System
ADEC	Alaska Department of Environmental Conservation
ADFG	Alaska Department of Fish and Game
AET	apparent effects threshold
AOC	area of concern
AOF	area of focus
APC	Alaska Pulp Corporation
ARAR	applicable or relevant and appropriate requirement
AVS	acid-volatile sulfide
AWPCB	Alaska Water Pollution Control Board
BMP	Best Management Practice
BOD	biochemical oxygen demand
BSAF	biota-sediment accumulation factor
CAD	confined aquatic disposal
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act of 1980
CLP	Contract Laboratory Program
CoC	chemical of concern
COD	chemical oxygen demand
CoPC	chemical of potential concern
Corps	U.S. Army Corps of Engineers
cy	cubic yard
DGPS	differential global positioning system
DRET	dredging elutriate test
DTSR	detailed technical studies report
EFDC	Environmental Fluid Dynamics Code
EPA	U.S. Environmental Protection Agency
ESP	electrostatic precipitator
ICP-AES	inductively coupled plasma-atomic emission spectrometry
KPC	Ketchikan Pulp Company
LC50	concentration lethal to 50 percent of the test population
LOAEL	lowest-observed-adverse-effect level
LTF	log transfer facility
MCUL	minimum cleanup level
MET	modified elutriate test
mgd	million gallons per day
MLLW	mean lower low water
NCDF	near-shore confined disposal facility

NCP	National Contingency Plan
NOAEL	no-observed-adverse-effect level
NPDES	National Pollutant Discharge Elimination System
OCDD	octachlorinated dibenzo-*p*-dioxin
OSHA	Occupational Safety and Health Administration
PAH	polycyclic aromatic hydrocarbon
PCDD/F	polychlorinated dibenzo-*p*-dioxin and polychlorinated dibenzofuran
PSDDA	Puget Sound Dredged Disposal Analysis
PSEP	Puget Sound Estuary Program
RAO	remedial action objective
RI/FS	remedial investigation and feasibility study
RPC	relative potency concentration
rpm	revolutions per minute
SEM	simultaneously extracted metals
SQS	sediment quality standard
TBC	to-be-considered
TEC	toxic equivalent concentration
TEF	toxicity equivalence factor
TOC	total organic carbon
TPH	total petroleum hydrocarbon
TRV	toxicity reference value
TSS	total suspended solids
TVS	total volatile solids
USFWS	U.S. Fish and Wildlife Service
$WCSQV_{(1)}$	Ward Cove sediment quality value analogous to sediment quality standard
$WCSQV_{(2)}$	Ward Cove sediment quality value analogous to minimum cleanup level

EXECUTIVE SUMMARY

OVERVIEW AND OBJECTIVES

In September 1995, Ketchikan Pulp Company (KPC) entered into a Consent Decree with the U.S. Environmental Protection Agency (EPA) to address environmental issues related to KPC's Ketchikan facility located on the shoreline of Ward Cove, Alaska. As part of the Consent Decree, KPC agreed to conduct a Ward Cove sediment remediation project to address sediments in the Cove. A technical studies work plan for the Ward Cove sediment remediation project was submitted to EPA in April 1996. The technical studies work plan described the studies and actions necessary to identify an appropriate remedy to address ecological and human health issues associated with Ward Cove sediments. An overview of the Ward Cove technical studies is provided in Figure ES-1.

The technical studies were conducted in two phases. In May and June of 1996 (Phase 1), surface sediments were sampled at 28 stations throughout Ward Cove and at 2 stations in a reference area (Moser Bay, Alaska) to characterize the horizontal distribution of chemicals of potential concern (CoPCs) and sediment toxicity throughout the Cove. Ecological and human health evaluations of the Phase 1 data were conducted to communicate the implications of the data to regulators and to build consensus on the appropriate evaluation techniques. The Phase 1 report (PTI 1997g) identified the CoPCs and areas of focus that warranted further study in Phase 2.

In March 1997, pulping activities at the KPC facility were terminated. This action ultimately resulted in an Administrative Order on Consent (Consent Order) between KPC, EPA, and the Alaska Department of Environmental Conservation (ADEC). Under the Consent Order, the KPC facility is divided into two operable units: the Marine Operable Unit and the Uplands Operable Unit. The Marine Operable Unit includes all of Ward Cove and other marine areas where contaminants have migrated from the KPC facility and may pose a threat to human health and the environment. The 1995 Consent Decree addresses Ward Cove. An uplands remedial investigation and feasibility study (RI/FS) is being prepared in response to the Consent Order.

The primary objective of the Ward Cove sediment remediation project is to determine the extent to which sediments in Ward Cove may pose risks of adverse effects to humans and ecological receptors and therefore potentially warrant remediation. The primary objectives of this report are to:

- Provide the detailed results of the technical studies required by the Consent Decree
- Satisfy the requirements of an RI/FS under the Comprehensive Environmental Response, Compensation and Liability Act of 1980

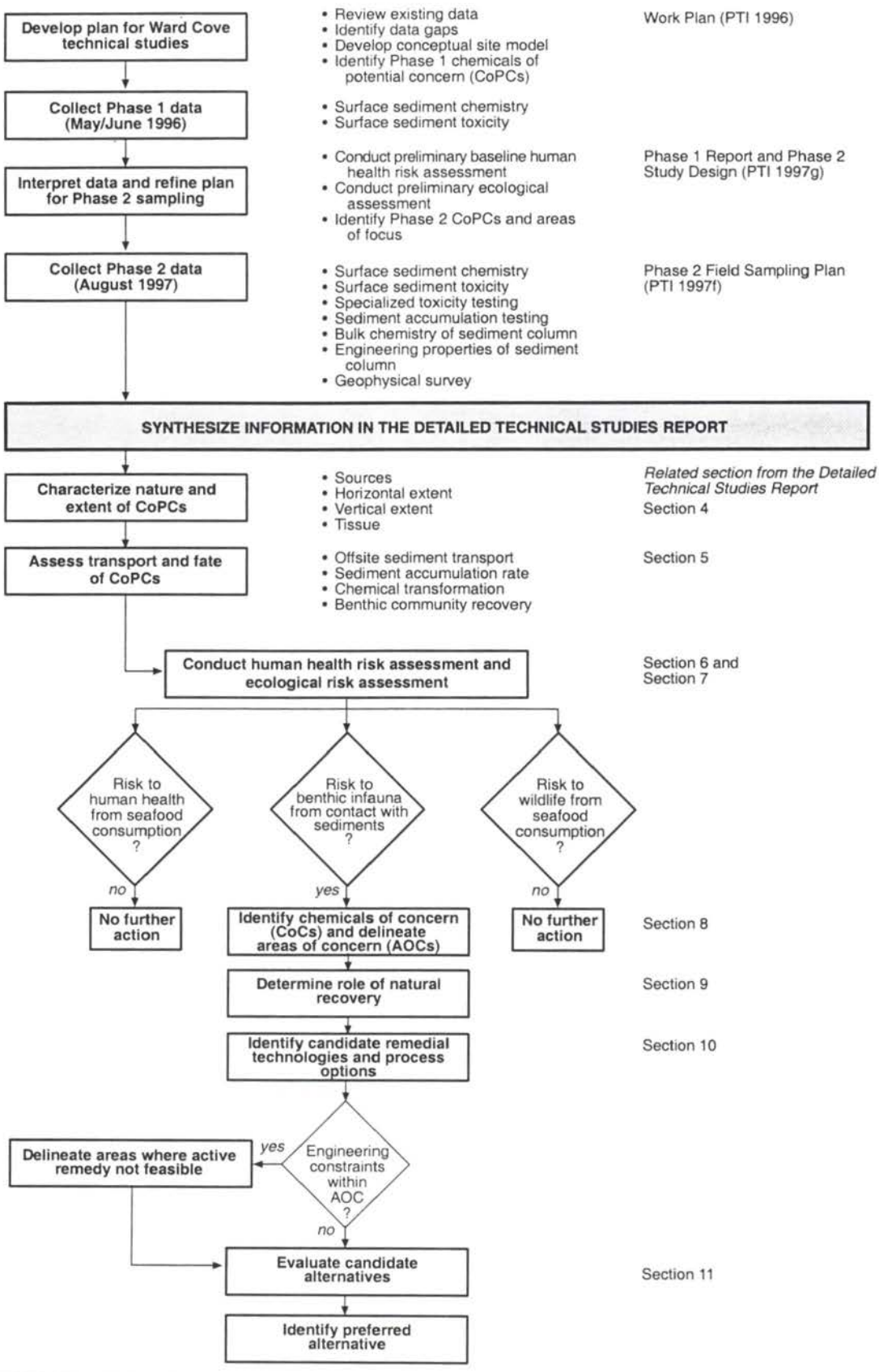

Figure ES-1. Overview of Ward Cove detailed technical studies.

- Satisfy the National Pollutant Discharge Elimination System (NPDES) requirements for KPC's 1997 sediment monitoring program.

This report builds on the Phase 1 report (PTI 1997g) and consolidates all evaluations relevant to site characterization and remedy development.

NATURE AND EXTENT OF CHEMICALS OF POTENTIAL CONCERN

There are three categories of CoPCs: 1) CoPCs for ecological risks associated with sediment toxicity, 2) CoPCs for ecological risks associated with food-web bioaccumulation, and 3) CoPCs for human health risks associated with food-web bioaccumulation. The following CoPCs have been identified:

- **Substances Associated with Organic Matter and Organic Matter Degradation**—Total organic carbon (TOC), ammonia, sulfide, biochemical oxygen demand (BOD), chemical oxygen demand (COD), phenol, and 4-methylphenol
- **Metals**—Arsenic, cadmium, mercury, and zinc
- **Organic Compounds**—Polycyclic aromatic hydrocarbons (PAHs), and polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) (PCDDs and PCDFs are referred to collectively as PCDDs/Fs or as dioxin/furan congeners in this report).

The rationale for selecting these chemicals is described in greater detail in the work plan (PTI 1996) and in the Phase 1 report (PTI 1997g). Based on a rigorous evaluation of their potential risk to human health and ecological receptors, many of these CoPCs were screened out and were not evaluated during the Phase 2 sampling effort (PTI 1997g). Phase 2 CoPCs include ammonia, sulfide, phenol, and 4-methylphenol. These CoPCs are products of microbial degradation processes that occur naturally in the sediment. Organic matter (i.e., woody material and wood by-products) historically released at the mill is a primary source material, or fuel, for these processes. TOC, BOD, and COD are also included as CoPCs; however, they are not considered problem chemicals or causative agents for toxicity. They are included as CoPCs because they are general indicators of the condition (elevated levels of organic matter) that leads to adverse effects on the benthic community.

The concentrations of most of the CoPCs throughout large portions of the Cove exceed the concentrations found in Moser Bay. The highest concentrations of many of the CoPCs were found near the KPC facility or the cannery. There are differences from year to year in the distributions of some, but not all, CoPCs. The greatest differences occur for those CoPCs that may be susceptible to seasonal changes in biological activity (e.g., ammonia, 4-methylphenol). Concentrations of CoPCs in Ward Cove intertidal sediments were negligible.

Visual observations were made of 16 deep sediment cores for sediment column characterization and of 2 deep sediment cores for sediment accumulation testing that were collected in Ward Cove. Those observations and the associated chemical data indicate that impacts to sediment from activities at the KPC facility, including log handling, have resulted in a black, organic-rich layer of sediment that contains wood debris. This layer of sediment is concentrated near the head of the Cove offshore of the KPC facility and along the north shore, and it generally ranges in thickness from 4 to 10 ft. This layer is distinguished from native sediment by higher concentrations of TOC, BOD, COD, ammonia, sulfide, phenol, and 4-methylphenol. Grain size differences between native and non-native sediment suggest that organic matter in the non-native sediment consists of particles of the size of medium to coarse sand.

The concentrations of metals and dioxin/furan congeners in bulk sediment (i.e., composite samples representing conditions throughout the affected horizon of the deep cores) are lower than those in surface sediments (i.e., the upper 10 cm [4 in.]); the concentrations of organic carbon, total sulfide, and BOD in bulk sediment are similar to those in surface sediments; and the concentrations of ammonia, phenol, and 4-methylphenol in bulk sediment are greater than those in surface sediments.

Prior to 1997 when the pulp mill was active, an important water quality concern in Ward Cove was potential oxygen depletion associated with the discharge of oxygen-demanding substances in the pulp effluent. This concern was addressed through effluent handling and treatment modification and was monitored as part of the NPDES program. Issues regarding potential oxygen depletion in bottom water and its possible relationship to oxygen depletion in sediment were expressed in agency comments on the draft detailed technical studies report (DTSR). The report has been modified as necessary to address these concerns.

The oxygen content of seawater is affected by a variety of processes, including gas exchange with the atmosphere (to produce oxygen saturation at the sea surface), oxygen production by photosynthetic organisms (which adds oxygen to seawater in the zone where light penetration is sufficient), and oxygen depletion during organic matter decomposition (which removes oxygen in the deeper waters as dead organisms fall through the water and are decomposed). The mixing of surface water (where oxygen is usually abundant) and deeper water (where oxygen is often depleted) is influenced by seasonal changes in the density structure of water.

NPDES monitoring data from Ward Cove provide a detailed record of horizontal and vertical concentrations of oxygen in the Cove over the last several years and reflect seasonal changes in the processes described above. Oxygen concentrations generally decrease with depth; seasonal depressions at depth in some portions of the Cove are most pronounced in late summer when density stratification in the water column limits mixing. Oxygen concentrations in Ward Cove were typically above 8 mg/L.

TRANSPORT AND FATE OF CHEMICALS OF POTENTIAL CONCERN

The two principal influences on transport of sediment-bound CoPCs in Ward Cove are the introduction of fresh water near the head of the Cove and the influence of tidal currents. The introduction of fresh water produces a bilayer flow, with net outflow from the inner Cove in the surface water and a net inflow to the inner Cove in the bottom water. Modeling of tidal currents indicates the presence of a counterclockwise circulation in the outer Cove, with net inflow near the southern shoreline and net outflow near the northern shoreline. As a consequence, CoPCs introduced into surface water anywhere in the Cove are expected to move toward the mouth of the Cove along the northern shoreline.

The combined effect of these two processes—transport of CoPCs on settleable solids and outflow along the northern shore of the Cove—is reflected in the distributions of CoPCs in the sediment. Elevated concentrations of CoPCs are generally not found in the outer Cove, with the exception of a narrow region near the north shore. Particle-associated sediment contaminants are not expected to be remobilized and exported from the Cove (PTI 1997b).

Several of the most important CoPCs in Ward Cove sediments are both produced and degraded by processes that occur naturally in the sediments. The woody material that was both raw material and product of the KPC mill is a primary source material and "fuel" for these processes. Microbially mediated decomposition of the woody material and wood by-products leads to oxygen depletion and production of ammonia, sulfide, and 4-methylphenol in the sediment. The resulting conditions affect the sediment's suitability as habitat for other organisms. However, both abiotic processes (e.g., porewater diffusion) and biotic processes (e.g., sediment irrigation) can act to mitigate the harmful effect of these conditions. The sediment quality in Ward Cove is therefore established by the interplay of multiple processes acting on several different sediment characteristics.

HUMAN HEALTH RISK ASSESSMENT

A human health risk assessment was conducted to identify potential risks posed by chemicals detected in sediments or seafood from Ward Cove. The consumption of seafood that may bioaccumulate chemicals from sediments was identified as the only complete exposure pathway by which humans could be exposed to chemicals from the sediments or seafood from the Cove. Direct contact with sediments in Ward Cove is unlikely because of the depth of the water overlying affected sediments and the cold climate. While recreational use of the lower portion of Ward Creek could result in contact with sediments, transport of site-related CoPCs to this area is not expected. Thus, direct human contact with CoPCs in sediments is highly unlikely. However, to provide a worst-case analysis, this pathway is addressed in the uncertainty assessment.

Potential human health risks associated with chemicals in Cove sediments were evaluated using two data sets: 1) tissue concentrations estimated through application of biota-sediment accumulation factors (BSAFs) to the maximum concentrations of chemicals detected in sediments near the KPC facility; and 2) tissue concentrations reported by

previous investigators for PCDDs/Fs and mercury in seafood from Ward Cove and Tongass Narrows. Estimated concentrations were consistently higher than measured concentrations. Both estimated and measured tissue concentrations were conservatively used to determine whether any chemicals detected in Cove sediments pose a risk to human health (i.e., to identify whether any chemicals in sediments were chemicals of concern [CoCs] for human health).

Maximum estimated or measured tissue concentrations were compared with available background concentrations for arsenic or PCDDs/Fs (no representative background tissue concentration data were identified for other chemicals). Maximum estimated seafood concentrations for arsenic were lower than background concentrations identified in the contiguous United States. Maximum estimated and measured concentrations of PCDDs/Fs were elevated over background concentrations in tissues collected in Alaska.

Maximum estimated and measured tissue concentrations were compared with available background concentrations and risk-based concentrations for chemicals in seafood derived using EPA guidance and site-specific seafood consumption rates for a local community with high dietary reliance on seafood (i.e., a subsistence community). Although application of subsistence-level consumption rates (i.e., 65 g/day of fish and 11 g/day of shellfish) greatly overestimates risks to the general population, these rates were used to provide a protective means of evaluating risks for all hypothetical current or future site users. Risk-based concentrations also incorporated a fractional intake of 5 percent to account for the fraction of all fishing that might occur in affected areas of Ward Cove and for the reduced bioaccumulation potential resulting from the migratory nature of salmon, the most popular species ingested by local anglers.

For carcinogens, risk-based concentrations were calculated using a target risk level of 10^{-5}. This target risk level is within the range of cancer risks of 10^{-4} to 10^{-6} identified by EPA as the acceptable risk range for Superfund sites and is consistent with ADEC's recent draft guidance. Thus, use of this target risk level incorporates a measure of protection for exposure to carcinogens at the site. Consistent with EPA and ADEC guidance, risk-based concentrations for noncarcinogenic CoCs were derived with a hazard index of 1.

Despite the use of conservative screening methods, estimated tissue concentrations exceeded risk-based concentrations only for PCDDs/Fs. The maximum estimated seafood concentration of 3.7×10^{-5} mg/kg wet weight was approximately 10 times higher than the risk-based concentration of 3.0×10^{-6} mg/kg wet weight and thus would be identified as a CoC on this basis. In contrast with the estimated tissue concentration for PCDDs/Fs (toxic equivalent concentration [TEC] relative to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin [2,3,7,8-TCDD]), however, the maximum measured PCDD/F concentration (TEC) of 0.78×10^{-6} mg/kg wet weight was lower than the risk-based concentration for PCDDs/Fs (TEC). Tissue concentrations are a more reliable basis for identifying CoCs than estimated concentrations because of the uncertainty in applying BSAF estimates. BSAF-derived estimates also represent whole-body concentrations, which tend to overestimate concentrations in tissues consumed by people. Thus, given consideration of

both the estimated and measured tissue concentrations, no CoCs were identified for human health. Thus, risks to humans, if any, appear to be within levels considered acceptable by regulatory agencies.

ECOLOGICAL EVALUATION

The ecological evaluation of Ward Cove sediments consisted of an assessment of sediment toxicity throughout the Cove and a food-web assessment to estimate risks of CoPCs in sediments to representative birds and mammals at the top of the Ward Cove food web.

Sediment Toxicity Assessment

The primary objective of the sediment toxicity assessment was to identify any potential areas of concern (AOCs) in Ward Cove that may pose a risk to organisms that live within or on the sediments of the Cove. Those organisms are considered at risk of exposure to CoPCs in the Cove because historical studies have documented that CoPC concentrations in sediments are elevated in parts of the Cove.

The sediment toxicity assessment was based primarily on the evaluation of 1) concentrations of CoPCs in Ward Cove surface sediments, and 2) results of surface sediment toxicity tests conducted with sensitive and representative test species. Both of the evaluations were conducted using the information collected during Phase 1 in 1996 (28 stations in Ward Cove and the 2 reference stations in Moser Bay) and during Phase 2 in 1997 (33 stations in Ward Cove and 2 reference stations in Moser Bay).

In Phase 1, sediment toxicity was evaluated at all 28 stations sampled in Ward Cove and the 2 reference stations sampled in Moser Bay using four standardized tests: the 10-day amphipod tests using *Rhepoxynius abronius* and *Leptocheirus plumulosus* (endpoint is percent survival); the 20-day juvenile polychaete test using *Neanthes* sp. (endpoint is individual growth rate); and the 96-hour echinoderm embryo test using *Dendraster excentricus* (endpoints are percent normal survival and percent normality). The survival of *L. plumulosus* and the growth of *Neanthes* sp. were similar to reference conditions for all samples collected in Ward Cove.

In 1997 (Phase 2), additional sediment samples were collected from 33 stations throughout the Cove to refine the spatial patterns of sediment toxicity identified using the 1996 data. In 1997, sediment toxicity was evaluated using the 10-day amphipod test based on *Rhepoxynius abronius* and the 96-hour echinoderm test based on *Dendraster excentricus*.

Also in 1997, sediments were collected from eight representative stations in Ward Cove and subjected to specialized toxicity tests that primarily used *R. abronius*. The specialized toxicity tests were performed using whole sediment and porewater manipulations (pore water is the interstitial water extracted from bulk sediments). The specialized test conducted on pore water and sediment used aeration, purging, or exposure to seaweed (*Ulva* sp.) that wholly or partially remove ammonia or sulfide. The primary objective of

the specialized toxicity testing was to determine the degree to which these two conventional variables were responsible for the observed sediment toxicity in the Cove.

Chemical concentrations in surface sediments and toxicity test results were evaluated using methods consistent with the Washington State sediment management standards (Ecology 1995 and updates). For each chemical or toxicity response, the standards specify two progressively adverse levels. The lower degree of adverse effects determines compliance with sediment quality standards (SQSs) and is used to evaluate whether sediments may be toxic and therefore warrant further study. The higher degree of adverse effects determines compliance with the minimum cleanup levels (MCULs) and is used in cleanup evaluations. Site-specific sediment quality values were developed for Ward Cove (WCSQVs) for selected CoPCs (i.e., TOC, total ammonia, BOD, COD, and 4-methylphenol) using methods consistent with those used to develop the Washington State sediment management standards. Site-specific sediment quality values are in general preferable to generic numerical criteria because they factor in site-specific bioavailability, matrix effects, and the synergistic and antagonistic effects associated with the mixture of CoPCs in Ward Cove.

The major results of the sediment toxicity assessment for Ward Cove sediments in 1996 and 1997 can be summarized as follows:

- Most metals and organic compounds of potential concern (e.g., cadmium, zinc, mercury, phenol) were below applicable sediment quality values throughout all or most of Ward Cove. This was also true for PAH compounds, which were not considered to be of potential concern but were included in the study as part of NPDES monitoring.

- Most stations at which CoPCs exceeded their respective site-specific sediment quality values were located offshore from the KPC facility and downcurrent from the facility along the northern shoreline of Ward Cove. Exceedances of site-specific sediment quality values were also found offshore from the fish cannery on the southern shoreline of the Cove.

- Most exceedances of $WCSQV_{(2)}$ (analogous to MCUL) values at the 44 stations sampled in Ward Cove were found for ammonia (13 stations) and 4-methylphenol (18 stations). By contrast, exceedances of $WCSQV_{(2)}$ values for TOC, BOD, and COD were found at only 6 or fewer stations, depending on the CoPC.

- Sediment toxicity was found in only two of the four toxicity tests used to evaluate Ward Cove sediments: the amphipod test based on *Rhepoxynius abronius* (percent survival) and the echinoderm embryo test based on *Dendraster excentricus* (percent normal survival). The response range for the *R. abronius* test was very broad, ranging from 0 to 96 percent survival. By contrast, the response range for the echinoderm embryo test was narrower, with most values ranging from 30 to 80 percent normal survival.

- Sediment toxicity was not found in 1996 in either the amphipod test based on *Leptocheirus plumulosus* or the juvenile polychaete test based on *Neanthes* sp. Percent survival at all stations was very high for the *L. plumulosus* test, ranging from 89 to 100 percent. Individual growth rate at all stations for the juvenile polychaete test was also high (0.51–0.74 mg/day), relative to mean individual growth rate in the reference area (0.60 mg/day).

- Most stations at which sediment toxicity was found were located offshore from the KPC facility and downcurrent from the facility along the northern shoreline of Ward Cove.

- The results of the four sediment toxicity tests were used to develop the WCSQVs for selected CoPCs, including TOC, total ammonia, BOD, COD, and 4-methylphenol. The site-specific sediment quality values were developed using the apparent effects threshold (AET) approach. The $WCSQV_{(1)}$ (analogous to SQS) for each CoPC was based on the lowest AET for the four toxicity tests, whereas the $WCSQV_{(2)}$ was represented by the second lowest AET for the four tests.

- Percent survival for *Rhepoxynius abronius* exhibited significant ($P \leq 0.05$) and strong ($r_s > -0.75$) negative correlations with sediment concentrations of ammonia and 4-methylphenol. By contrast, normal survival of echinoderm embryos did not correlate strongly with any CoPC. Percent survival for *R. abronius* also exhibited significant ($P \leq 0.05$) and strong ($r_s > -0.75$) negative correlations with concentrations of ammonia and sulfide in the pore water of the toxicity test chambers at the end of the 10-day test period.

- Comparisons with historical sediment data collected in 1994 and 1995 in Ward Cove showed that both chemical concentrations and sediment toxicity results in the top 2 cm of sediment (1994 and 1995) were similar to the values found for the top 10 cm of sediment (1996 and 1997).

- Results of four specialized toxicity tests applied to surface sediments from eight representative stations in Ward Cove suggest that sulfide, rather than ammonia, may be a major cause of the observed sediment toxicity. Because both CoPCs covaried, it was difficult to determine their independent contributions to toxicity. However, sulfide appeared to be the major cause of toxicity because porewater concentrations in most samples substantially exceeded the LC50 for *Rhepoxynius abronius* and because simple aeration of pore water (and the resulting oxidation of sulfide) eliminated toxicity in all but one sample. By contrast, ammonia was generally present at concentrations lower than the LC50 for *R. abronius*, and toxicity did not respond as strongly to reductions in ammonia concentrations as it did to reductions in sulfide concentrations.

- The CoCs for sediment toxicity are ammonia, sulfide, and 4-methylphenol. These CoCs are natural degradation products of organic matter that originated from historical releases of raw materials and by-products of the pulping process.

Food-Web Assessment

Food-web models were used to evaluate the potential for ecological risk to two mammal species and two seabird species resulting from exposure to chemicals in Ward Cove. Mammals evaluated were the harbor seal (*Phoca vitulina*) and river otter (*Lutra canadensis*). Seabirds evaluated were the marbled murrelet (*Brachyramphus marmoratus*) and pelagic cormorant (*Phalacrocoroax pelagicus*). These species were selected because they are upper-trophic level species whose habitat-use characteristics suggest that they have the highest potential for exposure to bioaccumulative chemicals in Ward Cove. Risks were evaluated using the maximum and mean chemical concentrations found in Cove sediments during the Phase 1 or Phase 2 studies and food-web models based on conservative, but realistic assumptions. Exposure to chemicals was expressed as a total daily dose for each ecological receptor and was estimated on the basis of the characteristics of each chemical and the natural history of each receptor. Chemical concentrations in the prey of each receptor were estimated from sediment concentrations using BSAFs. The chemicals evaluated in the food-web assessment were those identified as CoPCs from a bioaccumulation standpoint (total mercury and PCDDs/Fs), as well as other chemicals found to be present in sediments at concentrations greater than reference conditions (arsenic, cadmium, zinc, and PAHs).

The risks to the target birds and mammals were quantified as hazard quotients, which are calculated for each chemical by dividing the total daily dose by the appropriate toxicity reference value (TRV). TRVs estimate the threshold dose of a chemical that may result in an adverse effect to a particular ecological receptor. A hazard quotient less than 1.0 indicates that a CoPC is unlikely to cause adverse ecological risks, whereas a hazard quotient greater than 1.0 potentially indicates the presence of adverse ecological effects. Screening models using a BSAF approach indicate no risk to harbor seals or pelagic cormorants from any CoPC at Ward Cove. For river otters, there may be a risk of adverse effects from exposure to PCDDs/Fs, because the hazard quotient exceeds 1.0 when based on the maximum sediment concentration, although not when based on the mean sediment concentration. For marbled murrelets, there may be a risk of adverse effects from exposure to cadmium, because the hazard quotient exceeds 1.0 when based on the maximum sediment concentration, although not when based on the mean sediment concentration. A recalculation of hazard quotients for PCDDs/Fs using historical bioaccumulation data on several prey species at Ward Cove indicates that the BSAF approach overestimates risk to mammalian and avian receptors between 30- and 70-fold and that the actual hazard quotient for all receptors is substantially lower than 1.0. Similarly, historical data on bioaccumulation of mercury by mussels and clams suggest that the BSAF approach overestimates exposure to metals through the food web by up to 10-fold. Exposure models, when evaluated in consideration of these uncertainties, indicate that no risks of adverse

effects result from exposure to CoPCs through the food web for mammalian or avian receptors at Ward Cove. In addition, a maternal-egg transfer model used to evaluate potential effects of TCDD TECs on early life stages of fish indicates that concentrations of PCDDs/Fs in Ward Cove sediments should not be of concern for fish or other higher trophic-level organisms.

DELINEATION OF AREA OF CONCERN

Results of the baseline human health risk assessment and the ecological evaluation are used to identify the AOC. The AOC represents that area and/or volume of sediment where active remedial action may be warranted. The AOC is then subjected to an analysis of engineering feasibility for different remedial alternatives.

As previously described, three major types of exposure pathways were considered in evaluating risks:

- Human exposure to CoPCs through seafood consumption
- Wildlife (bird and mammal) exposure to CoPCs through seafood consumption
- Benthic organism exposure to CoPCs through direct contact.

The risks associated with the first two types of exposure were determined to fall within acceptable limits when considered in the context of the conservative modeling assumptions. However, sediment toxicity was present in portions of the Cove at levels that warrant consideration of sediment remediation. Sulfide, ammonia, and possibly 4-methylphenol were likely the primary causative agents associated with sediment toxicity. Based on the adverse effects associated with Ward Cove sediments, the remedial action objectives (RAOs) are as follows:

- Reduce sediment toxicity
- Enhance recolonization of surface sediments to support a healthy benthic infaunal community with multiple taxonomic groups
- Provide a benthic macroinvertebrate community that constitutes an abundant food source to larger invertebrates and fishes.

Delineation of the AOC relied on a weight-of-evidence approach (i.e., agreement between chemical and biological indicators) that is recommended by EPA. Under the weight-of-evidence approach, the inclusion of a station as part of an AOC requires agreement among multiple chemical and/or biological indicators that unacceptable risk exists at that station. The requirement for multiple lines of evidence enhances confidence that an unacceptable risk is truly present at a station. By contrast, delineation of an AOC using results of single indicators can be biased by potential artifacts encountered with the

individual indicators. For example, a significant toxicity response at a station in the absence of any chemicals that exceed sediment quality values may be the result of the test organisms being stressed during handling and testing or being sensitive to nonchemical factors (e.g., sediment grain size distribution). From another standpoint, the exceedance of a sediment quality value by a chemical in the absence of corroboration by a significant toxicity response could mean that the chemical was not sufficiently bioavailable to result in toxicity.

Although exceedances of SQS and $WCSQV_{(1)}$ values were evaluated, the AOC was delineated based on exceedances of MCUL and $WCSQV_{(2)}$ values because the latter values provide a greater degree of confidence that ecological risks are present. In this manner, it will be ensured that the evaluation of remedial options and any future remediation costs will be focused on those parts of Ward Cove that pose the greatest ecological risk.

The echinoderm embryo toxicity test was responsible for singularly identifying the most stations as exceeding their respective sediment quality values. This characteristic of the echinoderm test casts doubt on whether its singular exceedance at farfield stations is meaningful. The lack of corroboration of the echinoderm test exceedances by any of the sediment chemical exceedances or by toxicity in the two amphipod and other tests suggests that the echinoderm test may not be responding to CoC concentrations at those stations. Several aspects of the echinoderm embryo test make it a less robust tool for determining the AOC for Ward Cove. First, there is an unquantified error component associated with the determination of the *percent normal survival endpoint.* Second, higher variability has been observed for this test as compared to other tests used to characterize sediment toxicity. This higher variability prompted Washington State and the Puget Sound Dredged Material Management Program agencies to require that statistical comparisons using this test be conducted at a significance level of $P \leq 0.10$, whereas the results of all other toxicity tests used for regulatory purposes in Washington State are evaluated at $P \leq 0.05$ (Michelsen 1996). Moreover, at the national level, U.S. EPA (1998a) did not select the echinoderm (or any other larval test) for implementing its contaminated sediment management strategy.

As previously described, the delineation of the AOC in Ward Cove relies on a weight-of-evidence approach for identifying stations at which unacceptable ecological risks are posed. Given the concerns regarding the echinoderm test expressed at the national level and the higher variability and unquantified error associated with the percent normal survival endpoint, and given site-specific results showing the uncorroborated performance of the test (relative to other environmental indicators) at numerous stations in Ward Cove, this test will not be used to singularly identify potential sediment problems in Ward Cove.

The AOC identified for more detailed evaluation was delineated on the basis of the kinds of exceedances of sediment quality values found at individual stations. As part of the delineation process, stations were grouped into two categories based on whether they were

considered an AOC station or whether they were not considered an AOC station. The criteria used to designate stations were as follows:

- **AOC Stations:** Stations considered part of the AOC were those that had one or both of the following attributes:
  - The MCUL values for both the amphipod and echinoderm toxicity tests were exceeded
  - The MCUL value for one toxicity test was exceeded and the $WCSQV_{(2)}$ value for one or more CoPCs was exceeded

  Based on these criteria, 23 stations were designated as being part of the AOC located offshore and downcurrent from the KPC facility.

- **Non-AOC Stations:**
  - No chemical or biological indicator exceeded its MCUL or $WCSQV_{(2)}$ value. Based on this criterion, 10 stations were designated as not being part of the AOC.
  - A single exceedance of the MCUL value for a toxicity test or the $WCSQV_{(2)}$ value for a CoPC was found, but no other exceedances of those values for any of the other biological or chemical indicators were found that would corroborate the results of the single exceedance. Based on this criterion, 10 farfield stations were designated as not being part of the AOC. Nine of the 10 stations were based on single exceedances for a toxicity test.

    Of the nine stations designated as not being part of the AOC based on a single exceedance for a toxicity test, the designations for seven of those stations were based only on the results for the echinoderm embryo test. The normality endpoint was therefore used to corroborate the endpoint based on percent normal survival for the seven stations identified above. Based on the results of the normality endpoint for the echinoderm embryo test described above, the significant results for the endpoint based on normal survival were not corroborated for any of the seven stations at which that endpoint was the only environmental indicator that identified a potential problem. This lack of corroboration supports the decision not to include those seven stations as part of the AOC.
  - Station 25 was not included in the AOC because it is related to cannery activities.

Based on the criteria described above, one spatially contiguous AOC of approximately 87 acres was identified for more detailed evaluation (Figure ES-2). It is located offshore from the KPC facility and extends downcurrent along the northern shoreline of Ward Cove.

NATURAL RECOVERY

Natural recovery is an integral part of EPA's contaminated sediment management strategy (U.S. EPA 1998a). Ward Cove is an ideal site for considering natural recovery of surface sediments for all or part of the AOC for several reasons: the source of pulp mill effluent was eliminated with shutdown of the mill in 1997; the CoCs in sediments are natural products of organic matter degradation and are not persistent; chemical concentrations in surface sediments are within acceptable limits for human health and wildlife and of limited toxicity to the benthos; and existing sediment and hydrodynamic modeling indicate that offsite sediment transport is not a concern.

Numerical modeling of quantifiable natural recovery processes for surface sediments in Ward Cove indicates that the recovery period is likely to be longest directly offshore of the KPC mill and along the north shore to the west of the mill. Recovery of ammonia and 4-methylphenol to levels below the sediment quality values used in the numerical modeling is expected to take more than 20 years in this region. Recovery of sulfide, in contrast, is expected to require less than 8 years. The absolute durations of the predicted recovery periods are somewhat uncertain, as a result of their dependence on the organic carbon decay rate, for which there are no site-specific data. Differences in degradation rates of effluent solids and woody debris, in particular, could result in either an increase or decrease in recovery period. In addition, the steep slopes along the north shore of Ward Cove, the spatial variability of sediment deposition processes, and the positive feedback between chemical and biological recovery processes may all reduce recovery periods from those predicted. Despite these limitations in the model results, the predictions of areas requiring extended natural recovery periods are consistent with each other, consistent with 1996 and 1997 field data, and plausible with regard to current knowledge of conditions in Ward Cove. Because organically enriched sediment is confined to the inner part of Ward Cove even after decades of mill discharges and because field measurements indicate that there is little potential for sediment transport, the areal extent of affected sediment is not expected to increase as a result of sediment transport during the recovery period.

Results of the specialized toxicity tests, observations made on the benthic communities in Ward Cove, and case studies of other sites with organic-rich sediment provide compelling arguments for natural recovery in a reasonable time frame. The potential for benthic macroinvertebrate communities to recover naturally in Ward Cove is supported by the results of the specialized toxicity tests. Specifically, the results of the porewater specialized toxicity test using *Rhepoxynius abronius* indicate that sulfide appears to be the major cause of sediment toxicity in sediment samples from most areas of the Cove.

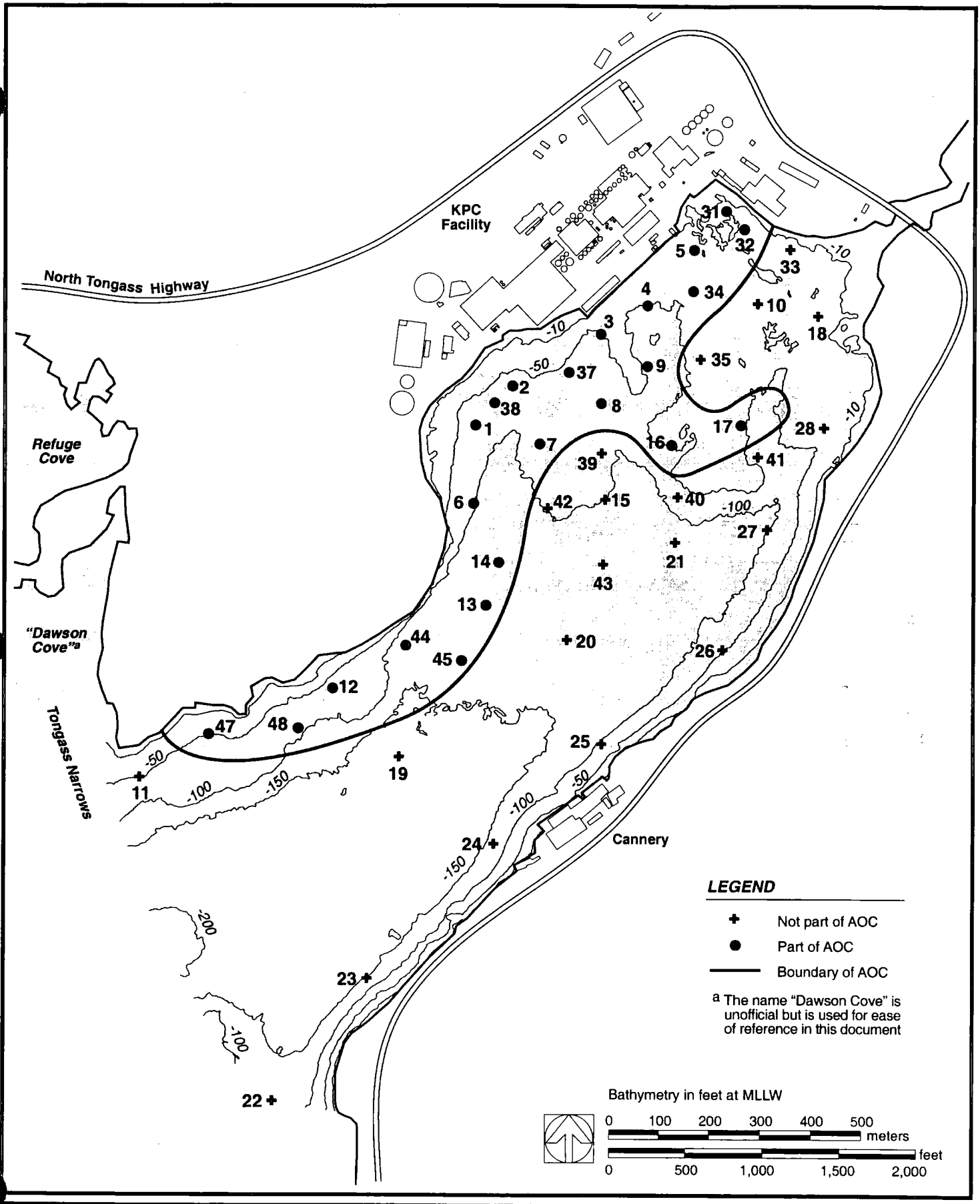

Figure ES-2. Delineation of area of concern for further evaluation.

The observed characteristics of benthic macroinvertebrate communities in Ward Cove are consistent with those documented for organically enriched areas and dredged material disposal areas in other studies, in which initial colonization by opportunistic species is followed by colonization by equilibrium species. The likely pattern of future recolonization in Ward Cove is illustrated by the results of several case studies of recolonization following closure of a pulp mill in Sweden, improvements in the effluent quality of a pulp mill in British Columbia, and sewage treatment abatement in Los Angeles, California. Based on the theoretical models of benthic recovery and the results of case studies, recovery of benthic macroinvertebrate communities in Ward Cove is predicted to occur within approximately 10 years.

EVALUATION OF REMEDIAL TECHNOLOGIES AND ALTERNATIVES

The unique physical and chemical characteristics of Ward Cove sediments are critical considerations in the selection and evaluation of potentially applicable technologies and process options. Sediments affected by releases from the KPC facility are distinctly different from the underlying native sediments and from sediment in many marine environments. Affected sediments contain wood debris, have high water and organic content, and are black in appearance. The CoCs (ammonia, sulfide, and 4-methylphenol) are all natural degradation products of organic matter and wood debris. Concentrations of persistent chemicals that are toxic or that have the potential to bioaccumulate (e.g., mercury or PCDDs/Fs) are low and do not pose a risk to human health or wildlife. A limited risk to benthic infauna is observed (i.e., a limited degree of sediment toxicity is observed); however, a benthic community is present, with characteristics consistent with those documented for organic-rich areas. The cessation of pulping activities in May 1997 (i.e., complete control of effluent from the pulping process), the nature of the CoCs, and the effectiveness of natural recovery processes all indicate that aggressive remedial efforts are not warranted.

Ward Cove sediments that do not currently meet the RAOs can be dredged, treated or capped in place, or left to recover naturally. If sediments are dredged, they can be disposed of in various ways. Potential disposal options include upland disposal (in an appropriate landfill), near-shore disposal (in a constructed facility along the shoreline), and confined aquatic disposal (CAD) (which generally includes capping in place with clean material). From an engineering and remediation perspective, problem sediments in Ward Cove have limited strength or have essentially no strength, depending on the water content. Placing cover material over the extremely soft, organic, fine-grained sediments presents unique challenges. The difficulty in dredging, transporting, and disposing of this very soft material also limits the range of feasible remedial options. Additional physical constraints on sediment remediation in Ward Cove include the following:

- There are no "hot spots" of contamination (i.e., there is **not** a small portion of the AOC that contains most of the mass of CoCs). The size of the AOC (87 acres) and the total volume of organic-rich sediment (approximately 840,000 cubic yards [cy], assuming an average thickness of 6 ft) poses unique challenges for balancing benefits and costs.

- There are significant areas of the bottom covered by sunken logs. In some areas, the sunken logs are several layers thick.
- There are few potential disposal sites in Ward Cove for dredged sediment because of the bathymetry and limited size of the Cove.
- It is believed that capping or dredging steep slopes (steeper than 4H:1V) in the AOC would not be successful. Capping or dredging in water depths greater than 120 ft would be difficult to achieve. Capping can be performed in deeper water (e.g., >120 ft) when the criteria for capping effectiveness are relaxed (e.g., partial coverage or mounding is acceptable). For the purpose of this report, −120 ft mean lower low water is used for the maximum depth for capping.

The DTSR describes potential remedial technologies for problem sediments. These potential technologies were screened to identify those that are most appropriate for Ward Cove. The technologies that remained after this screening process, and were carried forward to a more detailed analysis of alternatives, are as follows:

- In-place containment using thin capping techniques is feasible
- Sediment removal can be accomplished with a mechanical dredge
- Containment facilities that are suitable for dredged sediments include upland landfills, shallow CAD, and near-shore confined disposal facilities (NCDFs).

These technologies, along with natural recovery, were combined into six different cleanup alternatives, ranging from no action to an alternative that includes dredging and confining a large volume of problem sediments. The alternatives were compared against one another to allow selection of an appropriate remedy for the Cove. Given the large area and volume of the AOC (87 acres and 840,000 cy, respectively) and the limited human and environmental risks, thin capping and natural recovery are critical elements of most alternatives. Alternatives include dredging to either address future operational needs (i.e., navigational dredging) or illustrate the costs associated with the most feasible disposal options (shallow CAD and NCDF). For those alternatives that include CAD or NCDF, the volume of material dredged is based entirely on the capacity of the disposal site.

Several candidate remedial alternatives were developed, ranging from no action to an alternative that includes dredging and confining a large volume of problem sediments. These alternatives were compared against one another to allow selection of an appropriate remedy for the Cove (Table ES-1).

Both Alternatives A2 and B will achieve RAOs, but over different time periods and at different costs. Natural recovery (Alternative A2) is less expensive, but slower. Thin capping (Alternative B; also known as enhanced recovery) is more expensive, but

TABLE ES-1. SUMMARY OF DETAILED ANALYSIS OF ALTERNATIVES

	Alternative A1	Alternative A2	Alternative B	Alternative C	Alternative D	Alternative E
	No Action	Natural Recovery	Thin Cap, Dredge 12,300 cy, Dispose Upland[a]	Thin Cap, Dredge 80,000 cy, Dispose in Site 2 CAD[a]	Thin Cap, Dredge 175,000 cy, Dispose in Site 2 NCDF[a]	Thin Cap, Dredge 155,000 cy, Dispose in Site 1 NCDF[a]
Threshold Criteria						
Overall Protection of Human Health[b] and Environment	Sediments pose very limited hazard to environment. Natural recovery would likely occur to meet RAOs. Ability of tube-dwelling organisms to successfully colonize Ward Cove has been demonstrated by sediment toxicity tests. However, no monitoring would be conducted to verify recovery.	Same as Alternative A1 except monitoring would be conducted to verify recovery and ability to meet RAOs.	Sediments pose very limited hazard to environment. Natural recovery would likely occur to meet RAOs in uncapped areas. Ability of tube-dwelling organisms to successfully colonize Ward Cove has been demonstrated by sediment toxicity tests. Thin cap over portion of AOC would accelerate natural recovery. Dredging of small area would have minimal adverse impacts on environment, workers, and public.	Same as Alternative B except dredging larger volume, CAD construction, and log removal in areas to be dredged would have greater potential short-term adverse impacts on environment, workers, and public.	Same as Alternative C except dredging larger volume would have greater potential short-term adverse impacts on environment, workers, and public.	Same as Alternative D.
Compliance with ARARs	Will comply with ARARs.	Will comply with ARARs.	Will comply with ARARs.	Will comply with ARARs.	Will comply with ARARs.	Will comply with ARARs.
Primary Balancing Criteria						
Long-Term Effectiveness and Permanence	Would likely provide long-term protectiveness, but no monitoring would be conducted to verify it.	Would likely provide long-term protectiveness; monitoring would be conducted to verify it.	Same as Alternative A2.	Same as Alternative A2.	Same as Alternative A2.	Same as Alternative A2.
Reduction of Toxicity, Mobility, or Volume through Treatment	No treatment would occur.	Same as Alternative A1.	Same as Alternative A1.	Same as Alternative A1.	Same as Alternative A1.	Same as Alternative A1.
Short-Term Effectiveness	No additional risks to environment, workers, or public.	Minor safety hazards to workers during sampling.	Minimal risks to public. Construction related risks for remediation workers associated with working on water and with heavy equipment. Existing benthic communities would be largely eliminated by capping, but would recolonize. Water quality effects would need to be monitored during remediation.	Same as Alternative B except short-term risks would be greater because of larger volume of sediment dredged.	Same as Alternative C except short-term risks would be greater because of larger volume of sediment dredged.	Same as Alternative D. Short-term risks may be less if smaller volume of sediment is dredged.

TABLE ES-1. (cont.)

	Alternative A1	Alternative A2	Alternative B	Alternative C	Alternative D	Alternative E
	No Action	Natural Recovery	Thin Cap, Dredge 12,300 cy, Dispose Upland[a]	Thin Cap, Dredge 80,000 cy, Dispose in Site 2 CAD[a]	Thin Cap, Dredge 175,000 cy, Dispose in Site 2 NCDF[a]	Thin Cap, Dredge 155,000 cy, Dispose in Site 1 NCDF[a]
Implementability	No technologies are to be implemented.	No technologies are to be implemented.	Technically feasible to implement, but not for slopes steeper than 4H:1V and very high-density log area. A pilot study would be conducted to determine capping approach (thin layer vs. mounding), placement methods, and other implementability issues.	Same as Alternative B except removing larger quantity of sediment and constructing CAD would be more difficult to implement. Capping the CAD would be difficult. A special construction approach (building up the dikes to allow settling/dewatering, then partially removing the dikes) would be needed to facilitate capping CAD. Implementation would need to be coordinated with potential future development (e.g., a marina).	Appears to be technically feasible. Implementability same as Alternative B. Implementation would need to be coordinated with potential future development (e.g., a marina). After construction, NCDF could be used for storage or parking, but use for buildings would require pilings.	Appears to be technically feasible. Implementability same as Alternative B. Implementation would need to be coordinated with future use of KPC facility. After construction, NCDF could be used for storage or parking, but use for buildings would require pilings. Use of NCDF for log storage would require additional evaluation during design and could affect capacity of NCDF.
Cost (total present worth)	Minimal or none	$0.5 million	$4.5 million (KPC landfill) $5.6 million (Washington landfill)	$17 million	$33 million	$30 million

Note: AOC - area of concern
ARAR - applicable or relevant and appropriate requirement
CAD - confined aquatic disposal
cy - cubic yard
KPC - Ketchikan Pulp Company
NCDF - near-shore confined disposal facility
RAO - remedial action objective

[a] Alternative includes removal of logs in areas to be dredged.

[b] Sediments are within acceptable limits for human health.

quicker. Alternative B is expected to achieve a more advanced stage of benthic recolonization over a shorter period; however, the existing benthic community will be affected by placement of the thin cap.

The alternatives that involve extensive dredging (Alternatives C, D, and E) would also likely meet RAOs, but would be difficult to implement because of the high water content and very soft, fine-grained nature of the sediments. In addition, the incremental costs for Alternatives C, D, and E (compared to Alternative B) are disproportionate to their incremental benefits. There would be little or no gain in overall environmental benefits to the Cove for the additional actions and costs incurred. These removal alternatives address only a portion of the total volume of organic-rich sediments in Ward Cove and were included in the evaluation primarily to illustrate capacity limitations of disposal sites and the very high unit costs involved in dredging and confining Ward Cove sediments.

RECOMMENDED ALTERNATIVE

The recommended alternative for the Ward Cove AOC is Alternative B. Alternative B includes thin capping with navigation dredging and upland disposal of dredged material. Thin capping will reduce surface sediment toxicity, enhance recolonization of surface sediments, and provide an abundant and functioning benthic community that provides food to larger invertebrates and fish. This alternative is particularly suitable for the type of problem sediment present in Ward Cove, which has limited toxicity and does not contain persistent chemicals that are highly toxic or that have the potential to bioaccumulate to levels of concern. The thin cap would be placed on approximately 34–40 acres of the AOC (Figure ES-3), depending on the post-dredging area requiring capping if native sediments are not reached during dredging. The appropriate placement technique for a thin cap will be determined with a bench-scale or field pilot study. If the pilot study indicates that Ward Cove sediments cannot be capped effectively due to high water content and low engineering strength, an alternative placement technique, most likely island mounding, will be applied to that portion of the site where it is feasible (i.e., where accumulated thicknesses of organic matter can be displaced).

Natural recovery is a critical component of Alternative B and would be in effect for those portions of the AOC where thin-layer capping, amending the surface, or mounding is not feasible. Natural recovery is an integral component of EPA's sediment management strategy (U.S. EPA 1998a) and would also be an effective remedy for the entire AOC; however, uncertainties in the rate of natural recovery are a concern and may warrant a more active form of remediation (thin capping) where feasible. For those areas where thin capping is found to be unsuccessful in Ward Cove, it is anticipated that the remedy will be natural recovery. Limited dredging of the sediments in the vicinity of KPC's main dock would also be conducted under this alternative because a cap could not be placed in this portion of the AOC without affecting navigation. The dredged sediments would be disposed at an upland landfill that is authorized to accept the material. Thin capping would be conducted after navigational dredging unless native sediments are reached during dredging.

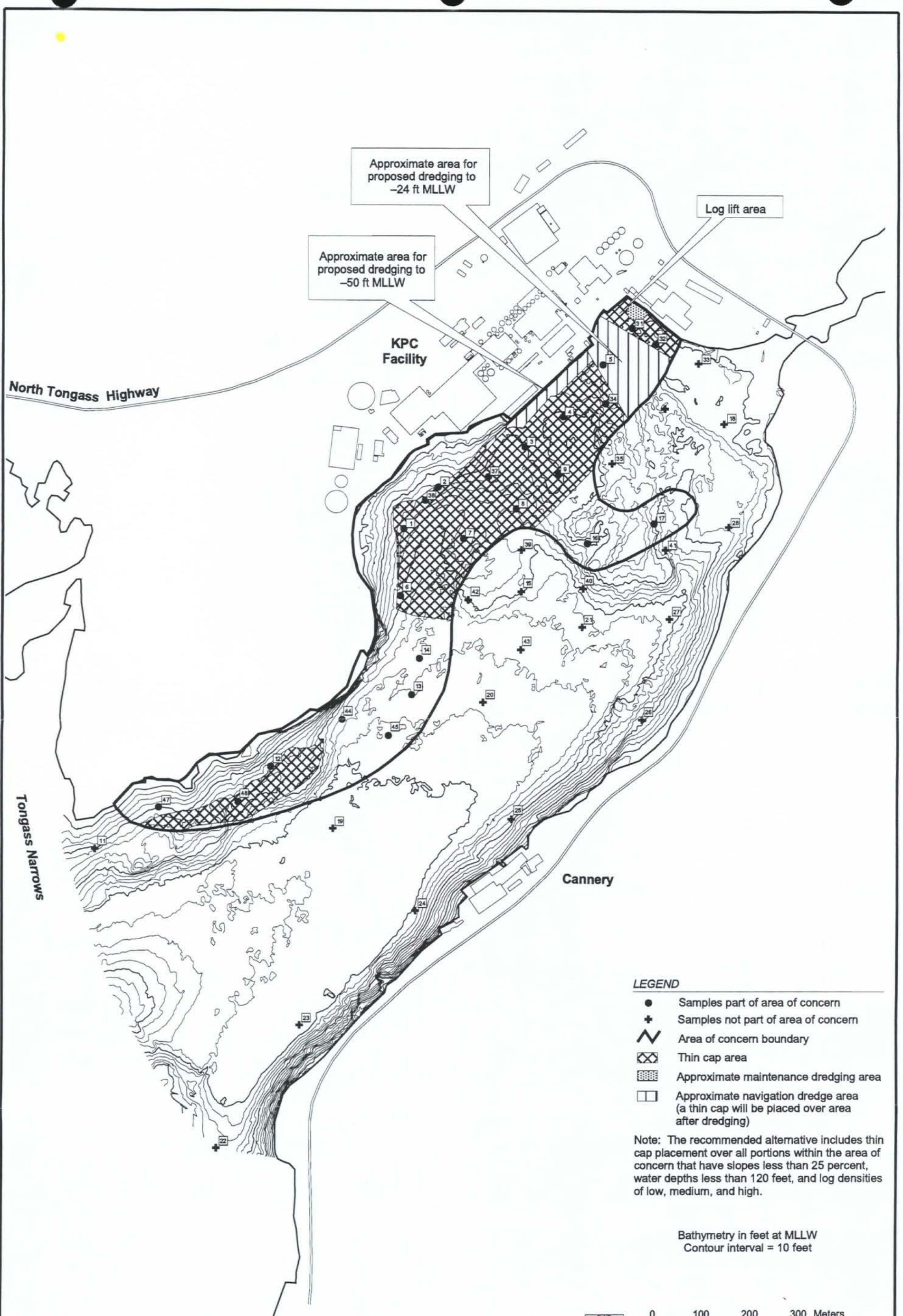

Figure ES-3. Recommended alternative: Alternative B, thin capping with navigational dredging and upland disposal.

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The recommended alternative will be integrated with ongoing and future development plans for Ward Cove. Ongoing business operations in Ward Cove consist of activities related to operation of the Ketchikan sawmill. However, other potential development options exist. The City of Ketchikan has expressed interest in developing a portion of the south shore of Ward Cove into a marina. Other possibilities include a small hydroelectric facility operated by Ketchikan Public Utilities, a fish by-products processing facility, and other light industrial uses that would take advantage of the industrial amenities offered by the site. With proper planning, all of these development possibilities could be integrated with the remedial alternatives that have been developed for Ward Cove.

PUBLIC COMMENT ON DTSR

The draft DTSR was made available for public review and comment from August 3 through October 1, 1998. An availability session, a public meeting, and a meeting with the Ketchikan Technical Discussion Group were held on September 17, 1998. EPA also provided notice of the public comment period and the availability session and public meeting in *The Ketchikan Daily News* and *The Local Paper*. EPA subsequently provided a summary of public comments and responses to those comments (U.S. EPA 1999a). All comments received during the public comment period were considered in revising the DTSR.

1. INTRODUCTION

The Ketchikan Pulp Company (KPC) facility is located on the shoreline of Ward Cove, near Ketchikan, Alaska (Figure 1-1). The facility began operations in 1954 and discharged pulp mill effluent to the Cove until March 1997, when pulping operations terminated. In September 1995, KPC entered into a Consent Decree with the U.S. Environmental Protection Agency (EPA) to address environmental issues related to KPC's Ketchikan facility. As part of the Consent Decree, KPC agreed to develop and implement a Ward Cove sediment remediation project to address environmental issues associated with sediments in the Cove. The major phases of the project include the following:

- Preparation of a technical studies work plan (submitted to EPA on April 18, 1996 [PTI 1996])
- Implementation of a Phase 1 sampling effort (conducted in May and June of 1996)
- Preparation of a report summarizing the Phase 1 results and presenting a general overview of the Phase 2 study design (submitted to EPA on March 31, 1997 [PTI 1997g])
- Preparation of a Phase 2 sampling and analysis plan and quality assurance project plan (submitted to EPA on June 26, 1997 [PTI 1997f])
- Implementation of a Phase 2 sampling effort (conducted in July and August of 1997)
- Preparation of a detailed technical studies report (DTSR; this report)
- Preparation of a remediation work plan for conducting the selected remedial actions
- Implementation of the remedial actions.

Sediment samples for the KPC 1996 National Pollutant Discharge Elimination System (NPDES) sediment monitoring program were also collected during the Phase 1 and Phase 2 sampling efforts.

This report incorporates the EPA comments that Exponent received following public review of the draft DTSR. Some of the issues and concerns raised during public review are directly addressed in EPA's response to public comments (U.S. EPA 1999a) and are not addressed in this report.

Figure 1-1. Location of Ward Cove.

The activities that led to the preparation of the DTSR and the key topics addressed in the DTSR are summarized in Figure 1-2.

1.1 REPORT OBJECTIVES

The primary objective of the Ward Cove sediment remediation project is to determine the extent to which sediments in Ward Cove may pose risks of adverse effects to humans and ecological receptors and therefore potentially warrant remediation. The primary objectives of this report are to:

- Provide the detailed results of the technical studies required by the Consent Decree
- Satisfy the requirements of a remedial investigation and feasibility study (RI/FS) under the Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA)
- Satisfy the NPDES requirements for KPC's 1997 sediment monitoring program.

This report builds on the Phase 1 report (PTI 1997g) and consolidates all evaluations relevant to site characterization and remedy development. The Consent Decree did not specify a Phase 1 report; Phase 1 data were evaluated and reported to communicate the implications of the data to regulators and to build consensus on the appropriate evaluation techniques.

1.2 OVERVIEW OF SITE ACTIVITIES AND THEIR RELATIONSHIP TO WARD COVE

Understanding the relationships among chemical sources, transport pathways, exposure pathways, and human and ecological receptors is critical to the development of human health and ecological hazard assessments and, ultimately, to the development of an appropriate cleanup remedy. Ward Cove, the focus of these technical studies, has received discharges from the KPC facility and associated water-based log handling activities for approximately 43 years. Chemicals and woody materials present in sediments in the vicinity of the KPC facility reflect the impacts of these releases. The following sections discuss chemicals of potential concern (CoPCs) in Ward Cove, sources of CoPCs, and dredging activities.

1.2.1 Ongoing Activities and Potential Future Development in Ward Cove

Ongoing business operations in Ward Cove consist of activities related to operation of the Ketchikan sawmill. These activities include towing and storing log rafts, dewatering log

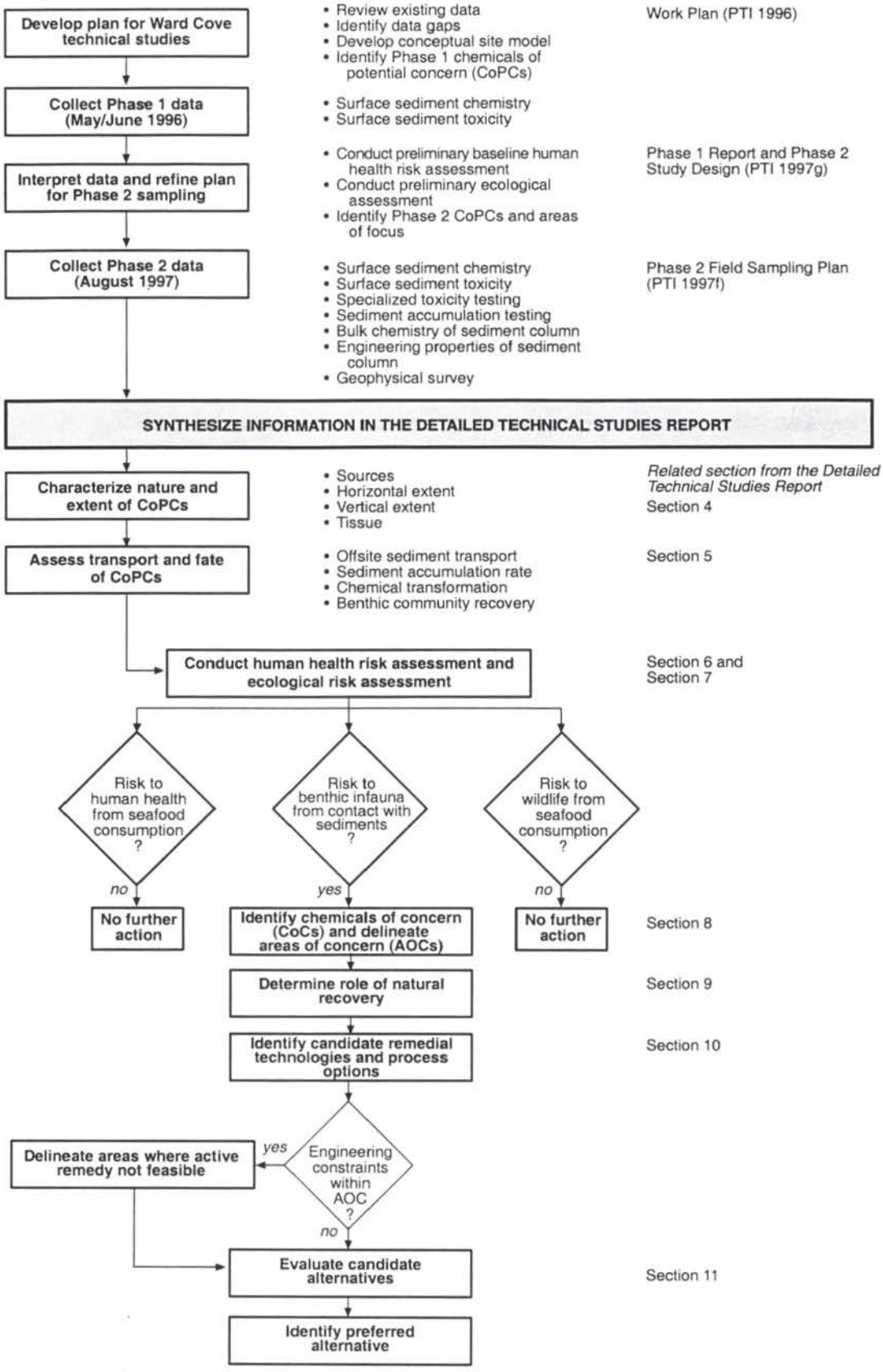

Figure 1-2. Overview of Ward Cove detailed technical studies.

bundles at the log transfer facility (LTF), sawing and chipping logs, hogging bark, transferring sawn wood products, chips, and hog fuel to barges, and loading logs onto barges.

In addition to continuing existing operations, KPC expects to construct and operate a green veneer mill at the Ward Cove facility by early 1999. Activities associated with operation of this veneer mill are essentially identical to those described above associated with the sawmill.

KPC intends to seek renewal of the individual NPDES permit for the LTF located at the sawmill. The permit will authorize the discharge of bark and other organic debris to Ward Cove in conjunction with operation of the LTF. KPC expects that future permits and the State of Alaska Certificate for Reasonable Assurance will impose more stringent and comprehensive Best Management Practices (BMPs) designed to minimize discharge, and subsequent deposition, of bark and other debris in Ward Cove. Development and implementation of these BMPs would help ensure consistency with potential remedial alternatives proposed for Ward Cove, including natural recovery.

Other than the veneer mill, no future development is definitively planned for Ward Cove. However, other potential development options exist. The City of Ketchikan has expressed interest in developing a portion of the south shore of Ward Cove into a marina. Other possibilities include a small hydroelectric facility operated by Ketchikan Public Utilities, a fish by-products processing facility, and other light industrial users that would take advantage of the industrial amenities offered by the site. With proper planning, all of these development possibilities could be integrated with the remedial alternatives that have been developed for Ward Cove.

The listing of Ward Cove as a 303(d) water body is also relevant to future uses and development. Section 303(d) of the Clean Water Act requires states to identify water bodies that do not meet state clean water goals, called water quality standards. Ward Cove is on Alaska's 303(d) list of "impaired" water bodies because it does not meet Alaska's water quality standards for sediment toxicity, dissolved gas (oxygen is depleted in portions of the water column in the summer), and residue (sunken logs and bark debris are present). A water body remains on the 303(d) list until it meets the standards or until a total maximum daily load is established for the pollutants exceeding water quality standards. EPA and the Alaska Department of Environmental Conservation (ADEC) have stated that they believe that in time the sediment cleanup in Ward Cove will attain the Alaska water quality standard for sediment toxicity (see Fact Sheet on Ward Cove Water Quality and 303(d) Issues, ADEC and U.S. EPA 1998).

As explained in the Fact Sheet developed by ADEC and EPA, the listing of a water body on the 303(d) list does not by itself prohibit the permitting of facilities that are expected to discharge into that water body, and options for future permitting in Ward Cove do exist. For example, if a new discharge from a facility does not affect a listed pollutant parameter, the facility could be issued a permit in the same way that any other discharge is issued a permit. If a discharge affects a listed pollutant parameter, then the discharge must generally meet the state water quality standard for that parameter at the point of

discharge because EPA's regulations require that discharges must not cause or contribute to any exceedance for which the water body is listed. The Fact Sheet explains that the first step ADEC takes to address a 303(d) listed water body is to assess the water body through the development of a water body recovery plan. ADEC plans to use the watershed approach for developing a water body recovery plan for Ward Cove. This approach will involve broad public participation from citizens and stakeholders, the Borough of Ketchikan, and other state and federal agencies.

1.2.2 Chemicals of Potential Concern

A preliminary list of CoPCs was identified during preparation of the technical studies work plan for Ward Cove (PTI 1996). These Phase 1 CoPCs were selected on the basis of historical studies that documented chemical concentrations in sediments and in fish and shellfish tissue. Phase 1 CoPCs were reevaluated, and the list of chemicals was refined prior to conducting the initial human health and ecological assessments presented in PTI (1997g) to more clearly distinguish three categories of CoPCs: 1) CoPCs for ecological risks associated with sediment toxicity, 2) CoPCs for ecological risks associated with food-web bioaccumulation, and 3) CoPCs for human health risks associated with seafood consumption. These categories of CoPCs are summarized in Table 1-1 and are the basis for the human health and ecological evaluations presented in Sections 6 and 7, respectively.

The Phase 1 CoPCs were subjected to a rigorous evaluation of their potential risk to human and ecological receptors as part of the Phase 1 report (PTI 1997g). The revised list of CoPCs was a key factor in the selection of target analytes for Phase 2 of the sediment investigation (PTI 1997f).

1.2.3 Sources of Chemicals to Ward Cove

Potential mechanisms of contaminant transport from the mill area to Ward Cove, both historical and current, include partially treated pulp mill effluent, wastewater treatment discharge, storm water discharge, surface water runoff, groundwater transport, aerial deposition, and spills/accidental releases. Historical discharges from the pulping process and wood handling (in-water rafting) are considered the most significant potential sources of chemicals and organic material to Ward Cove. Sources of chemicals and organic material to Ward Cove are discussed in greater detail in Section 4.1.

Historically, KPC discharged an average of 35–45 million gallons per day (mgd) of wastewater to Ward Cove through several outfalls: 001, 002, 003, and 004 (Figure 1-3; Jones & Stokes and Kinnetic 1989; Hayes 1998, pers. comm.). From 1954 until 1972, wastewater was discharged at the shoreline to Ward Cove through four separate outfalls which were located progressively from west to east. Untreated wastewater primarily from the acid plant, wash plant, bleach plant, and machine room was discharged through the main outfall (001), which was located west of the No. 1 warehouse. Partially treated wastewater from the boiler house was discharged through Outfall 002. Wastewater

TABLE 1-1. CHEMICALS OF POTENTIAL CONCERN

Ecological CoPCs		
Sediment Toxicity	Ecological Food Web	Human Health CoPCs
TOC	Arsenic	Arsenic
Total ammonia	Cadmium	Cadmium
Total sulfide	Mercury	Methylmercury
BOD	Zinc	Zinc
COD	PAHs	Phenol
Cadmium	Dioxins and furans	4-Methylphenol
Mercury		PAHs
Zinc		Dioxin (TCDD)
Phenol		
4-Methylphenol		
Dioxins and furans		

Note: BOD - biochemical oxygen demand
COD - chemical oxygen demand
CoPC - chemical of potential concern
PAH - polycyclic aromatic hydrocarbon
TCDD - tetrachlorodibenzo-*p*-dioxin
TOC - total organic carbon

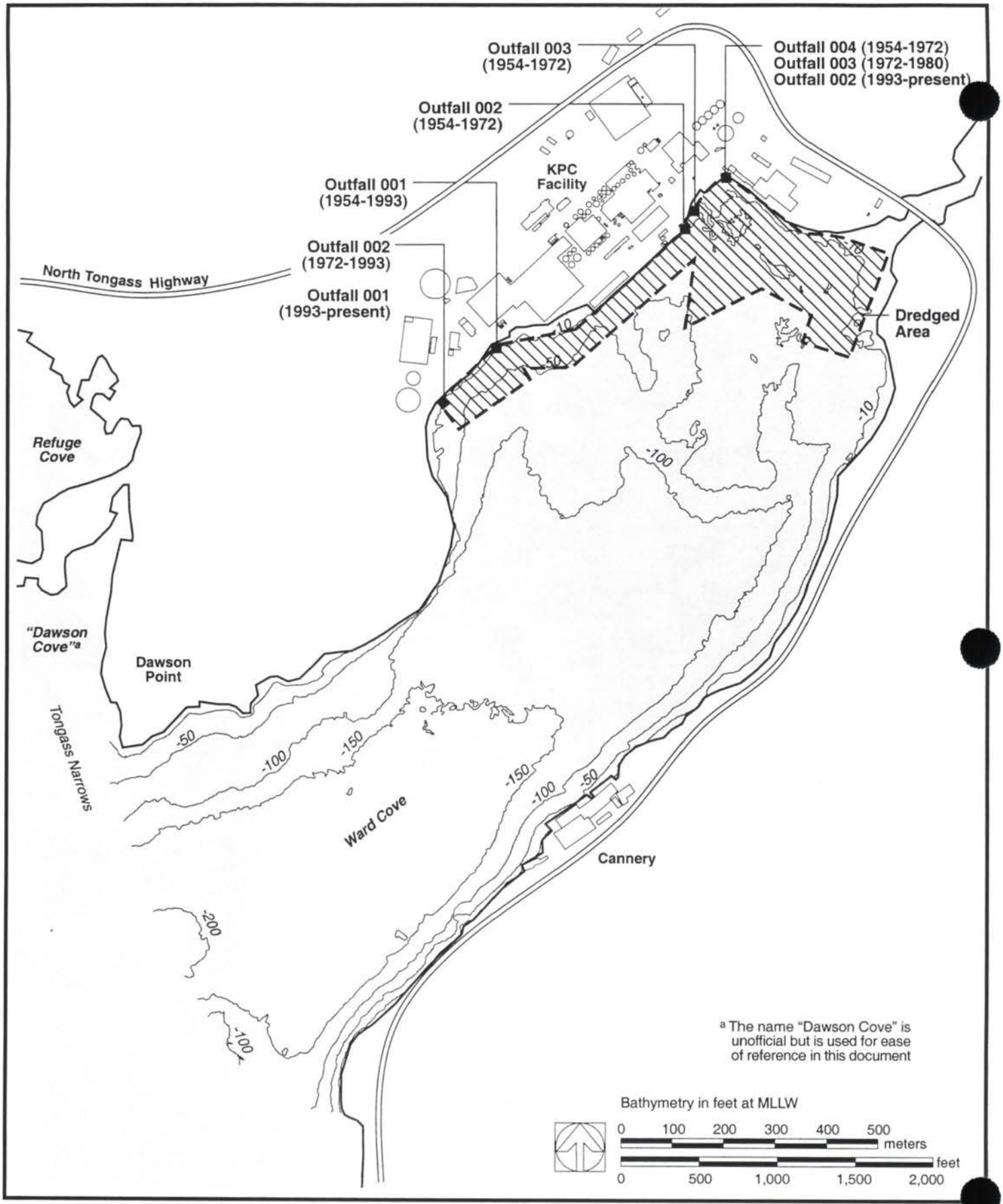

Figure 1-3. Locations of original and current KPC outfalls and location of dredged area.

generated in the wood rooms passed through North rotary screens and was discharged, via the hog house, through Outfall 003. Sediments and filter backwash from the water treatment plant were discharged through Outfall 004.

The primary treatment facility was constructed in 1971 to reduce discharges of suspended solids and included a vacuum filter, a "V" press, and a grit chamber (which collected solids screened during treatment of wastewater from the wood rooms). As a result, Outfall 003 was eliminated in 1972 and routed to primary treatment. Wastewater from the primary treatment facility was discharged from a separate outfall. In 1972, Outfall 002 was eliminated by rerouting to the main outfall. At this time, outfall numbers were redesignated. The main outfall (001) remained the same. The primary discharge was designated 002, and the water treatment plant became 003. The secondary activated sludge treatment system was installed in 1980 to reduce biochemical oxygen demand (BOD) discharges and included an aeration basin and a secondary clarifier. Primary and secondary effluents were combined and discharged through a newly constructed outfall separate from the main outfall. In 1993, the effluent neutralization system was installed to combine all process discharges and control pH of the combined discharge. This discharge was designated as Outfall 001 and the water treatment plant outfall became 002. Discharge of pulping waste ended with the shutdown of the pulp mill; however, the powerhouse remained active until March 1998, and the sawmill is still active. Outfall 001, which discharges about 50 ft offshore, currently discharges approximately 2–3 mgd of water to preserve a pipeline constructed of wood staves. This pipeline formerly provided process water to the pulp mill. Outfall 002 (originally called Outfall 003) discharges a small volume of natural influent water from the water supply plant.

Specific sources of CoPCs in Ward Cove sediments can in part be determined from knowledge of site activities, which were evaluated in greater detail during the preparation of the site background document (PTI 1997c). Information on chemicals used at the facility, general process knowledge, and reported use or spills of potential source materials were reviewed to identify the ultimate sources of Ward Cove CoPCs. The following information about sources was inferred:

- **Organic Matter and Organic Matter Degradation Products—** Organic matter (i.e., woody material and wood by-products) was a major constituent of effluent from the mill (ENSR 1996b). This woody material, which was both raw material and product of the pulp mill, is a primary source material, or fuel, for processes that occur naturally in the sediments. Microbial degradation of the organic matter (i.e., the woody material and wood by-products) leads to oxygen depletion and production of ammonia, sulfide, and 4-methylphenol (Sjöström 1981) in the sediment. Log rafting in the Cove is also a major source of organic-rich wood debris to the sediments.

  4-Methylphenol may have been present in historical KPC effluent discharges as a by-product of lignin degradation; however, it was not a target analyte for NPDES monitoring and was not analyzed in the

effluent when the KPC facility was active. 4-Methylphenol is readily degraded in the environment, making it unlikely that the 4-methylphenol detected in sediment adjacent to the KPC facility represents historical releases of 4-methylphenol. The source of the 4-methylphenol in sediments adjacent to the KPC facility is likely the partially degraded lignin deposited from historical discharges.

- **Metals**—None of the metals identified in sediment is associated with known source materials or process chemicals or with releases associated with site activities; however, arsenic has been detected at elevated concentrations in mill area surface and subsurface soils and in wastewater treatment plant sludge and grit samples. Although arsenical insecticides have been used at other log storage areas, they were not used on logs in Ward Cove (Maloy 1997, pers. comm.). Crushed gravel has been sampled and identified as a potential source of arsenic in upland soils. Flyash generated from burning wood and sludge in the power boilers and collected in the electrostatic precipitators on the boilers (i.e., electrostatic precipitator [ESP] flyash) is another potential source of arsenic in upland soils. No other metals have been identified in soil or process waste samples collected from the mill area.

  Copper and zinc in sediments may have been released from the extensive piping at the facility by contact of the piping with sulfurous acid used in the pulping process combined with saltwater-induced corrosion, although copper is found at slightly elevated concentrations in the mill's influent water from Connell Lake. In the past, elevated concentrations of mercury were detected in process water. The problem was traced to the caustic (sodium hydroxide) solution used in the pulp bleaching process. The caustic had been obtained from a facility that used a mercury cell production process (ENSR 1995c). The problem was corrected, and mercury was not detected in subsequent monitoring. Wastewater treatment plant sludge samples and samples of soil, foam, and grit from areas where wastewater treatment plant liquids had overflowed were collected during the uplands remedial investigation and did not have elevated concentrations of mercury.

- **Petroleum Hydrocarbon Compounds and Polycyclic Aromatic Hydrocarbon (PAH) Compounds**—Petroleum hydrocarbon compounds and PAHs have been detected in several soil samples collected at the mill. Fuel oil, hydraulic fluids, and diesel were all stored and used extensively at the site, and spills and chronic leakages of these materials have been reported. PAHs are common constituents of petroleum products.

- **Polychlorinated Dibenzo-*p*-dioxins (PCDDs) and Polychlorinated Dibenzofurans (PCDFs) (PCDDs and PCDFs are referred to collectively as PCDDs/Fs in this report)**—PCDDs/Fs were detected at elevated concentrations in wastewater treatment plant sludge/grit and

flyash samples collected from the site while the mill was still in operation and have been detected at elevated concentrations in soil samples from several areas of the site. PCDDs/Fs were not detected at elevated concentrations in bottom ash samples collected from the bottom of the power boilers (i.e., bottom ash). PCDDs/Fs may have been formed during combustion of wastewater treatment plant sludge and salt water-laden wood in the power boilers and were detected in ESP flyash. Prior to 1972, wastewater was not treated before discharge into Ward Cove, and a mixture of sludge and ESP flyash (i.e., slurried ash) was often discharged along with the wastewater. After the installation of the wastewater treatment plant in 1972, the sludge that was generated at the plant (which, until 1992, included some slurried flyash) was generally mixed with wood waste and burned in the power boiler, although some sludge was placed near the west parking lot during the mid-1970s and small amounts of sludge have been sent to the landfill. Sludge samples collected from the west parking lot did not have elevated concentrations of PCDDs/Fs. PCDDs/Fs formed during combustion of sludge and not captured by ash collectors in the boiler stacks could also have been deposited into Ward Cove. However, the amount of material would most likely be insignificant compared to that from wastewater discharges, given that since the beginning of the burning of sludge in 1972, several increasingly more efficient mechanical and/or electrostatic ash collection systems were installed in the boiler stacks.

1.2.4 Dredging Activities

KPC performed various marine activities in Ward Cove, including the transport of log rafts using tugboats; delivery of wood chips, chemicals, and supplies using tugs and barges; and shipment of pulp using large oceangoing ships. To perform these activities, it was necessary to maintain sufficient water depths within Ward Cove to allow ship and barge traffic. Even though KPC had systems and equipment to minimize the loss of settleable solids to Ward Cove, navigable depths were compromised over time by the buildup of settled materials generated during the course of normal operations and from normal Ward Creek drainage. It was therefore necessary for KPC to dredge selected areas of Ward Cove as a means of maintaining navigable water depths. The areas dredged each year vary based on operational requirements. KPC dredges the area beneath the sawmill log lift (where wood debris accumulates) each year and dredges portions of the maintained moorage areas, shown in Figure 1-3, as needed. Currently, marine activities are limited to the transport of logs and hog fuel to the KPC sawmill and the transport of wood chips and hog fuel from the sawmill.

1.3 CONCEPTUAL MODEL

A conceptual site model for Ward Cove sediments is presented in Figure 1-4. The model identifies potential human and ecological receptors in the Cove and the major pathways by which they may be exposed to CoPCs from sediments. The conceptual model provides the general framework for the human health and ecological assessments described in Sections 6 and 7. Shutdown of the KPC facility in March 1997 eliminated discharges to the Cove from the facility's pulp processes. In the future, the major potential source of CoPCs related to the KPC facility will therefore be the Cove sediments.

Recreational anglers are the most likely human receptors at Ward Cove. Conversations with staff of the Alaska Department of Fish and Game (ADFG) indicate that salmon are the most popular fishes harvested, that most fishing takes place near Ward Creek, and that shellfish consumption rates are uncertain and likely to be low (Freeman 1995, pers. comm.). Alaska State regulations designate Ward Cove as a nonsubsistence area (per 18 AAC Parts 1, 2, and 99). Ward Cove is not designated for Customary and Traditional use. A nonsubsistence area is an area or community where dependence upon subsistence is not a principal characteristic of the economy, culture, and way of life of the area or community [see 5 AAC 99.016(a)]. Ordinary fishing and gathering are allowed.

Current site conditions also limit the potential for fishing or collecting shellfish; however, to be protective of all possible populations who might use the site now or in the future, the human health assessment in Section 6 evaluates risks for subsistence anglers who might collect fish and shellfish from areas with the most affected sediments. Potential risks for subsistence-level anglers are evaluated using CoPC concentrations in tissues of salmon and mussels collected in Ward Cove and estimates of CoPC concentrations in tissues of marine organisms derived from sediment data.

Human populations could hypothetically come into contact with CoPCs in sediments through direct contact with sediments. Because of the depth of the water overlying affected sediments and the cold climate, little or no direct contact with sediments is expected in Ward Cove. While recreational use of the lower portion of Ward Creek may result in contact with sediments, transport of site-related CoPCs to this area is not expected. Thus, direct human contact with CoPCs in sediments is highly unlikely. However, to provide a worst-case analysis, this pathway is addressed in the uncertainty assessment discussed in Section 6.1.1.

The major groups of ecological receptors in Ward Cove include plankton, benthic invertebrates, fishes, birds, and marine mammals (Figure 1-4). These receptors may be exposed to CoPCs from Cove sediments by interactions with the sediments, water, or biota from the Cove. CoPCs identified for Ward Cove have strong particle affinities and would be expected to associate with particles and settle to the bottom of the Cove. Therefore, the most likely exposure routes are through contact with sediments or by consumption of organisms that are part of the food web that originates with sediments. Therefore, it is unlikely that plankton, filter-feeding intertidal invertebrates, or planktivorous fishes are at substantial risk of exposure to CoPCs from Ward Cove sediments.

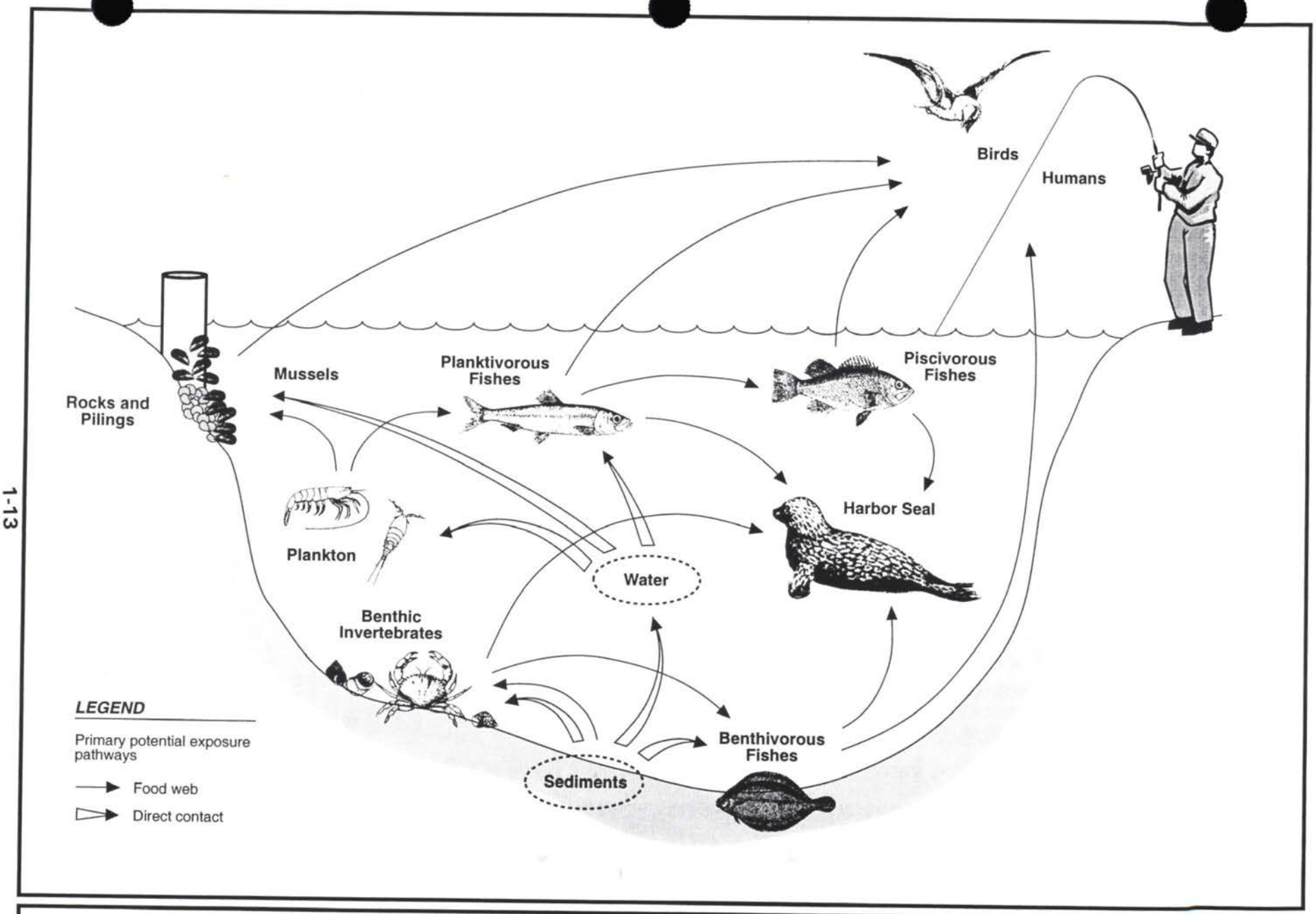

Figure 1-4. Conceptual model of potential exposure pathways from Ward Cove sediments.

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Chemicals in sediments can be transferred to benthic invertebrates (including both epifaunal and infaunal species) by direct contact with sediments, by consumption of organic matter in sediments, or by consumption of other benthic invertebrates. Chemicals can be transferred to benthivorous fishes by direct contact with sediments or by consumption of benthic invertebrates. Chemicals can be transferred to piscivorous fishes, birds, and marine mammals primarily by consumption of fishes that are part of the food web that originates with sediments.

1.4 DOCUMENT ORGANIZATION

The DTSR is presented in two volumes: Volume I is the main text of the DTSR, and Volume II is all related appendices. The remainder of this document, Volume I, consists of the following sections:

- *Ward Cove and KPC Facility Investigations* (Section 2) describes the ongoing KPC facility uplands RI/FS, provides a brief overview of the results of historical investigations of Ward Cove, and describes the overall design and methods used for the Ward Cove sediment investigation.
- *Physical Characteristics of Ward Cove* (Section 3) describes local meteorology; the bathymetry, water circulation, and major physical features of Ward Cove; and demography, land use, ecology of the surrounding area, and local ecology.
- *Nature and Extent of Chemicals of Potential Concern* (Section 4) describes sources of chemicals to Ward Cove, their vertical and horizontal distribution in Ward Cove sediments, and their concentrations in tissues of marine organisms.
- *Chemical Transport and Fate* (Section 5) describes the processes affecting the transport and fate of chemicals in Ward Cove.
- *Baseline Human Health Risk Assessment* (Section 6) presents an evaluation of sediment data and applicable tissue data in terms of potential human health risk.
- *Ecological Evaluation* (Section 7) presents an evaluation of sediment data and applicable tissue data in terms of potential sediment toxicity and food-web bioaccumulation.
- *Delineation of Area of Concern* (Section 8) identifies and prioritizes problem areas subject to natural recovery or remedial activities.
- *Natural Recovery* (Section 9) presents an evaluation of the potential for sediments to recover naturally.

- *Technology Screening* (Section 10) describes physical constraints, screening criteria, and remedial technologies that are potentially applicable to Ward Cove sediments.
- *Assembly of Alternatives and Detailed Evaluation* (Section 11) develops viable remedial alternatives from the technologies and process options identified in Section 10 and provides a detailed evaluation of their suitability. A recommended alternative is also identified in Section 11.
- *References* (Section 12) provides a list of all documents cited in this report.

The interrelationship between Sections 4 through 11 and key decision points is shown in Figure 1-2.

Appendices are provided in Volume II. All Phase 1 and Phase 2 data are provided in Appendix A. Quality assurance review summaries for 1996 and 1997 results are provided in Appendix B. Sediment core and compositing information, along with copies of the detailed core logs, is provided in Appendix C. Historical bioaccumulation data are provided in Appendix D. An assessment of the vertical extent of mill-impacted sediment by ENSR and the initial estimation of total organic carbon (TOC) in sediments predating mill activities in Ward Cove is provided in Appendix E. Results of the model used to simulate sediment processes leading to reductions in CoPC concentrations are presented in Appendix F. An evaluation of site risks based on the maximum sediment chemical concentrations identified in 1994, 1995, or the present investigation is provided in Appendix G. Key factors used to prepare the human health risk assessment are provided in Appendix H. Scatter plots for CoPC concentrations and sediment toxicity results for 1996 and 1997 are presented in Appendix I. A comparison of various sediment quality values for metals, PAH compounds, and total PCBs is provided in Appendix J. Details of the selected remedial technologies are provided in Appendix K. Potential applicable or relevant and appropriate requirements (ARARs) and to-be-considered (TBC) criteria for the Ward Cove sediment remediation project are discussed in Appendix L.

2. WARD COVE AND KPC FACILITY INVESTIGATIONS

The KPC site is being evaluated under two separate investigations: 1) Uplands Operable Unit—the uplands RI/FS, which includes the former mill areas and other upland areas that may have been affected by past operations, and 2) Marine Operable Unit—the Ward Cove sediment remediation project. In this section, the uplands investigation is briefly described, historical Ward Cove investigations are summarized, and the design and methods associated with the Ward Cove sediment investigation are provided. The results of previous investigations of Ward Cove provided the basis for the design of the studies conducted in the current Ward Cove sediment remediation project.

2.1 UPLANDS INVESTIGATION

Parallel with the development of this DTSR for Ward Cove, KPC is conducting an RI/FS for the Uplands Operable Unit of the former facility under a Consent Order providing joint oversight by EPA and ADEC. The uplands RI/FS has proceeded through sampling of uplands site and offsite areas of potential concern. The methods used in conducting the RI/FS are presented in the following documents:

- *Scoping Document for the RI/FS* (PTI 1997c): This document provides summaries of relevant historical information, provides a preliminary conceptual site model, presents a preliminary list of ARARs, and establishes a decision-making framework to be used by the agency and KPC project managers throughout the RI/FS.
- *Compilation of Existing Data, Ketchikan Pulp Company Site* (PTI 1997a): This document summarizes data gathered during uplands investigations prior to July of 1997, including data from routine monitoring events. It was prepared to supplement the scoping document for the RI/FS.
- *Work Plan for the RI/FS* (PTI 1997h): The work plan provides a review of relevant information including analytical results of source material samples and identifies samples and analyses to be conducted during uplands sampling. The work plan also identifies procedures to be used in transport and fate analyses and risk assessments to be conducted based on site data.
- *Technical Memoranda* (PTI 1997d,e): Technical memoranda have also been prepared documenting the results of aerial deposition modeling and identifying soil sampling locations to evaluate aerial deposition.

- *Preliminary Site Characterization* (PTI 1997i): This report summarizes uplands site data and identifies CoPCs in uplands site areas.
- *Remedial Investigation* (Exponent 1998): This report, which includes a baseline risk assessment, was submitted in October 1998.

Findings of the uplands investigation that are most relevant to the Ward Cove sediment remediation project pertain to potential sources of CoPCs. Source information is summarized in Section 1 and discussed in greater detail in Section 4.1.

2.2 PREVIOUS STUDIES OF WARD COVE

Numerous environmental studies of Ward Cove have been conducted to evaluate the potential environmental effects associated with discharges from the KPC facility. The major findings of these studies are described in the technical studies work plan (PTI 1996) and are summarized below. More detail is provided on studies of sediment contamination and associated biological effects occurring between 1988 to 1995 than for the earlier historical studies (1951–1974) because the more recent studies provided the most relevant information and were used to design the technical studies conducted for the Ward Cove sediment remediation project.

In addition, in 1997, an expanded site investigation (E&E 1998) was performed at the KPC site to provide EPA with adequate information to determine whether the site is eligible for placement on the National Priorities List based on the Hazard Ranking System. This work was separate from the detailed technical studies (i.e., presented in this report). The expanded site investigation data were considered in Appendix G as part of the human health and ecological risk assessments. However, these data were not used to delineate the AOC because of problems associated with the accuracy of the station locations (U.S. EPA 1998f).

2.2.1 Historical Studies (1951–1974)

During 1951–1952, the Alaska Water Pollution Control Board (AWPCB) collected information on water column characteristics and plankton and benthic macroinvertebrate assemblages to characterize Ward Cove baseline conditions prior to the opening of the KPC facility (AWPCB 1953). A follow-up study was then conducted during 1955–1957, after the KPC facility had been operating for more than a year (AWPCB 1957). Observed environmental effects potentially associated with the facility included oxygen depletion in the water column near the facility and dead clams near the facility. During that period, fish kills were also reported by the U.S. Fish and Wildlife Service (USFWS) and fishermen, and dying fishes were sometimes observed during the AWPCB study.

In August 1965, the Federal Water Pollution Control Administration evaluated water quality in Ward Cove (FWPCA 1965). Concentrations of dissolved oxygen in near-surface and near-bottom waters were found to be less than 2 mg/L, whereas concentrations at mid-depths were between 5 and 6 mg/L. Concentrations in Tongass Narrows

exceeded 7 mg/L. The near-surface declines in dissolved oxygen concentrations were attributed to the presence of KPC effluent in the water column. The near-bottom declines were attributed to the elevated oxygen demand of settleable solids from facility discharges.

During 1968–1969, the Federal Water Quality Administration evaluated water quality in Ward Cove and in Tongass Narrows near the mouth of the Cove (FWQA 1970). Benthic invertebrate assemblages and intertidal blue mussel (*Mytilus edulis)* populations were also evaluated. The study concluded that 40 percent of the dissolved oxygen measurements were less than 6 mg/L (the Alaska water quality standard) and sulfite waste liquor concentrations were high (greater than 50 mg/L) throughout the surface waters of the Cove and in the Tongass Narrows near the mouth of the Cove. In addition, mussel abundances in the Cove generally were lower than in Tongass Narrows, and the lowest mussel abundances were found at the stations located downcurrent from the KPC facility, along the north shoreline toward the mouth of the Cove. Also, black, organic-rich deposits blanketed much of the bottom of the Cove, and few benthic invertebrates were observed.

In September 1974, U.S. EPA (1975) evaluated water quality in Ward Cove and Tongass Narrows and concluded that conditions had not improved since the 1968–1969 study conducted by FWQA (1970). EPA's study was also the first to measure chemical concentrations in sediments. Evaluations of sediments at eight stations in the Cove revealed high concentrations of total volatile solids (TVS), organic nitrogen, total sulfides, and chemical oxygen demand (COD). Macroscopic evaluation of sediment samples revealed that polychaetes were common, except at the two stations closest to the KPC facility, where no organisms were observed. U.S. EPA (1975) concluded that the increase in polychaete abundances relative to the 1968–1969 study indicated that installation of the primary treatment system for facility wastewater resulted in improved bottom conditions.

2.2.2 Recent Sediment Chemistry and Toxicity Studies (1988–1995)

Information on Ward Cove sediments that was collected as part of studies conducted between 1988 and 1995 was used to develop a preliminary list of CoPCs and ecological receptors for the Cove and subsequently to design the technical studies conducted as part of the current Ward Cove sediment remediation project. The major elements and conclusions of these studies are presented below. Additional details on sampling locations, chemical concentrations, and toxicity are provided in PTI (1996).

At present, sediment quality criteria are not available for the State of Alaska. Therefore, Washington State sediment quality standards (SQSs) were used in this section to evaluate the concentrations of chemicals found in Ward Cove sediments. The Washington State SQSs are considered appropriate for evaluation of sediment chemical concentrations in Ward Cove for several reasons. First, they are environmentally protective because they have been adopted by the State of Washington to "correspond to a sediment quality that will result in no adverse effects, including no acute or chronic adverse effects on biological resources" (Ecology 1995). Second, they are credible because they have received extensive scientific and public review. Finally, they have some natural applicability to

the marine waters of southeastern Alaska because they are considered protective of Puget Sound marine species, many of which are found in southeastern Alaska, including Ward Cove.

2.2.2.1 1988 Sediment Study

Jones & Stokes and Kinnetic (1989) sampled sediments at 26 stations throughout Ward Cove in August and September of 1988. Sediment samples were analyzed for TOC, total sulfide, BOD, oil and grease, and nine metals (arsenic, cadmium, chromium, copper, lead, mercury, nickel, silver, and zinc). The toxicity of whole sediments was evaluated at five stations near the KPC facility using the 10-day amphipod test with *Rhepoxynius abronius*, and the toxicity of sediment elutriates was evaluated at three stations near the facility using the 96-hour test with the mysid *Acanthomysis sculpta.*

Jones & Stokes and Kinnetic (1989) found that sediments throughout most of Ward Cove were black in color, but varied greatly with respect to texture and the presence of debris, such as wood fiber, wood chips, bark, twigs, and logs. A relatively large area near the head of the Cove could not be sampled because of the extensive amount of debris that was present. Sediments in the vicinity of the KPC facility generally were fine-grained in texture. Sediments collected directly off the KPC dock were "oily and consisted of a black ooze mixed with some bark." Sediments collected down the shoreline from the facility "were viscous and gelatinous in texture . . . Most of the samples appeared to be dominated by fiber mats . . ."

Jones & Stokes and Kinnetic (1989) found that concentrations of TOC, total sulfide, BOD, and oil and grease generally declined with increasing distance from the KPC facility. High values of total sulfide, BOD, and oil and grease were also found offshore from the fish cannery. Metals concentrations generally were highest off the KPC facility. However, a Washington State SQS was exceeded in only one instance. The concentration of cadmium at one location exceeded the SQS of 5.1 mg/kg by a small margin.

Amphipod survival ranged from 7 to 64 percent off the facility and was significantly different ($P \leq 0.05$) from the reference value of 100 percent at all five stations from Ward Cove. Jones & Stokes and Kinnetic (1989) note that the highest value of amphipod survival in the Cove was associated with the highest metals concentrations and that the particle size of Ward Cove sediments may have contributed to the observed levels of toxicity in the amphipod test. The LC50 values for the mysid test ranged from 40 percent to greater than 100 percent and were significantly different from the reference value at only one station.

Jones & Stokes and Kinnetic (1989) concluded that high sulfide content, oxygen demand, or unmeasured organic contaminants in Ward Cove sediments may pose a greater risk to biota than do metals. This conclusion was supported by the relatively low concentrations of metals found throughout the Cove and the fact that several sediment elutriate samples were not found to be highly toxic in the mysid test. Because the sediment elutriates were

well-aerated during testing, oxygen consumption by sulfides and organic material was factored out of those tests.

2.2.2.2 1992 Sediment Study

EVS (1992) sampled sediments at five stations in the inner part of Ward Cove in January 1992. Two stations were sampled in Moser Bay, a documented reference area located approximately 17 km north of Ward Cove. Sediment samples were analyzed for TOC, grain size, pH, alkalinity, chloride, sulfate, and four leachable metals (calcium, magnesium, potassium, and sodium). The toxicity of whole sediments was evaluated at all five stations using the 10-day amphipod test with *Rhepoxynius abronius*. Benthic macroinvertebrate assemblages were also evaluated microscopically at all five stations. This evaluation represents the most detailed and quantitative characterization of benthic macroinvertebrate assemblages in the Cove.

EVS (1992) found that TOC concentrations in Ward Cove sediments ranged from 14 percent on the southeast shoreline to 37 percent off the KPC facility. In general, TOC concentrations declined with increasing distance from the facility. TOC concentrations in Moser Bay ranged from 4 to 5 percent.

Amphipod survival in Ward Cove ranged from 4 percent off the KPC facility to 94 percent near the mouth of Ward Creek and was significantly different ($P \leq 0.05$) from the control only at the two stations closest to the facility. The values of survival at the remaining three stations in Ward Cove (88–94 percent) were similar to the values of survival found in Moser Bay (91–93 percent).

Taxa richness (i.e., total number of taxa) of benthic macroinvertebrate assemblages in Ward Cove was considerably lower than taxa richness in Moser Bay. The lowest richness values in Ward Cove (2–3 taxa) were found at the two stations off the KPC facility. Total abundance of benthic macroinvertebrate assemblages at 20 m depth in Ward Cove (51–891 individuals per sample) was also considerably lower than total abundance in Moser Bay (1,449 individuals per sample). By contrast, total abundance at 40 m depth in Ward Cove (1,064 individuals per sample) was greater than the value for Moser Bay (670 individuals per sample). The lowest value of total abundance in Ward Cove (51 individuals per sample) was found off the KPC facility. At both depths in Ward Cove, benthic assemblages were dominated by polychaetes (primarily the opportunistic species *Capitella capitata*) and nematodes, whereas assemblages in Moser Bay were dominated by molluscs (primarily *Axinopsida serricata*).

EVS (1992) concluded that the aquatic environment of Ward Cove has been perturbed by activities on the adjacent land. The benthic macroinvertebrate assemblages found in the Cove were considered characteristic of areas affected by high levels of organic enrichment. The authors recommended that future sediment studies focus on areas near the KPC facility.

2.2.2.3 1994–1995 Sediment Study

ENSR (1995b) sampled surface sediments at eight stations throughout Ward Cove and one station in the Tongass Narrows in November 1994 as part of the sediment monitoring component of the KPC NPDES permit. Although an additional three stations near the head of the Cove were specified in the permit, sediment samples could not be collected from those locations because of the presence of debris. ENSR (1995b) also sampled surface sediments at one station in Ward Cove and one reference station in Moser Bay in February 1995 as part of the bioaccumulation monitoring component of the KPC NPDES permit.

Sediment samples in both ENSR studies were analyzed for TOC, grain size, total sulfide, acid-volatile sulfide (AVS), BOD, COD, four metals (arsenic, cadmium, methylmercury, and zinc) and various organic compounds (dioxins and furans, PAHs, phenol, 4-methylphenol, and benzoic acid). Toxicity of whole sediments was evaluated at all nine stations sampled by ENSR (1995b) in 1994 using the 10-day amphipod test with *Rhepoxynius abronius*, and the toxicity of sediment elutriates was evaluated at those stations using the 96-hour echinoderm embryo test with the purple sea urchin *Strongylocentrotus purpuratus*.

Concentrations of TOC, total sulfide, AVS, BOD, and COD generally decreased with increasing distance from the facility. Total sulfide, AVS, and BOD were also elevated near the cannery. Concentrations of total sulfide, AVS, BOD, and COD were also relatively high near the facility. Concentrations of all four metals generally were highest off the KPC facility or the cannery and declined with increasing distance from those facilities. Washington State SQSs are available for arsenic, cadmium, and zinc. Although the SQS for arsenic was not exceeded at any station, relatively minor exceedances were found for cadmium and zinc. It should be noted that the 1995 data set for zinc is considered unreliable because all 1995 concentrations are inconsistent with the concentrations found in 1994, 1996, and 1997.

Concentrations of most organic compounds were also highest off the KPC facility and the cannery and declined with increasing distance from those facilities. Washington State SQSs are available for phenol, 4-methylphenol, benzoic acid, and PAHs. The SQSs for benzoic acid and all PAHs (conservatively normalized to 1 percent sediment organic carbon content) were not exceeded at any station. Exceedances of SQSs were found for phenol and 4-methylphenol. The greatest exceedance of the SQS for phenol was found at a station downcurrent from the KPC facility, where the observed concentration was more than 14 times greater than the SQS. The greatest exceedances of the SQS for 4-methylphenol were found at Station 43 off the KPC facility and Station WC1 downcurrent from the facility, where the observed concentrations were approximately 13 and 65 times greater than the SQS, respectively.

Although no sediment quality values exist for dioxins and furans, U.S. EPA (1993a) estimates that sediment concentrations of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD) greater than 50 ng/kg may pose a low risk to fishes. 2,3,7,8-TCDD was detected only at one station off the KPC facility, and that concentration was 20 times lower than the EPA risk value. Although 2,3,7,8-TCDD was rarely detected in Ward Cove sediments, other dioxin and furan congeners were detected. The concentrations of those congeners were converted into equivalent concentrations of 2,3,7,8-TCDD by ENSR (1995b) using the equivalent concentration factors recommended by U.S. EPA (1989c) and then summed to estimate the total toxic equivalent concentrations (TECs). The TEC values were highest off the KPC facility (16–46 ng/kg) and declined with increasing distance from the facility.

For the sediment toxicity tests, amphipod survival in Ward Cove ranged from 0 percent off the KPC facility to 90 percent in the outer Cove and was significantly different ($P \leq 0.05$) from the reference value of 94 percent at all but one station. Survival at all stations off the KPC facility, along the north shoreline, and off the cannery was very low (less than 10 percent).

Echinoderm survival in sediment elutriates from Ward Cove ranged from 4 percent off the KPC facility to 45 percent down the shoreline from the facility and was significantly different ($P \leq 0.05$) from the reference value of 67 percent at all stations. Echinoderm normality (i.e., development to the pluteus stage) in Ward Cove ranged from 62 percent adjacent to the facility to 82 percent down the shoreline from the facility and was significantly different ($P \leq 0.05$) from the reference value of 93 percent at all stations.

2.2.2.4 Summary of Recent Sediment Studies

Results of the studies of sediment chemistry and associated biological effects conducted in Ward Cove between 1988 and 1995 indicate that surface sediments in parts of the Cove were characterized by elevated concentrations of selected metals, organic compounds, and conventional variables. The concentrations of most substances were highest near the KPC facility and the cannery and declined with increasing distance from those facilities. For the 8 metals and 20 organic compounds having Washington State SQSs, concentrations of 2 metals (cadmium and zinc) and 2 organic compounds (phenol and 4-methylphenol) exceeded their SQSs at one or more stations in the Cove. However, neither arsenic nor any of the 17 PAH compounds evaluated exceeded their SQSs. In addition, concentrations of 2,3,7,8-TCDD were 20 times lower than the level that may pose a low risk to fishes, as estimated by EPA. Several conventional variables, including TOC, total sulfide, AVS, BOD, and COD, were also elevated in sediments from parts of Ward Cove relative to the concentrations typically found in shallow marine sediments. However, relatively high concentrations of total sulfide, AVS, BOD, and COD were also found in sediments from Moser Bay, suggesting that these variables may be naturally elevated in sediments from parts of southeastern Alaska. As noted earlier, the 1995 data set for zinc is considered unreliable.

The results of the toxicity tests conducted on Ward Cove sediments were somewhat contradictory. Although all tests identified sediments immediately off the KPC facility as being toxic, results for sediments from other parts of the Cove did not always agree. Jones & Stokes and Kinnetic (1989) suggest that contradictory results were found between the amphipod and mysid tests because the effects of oxygen-demanding substances were largely factored out of the latter tests (i.e., the test chambers were well-aerated during testing). However, contradictory results were also found among the three sets of amphipod tests conducted in 1988, 1992, and 1994 using nearly identical protocols. In 1988 and 1994, sediments were found to be toxic to amphipods throughout most of the Cove. By contrast, sediments were found to be toxic to amphipods only near the KPC facility in the 1992 study. The reason for these contradictions in the amphipod test results is unknown.

2.2.3 Tissue Chemistry Studies

The bioaccumulation of total mercury, methylmercury, and PCDDs/Fs in fish, crab, and bivalves from Ward Cove has been evaluated in several recent studies, and PCDD/F data are available for seals killed by subsistence hunters in the Ketchikan area. In addition, data are available for mercury and PCDDs/Fs in sediments and tissues of several marine species collected near the former Alaska Pulp Corporation (APC) mill in Sitka, Alaska, which shared some common operational characteristics. These studies and their results are described briefly here, and the Ward Cove data are further summarized in the Ward Cove technical studies work plan (PTI 1996). Summary tables of relevant data are included in Appendix D (Tables D1-1, D1-2, D1-3, D1-4, D1-5, and D2-1). Concentrations of PCDDs/Fs are presented in the current investigation as TECs with undetected congeners included in the TEC as one-half the detection limit except as indicated. This approach was also applied to historical data sets, where sufficient PCDD/F data were reported (i.e., where concentrations or detection limits were available for all relevant congeners).

2.2.3.1 Ward Cove Data

In 1990, ADEC collected three pink salmon (*Oncorhynchus gorbuscha*) and seven sockeye salmon (*Oncorhynchus nerka*) from a site in Ward Creek and three pink salmon from a site in Signal Creek (Figure D-1 in Appendix D; Spannagel 1991). All salmon collected were adults. The concentrations of PCDDs/Fs were measured in five composite samples of whole bodies or livers. One of the composites was analyzed after removal of the livers, and a composite of the livers was analyzed separately. Concentrations of 2,3,7,8-tetrachlorodibenzofuran (2,3,7,8-TCDF) ranged from 0.45 ng/kg in the composite without the livers to 1.8 ng/kg in the liver composite (Table D1-1 in Appendix D1). No other PCDD/F congeners were detected. (Detection limits of undetected congeners were not provided by Spannagel [1991] and thus TECs could not be calculated from these data.) PCDD/F concentrations in female salmon from this investigation may have been reduced because salmon were collected post-spawning and transfer of PCDDs/Fs in maternal lipids to eggs would have reduced PCDDs/Fs in maternal tissues.

In 1991, ADEC collected five Dungeness crabs (*Cancer magister*) and seven rockfishes (*Sebastes* sp.) from four stations in Ward Cove (Figure D-1 in Appendix D; Spannagel 1991). In addition, five adult pink salmon were collected from a reference area off Mountain Point, approximately 18 km southeast of Ward Cove. Concentrations of PCDDs/Fs were measured in composite samples of crab muscle, crab hepatopancreas, rockfish fillets, and salmon fillets (Tables D1-2 and D1-4 in Appendix D1). Concentrations of PCDDs/Fs were highest in the hepatopancreas and muscle tissues of crabs from Ward Cove and lowest in fillets of pink salmon from Mountain Point. Although these samples are not directly comparable, TECs in muscle tissue of rockfishes from Ward Cove were similar (0.26 ng/kg wet weight) to TECs in salmon from the reference location at Mountain Point (0.23 ng/kg wet weight) suggesting minimal or no effect from the site.

As part of the monitoring requirements of the KPC NPDES permit, controlled exposure experiments were conducted in the spring of 1995 (ENSR 1995a) and in December 1995 (EVS 1996) to evaluate the bioaccumulation of mercury, methylmercury, and PCDDs/Fs in clams (*Macoma nasuta*) and mussels (*Mytilus trossulus*). Results for the first study (ENSR 1995a) showed maximum concentrations of TCDD and TCDF homologous groups of 0.37 ng/kg wet weight (TECs were not calculated because of the predominant number of samples for which PCDDs/Fs were undetected) and no detections of mercury at 0.1 mg/kg. Results of ENSR (1995a) are further summarized in the work plan (PTI 1996).

Results for the second study (EVS 1996) became available after the work plan was prepared and are provided in Appendix D of this document. These results include tissue concentrations for mussels exposed *in situ* to the effluent from Outfall 001 and for clams exposed to the sediments underlying the effluent plume during a laboratory bioassay. A map of station locations is also included in Appendix D. This investigation yielded the highest TECs identified in historical data reviewed for Ward Cove (i.e., 2.32 ng/kg wet weight, in whole bodies of mussels) (EVS 1996). However, in conversations with EVS staff, it was discovered that TECs had been calculated incorrectly by counting replicate analyses as individual results. When this error was corrected, a maximum TEC of 0.78 ng/kg wet weight was calculated (Salazar 1998, pers. comm.) (Table D1-5).

The National Marine Fisheries (Triangle Labs 1996) reported concentrations of PCDDs/Fs in blubber from five seals killed by subsistence hunters in the Ketchikan area (i.e., four near Tatoosh Island and one in Coon Cove). PCDDs/Fs were predominantly undetected in one sample from each of the five seals (Table D1-3 in Appendix D1). In three samples, there were no detections of PCDD/F congeners considered by EPA to pose a human health threat (i.e., PCDD/F congeners substituted with chlorine at the 2,3,7,8-positions). Only three relevant PCDD/F congeners were detected in the fourth sample, and one relevant congener was detected in the fifth sample (Table D1-3 in Appendix D1).

TECs for PCDDs/Fs of 5.4 and 5.5 ng/kg were calculated for the two samples with at least one detected congener, using a value of one-half the detection limit for each relevant

undetected PCDD/F congener. TECs of 0.40 and 0.0079 ng/kg were calculated when undetected congeners were excluded from the calculations (Table D1-3 in Appendix D1). For the three samples in which congeners were not detected, a maximum TEC of 29.3 ng/kg was calculated using the full detection limits for undetected congeners, and a maximum TEC of 14.7 ng/kg was calculated using one-half the detection limits for undetected congeners (Table D1-3 in Appendix D1). Use of such an assumption greatly overestimates actual concentrations. Risk estimates for exposure to PCDDs/Fs in seal tissues are provided in Appendix H.

2.2.3.2 Data Collected near APC Mill in Sitka, Alaska

As part of the remedial investigation for the APC mill, 26 sediment samples, 4 mussel samples, and 1 rockfish sample were collected from West Sawmill Cove near the former APC mill and analyzed for PCDDs/Fs (Table D2-1, Appendix D2). PCDD/F concentrations in tissues and sediments from six other nearby marine locations in the Sitka, Alaska, area were also reported (Foster Wheeler 1997) (Table D2-1 and Figure 3-1 in Appendix D). As in the Ward Cove investigation, results for PCDDs/Fs are shown as TECs, and in calculating TECs, one-half the detection limit was used for congeners that were not detected.

Results from the West Sawmill Cove data are judged to be the most comparable with PCDD/F sediment sampling results in the Ward Cove investigation because of the similar ranges in TECs in sediments. Specifically, PCDD/F TECs in 26 sediment samples collected from West Sawmill Cove ranged from 4.13 to 54 ng/kg (dry weight) with a mean of 17.4 ng/kg (Table D2-1 in Appendix D2), while TECs in 42 sediment samples from Ward Cove ranged from 1.1 to 46 ng/kg (dry weight) with a median of 15 ng/kg. TOC concentrations were also similar, ranging from 1 to 42 percent in West Sawmill Cove (Table D2-1 in Appendix D2) and from 10 to 40 percent in Ward Cove.

PCDD/F concentrations in four mussel samples collected from West Sawmill Cove ranged from 0.37 to 4.5 ng/kg wet weight, and the PCDD/F concentration in one rockfish fillet was 0.004 ng/kg wet weight (Table D2-1 in Appendix D2).

In Ward Cove, PCDD/F concentrations in mussels were somewhat lower than those measured in West Sawmill Cove (i.e., PCDD/F concentrations in mussels ranged from 0.18 to 0.78 ng/kg wet weight [Table D1-5 in Appendix D]). The concentration of PCDDs/Fs in a composite of five rockfish collected in or near Ward Cove in 1991 was 0.26 ng/kg wet weight (Table D1-2 in Appendix D1). The higher maximum TECs in the Ward Cove data could be due to higher detection limits available in the 1991 analyses relative to those reported by Foster Wheeler (1997).

The finding of similarly low concentrations of PCDDs/Fs in tissues collected in the APC investigation where sediment concentrations were similar to those in Ward Cove provides further evidence that bioaccumulation of PCDDs/Fs is limited.

2.3 WARD COVE SEDIMENT REMEDIATION PROJECT INVESTIGATION

Field studies performed during the Ward Cove sediment remediation project were conducted in two phases. An overview of the study design is illustrated in Figure 2-1. The overall objectives of Phase 1 sampling were to delineate areas of focus (AOFs) within Ward Cove, establish relationships between sediment toxicity and chemical and/or organic-derived wastes (e.g., ammonia, sulfide), establish sediment conditions prior to initiation of wood pulping activities in Ward Cove, and identify CoPCs to carry forward into the Phase 2 evaluation. For efficiency and consistency, Phase 1 sampling was consolidated with the 1996 sediment sampling conducted under KPC's NPDES permit. The study design for Phase 2 was based on results obtained from the Phase 1 investigation. The objectives of Phase 2 sampling activities were to provide a detailed characterization of the physical features of Ward Cove (i.e., bathymetry, sediment surface and subsurface characteristics), refine the characterization of the horizontal extent of AOFs, evaluate natural recovery rates for sediment, distinguish removal and/or capping areas from no-action areas, and characterize bulk chemistry and engineering properties of sediments targeted for remediation. Phase 2 sampling was also consolidated with the 1997 sediment sampling conducted under KPC's NPDES permit.

Key elements of Phase 1 included surface sediment characterization (both chemistry and toxicity) and initial assessment of subsurface sediments (based on historical data). The target horizon for surface sediments was the top 10 cm of the sediment column. Key elements of Phase 2 included additional characterization of surface sediments (both chemistry and toxicity), chemical characterization of subsurface sediments, specialized toxicity testing, sediment accumulation rate measurements, bulk characterization of subsurface sediment to support an engineering assessment of disposal options, a geophysical survey of Ward Cove, and current measurements at selected locations.

During Phase 2, surface sediments from the margins of the AOFs (identified during Phase 1) were analyzed to better determine the horizontal extent of the AOFs potentially related to the KPC facility and to delineate areas requiring sediment removal from those areas where sediment may be left in place for capping or natural recovery. In addition, during Phase 2, composite samples from sediment cores were analyzed for chemical and engineering properties that may affect remediation options.

2.3.1 Phase 1

2.3.1.1 Phase 1 Study Design

Phase 1 sediment analyses were conducted to determine the physical, chemical, and toxicity characteristics of surface sediments throughout Ward Cove. As part of Phase 1, 28 stations were sampled in Ward Cove (Figure 2-2) and 2 stations were sampled in Moser Bay, the selected reference area (Figure 2-3). Sampling was conducted in May and June 1996. Twelve of the 28 stations in Ward Cove were stations identified in the sediment monitoring component of the KPC NPDES permit (Figure 2-2). The remaining

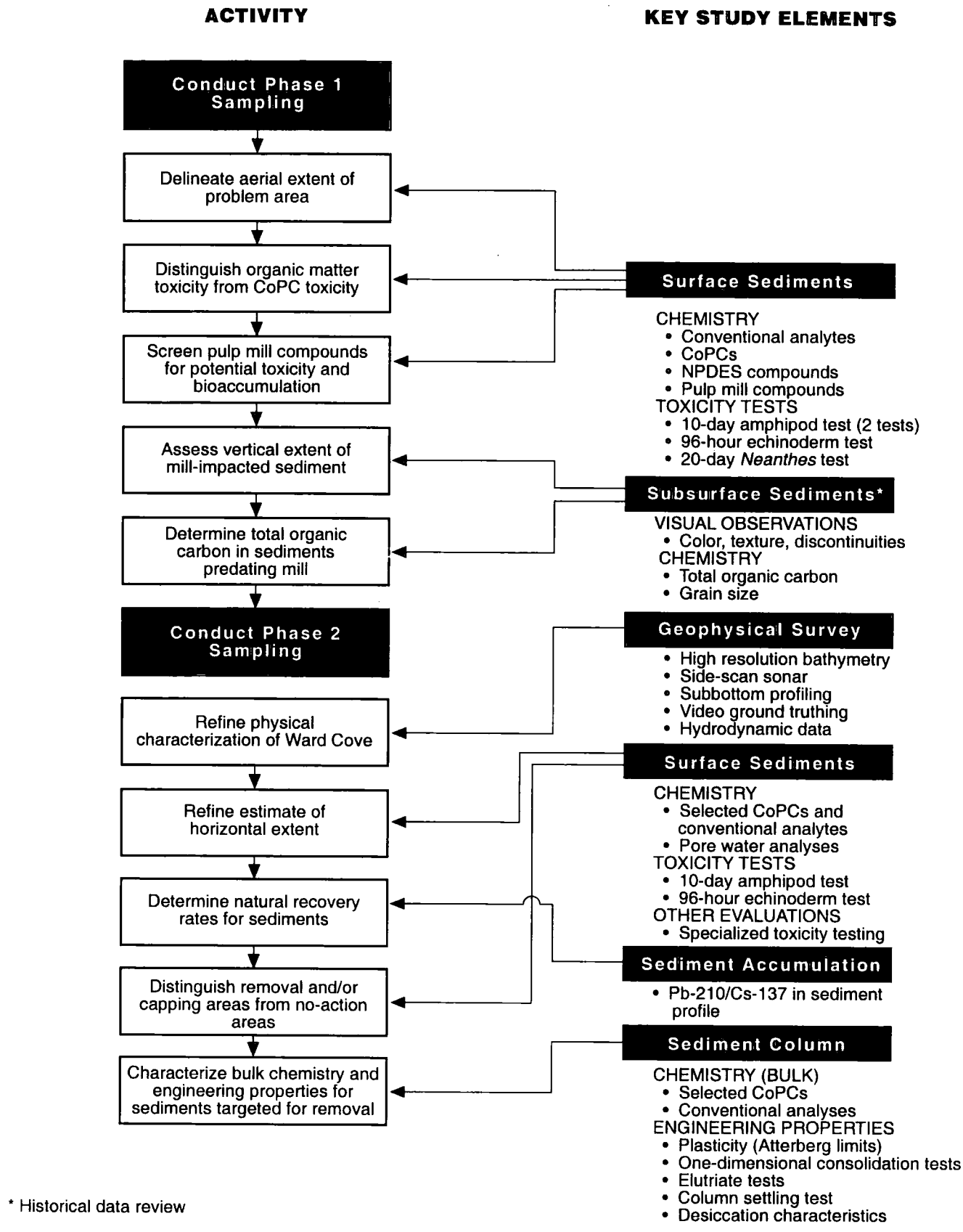

Figure 2-1. Overview of phased study design for the Ward Cove sediment remediation project.

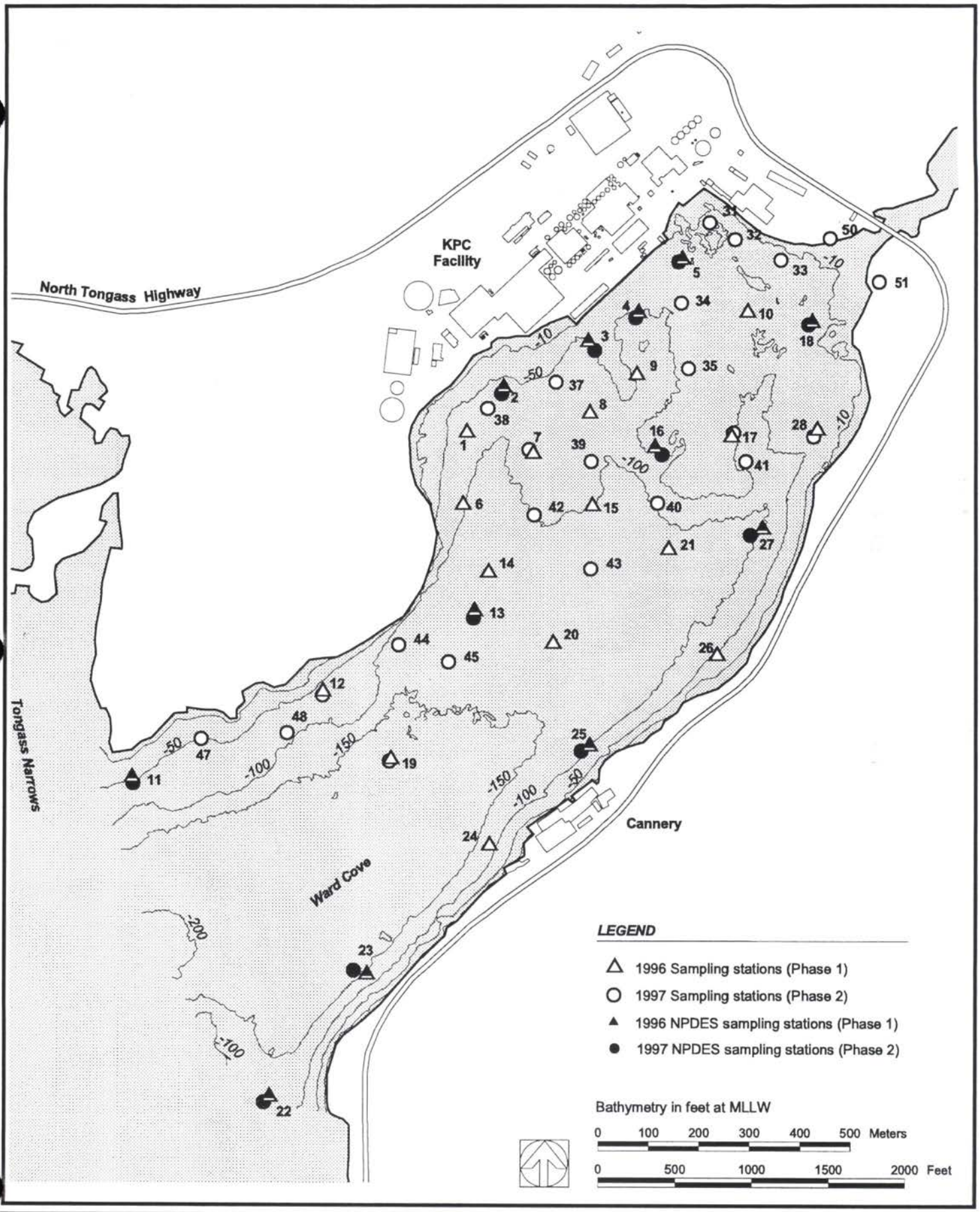

Figure 2-2. Station locations in Ward Cove at which surface sediment samples were collected in 1996 and 1997.

CB0W-16-01 06/15/98 GIS

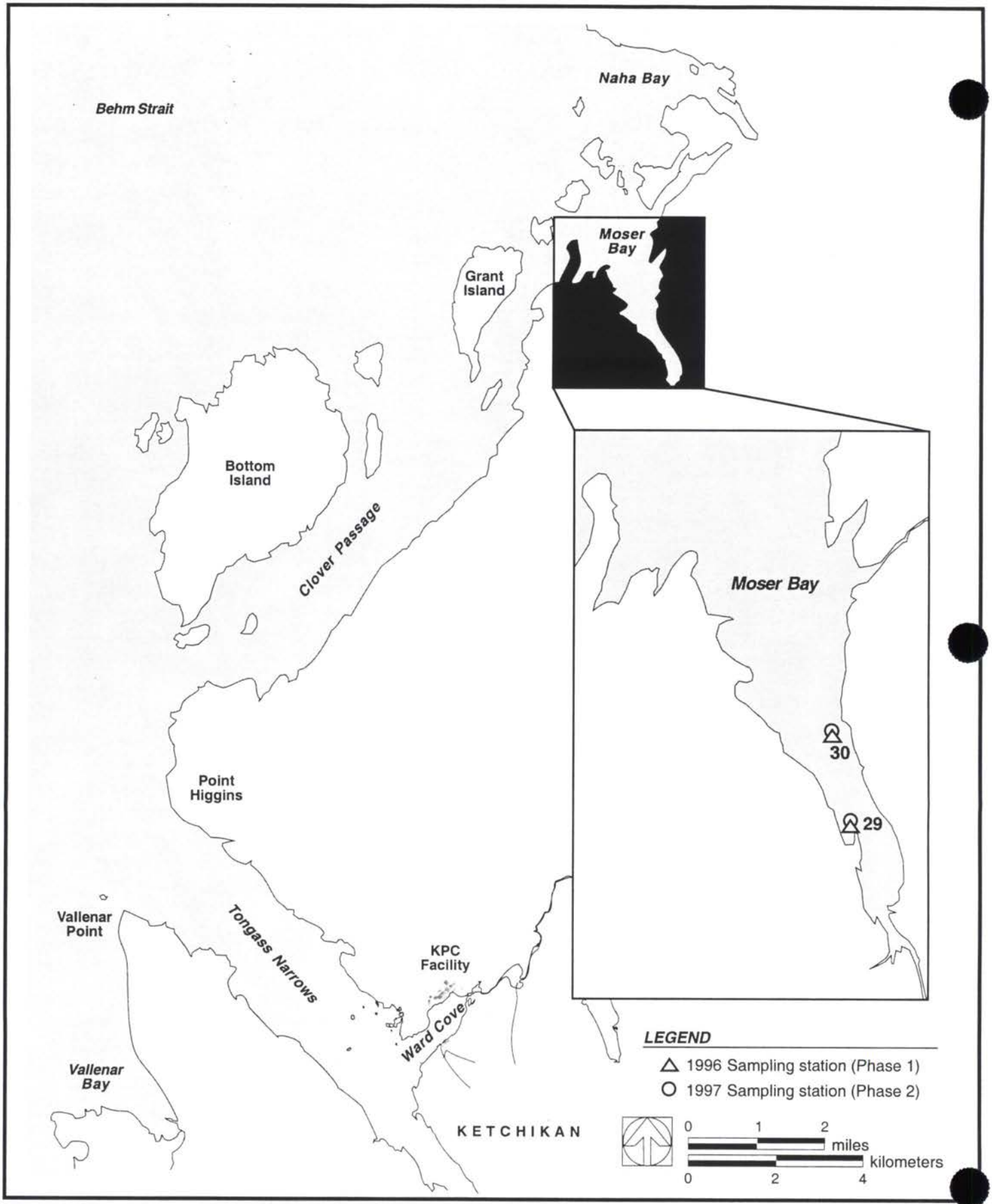

Figure 2-3. Station locations in Moser Bay at which surface sediments were collected in 1996 and 1997.

Phase 1 station locations in Ward Cove were selected to fill data gaps in the spatial distribution of NPDES stations, provide more detailed information on small-scale spatial distributions of CoPCs and sediment toxicity near the KPC facility, and evaluate potential relationships of CoPCs and sediment toxicity to the fish cannery located on the southern shoreline of the Cove. Two stations were sampled in Moser Bay to estimate concentrations of CoPCs and toxicity levels in a representative reference area of southeastern Alaska.

The same suite of sediment toxicity tests was evaluated for all 30 stations, but the suites of chemical analytes differed among stations (Table 2-1). The following three major groups of analytes were measured:

- **Group 1 (toxicity-related CoPCs and associated conventional analytes):** Most of these analytes were measured at all 30 stations. They include the chemicals identified as preliminary CoPCs in Section 3.3 of the work plan (PTI 1996) (i.e., TOC, total ammonia, total sulfide, BOD, COD, cadmium, total mercury, zinc, PCDDs/Fs, phenol, and 4-methylphenol), as well as additional conventional analytes (i.e., grain size and total solids) considered essential for interpreting results of the chemical analyses and toxicity tests.

- **Group 2 (additional NPDES analytes):** These analytes were measured at the 12 NPDES stations to satisfy the requirements of the KPC permit. They include the NPDES analytes that are not included in Group 1 (i.e., AVS, methylmercury, benzoic acid, PAH compounds, and extractable organic halides).

- **Group 3 (pulp mill compounds):** These analytes commonly associated with pulp mills (i.e., chlorinated phenols and related compounds and resin and fatty acids) were measured at five stations near the KPC facility to estimate the maximum concentrations found in Ward Cove and the near-field spatial patterns of these compounds.

- **Group 4 (bioaccumulation-related CoPCs):** These analytes include chemicals identified as CoPCs from a bioaccumulation standpoint in the Phase 1 work plan (mercury and PCDDs/Fs), as well as other chemicals found to be present at elevated concentrations (relative to reference conditions) throughout relatively large areas of Ward Cove (arsenic, cadmium, zinc, and PAH compounds).

The sediment toxicity tests included the following four tests:

- 10-day amphipod test using *Rhepoxynius abronius*
- 10-day amphipod test using *Leptocheirus plumulosus*

TABLE 2-1. SUMMARY OF ANALYTES EVALUATED AT EACH STATION IN WARD COVE AND MOSER BAY IN 1996

Station	NPDES Number[a]	Conventional Analytes: Grain Size	Total Solids	TOC	Total Ammonia	Total Sulfide	AVS	BOD	COD	Metals: Arsenic	Cadmium	Methyl-mercury	Total Mercury	Zinc	Organic Compounds: Dioxins/ Furans	Phenol	4-Methyl-phenol	Benzoic Acid	PAH Compounds	EOX	Pulp Mill Compounds[b]
Ward Cove																					
1		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A
2	43	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
3	40	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	A
4	39	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
5	41	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	A
6		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A
7		X	X	X	X	X	--	X	X	--	X	--	X	X	X	X	X	--	--	--	X
8		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A
9		X	X	X	X	X	--	X	X	--	X	--	X	X	X	X	X	--	--	--	X
10		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A
11	48	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	A
12		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A
13	46	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	A
14		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A
15		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A
16	44	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
17		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A
18	42	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	A
19		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A
20		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A
21		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A
22	51	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	A
23	49	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	A
24		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A
25	47	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	A
26		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A
27	45	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	A
28		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A
Moser Bay																					
29		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A
30		X	X	X	X	X	--	X	X	--	X	--	X	X	A	X	X	--	--	--	A

Note:
- A - sediment sample archived for possible future analysis
- AVS - acid-volatile sulfide
- BOD - biochemical oxygen demand
- COD - chemical oxygen demand
- EOX - extractable organic halides
- NPDES - National Pollutant Discharge Elimination System
- PAH - polycyclic aromatic hydrocarbon
- TOC - total organic carbon
- X - analyte measured
- -- - analyte not measured

[a] Corresponding station identified in KPC's NPDES permit.

[b] Includes chlorinated phenols, resin and fatty acids, and guaiacols.

- 96-hour echinoderm embryo test using the sand dollar *Dendraster excentricus*
- 20-day juvenile polychaete test using *Neanthes* sp.

These tests were selected because they represent a range of test species, exposure conditions, and endpoints. In addition, three of the tests (those based on *Rhepoxynius abronius*, *Dendraster excentricus*, and *Neanthes* sp.) are currently used in a regulatory context to manage contaminated sediments in the state of Washington by the Washington State Department of Ecology and the Puget Sound Dredged Disposal Analysis (PSDDA) program. As a result, considerable experience exists as to how these tests perform under various conditions and how to interpret the test results. Although the echinoderm embryo test has been used in a regulatory context in Washington State, quality assurance problems have been noted for this test (e.g., see Appendix C of PSDDA 1996).

In addition, the high variability among replicate samples often encountered with this test has required that statistical comparisons using this test be conducted at a significance level of $P \leq 0.10$, whereas the results of other toxicity tests used for regulatory purposes in Washington State are evaluated at $P \leq 0.05$ (Michelsen 1996). The Washington State Department of Ecology convened a special workshop in 1998 where a panel of scientists evaluated the appropriateness of the combined mortality/abnormality endpoint of the test and discussed possible modification. The limitations of the echinoderm test should therefore be considered when interpreting the results of this test in the present study.

Although evaluations of benthic macroinvertebrate assemblages are sometimes used as measures of chronic effects when determining compliance with the Washington State sediment management standards (Ecology 1995), that kind of indicator was not used in the present study. Instead, the 20-day juvenile polychaete test based on *Neanthes* sp. was used as the indicator of chronic effects, which is fully consistent with specifications of the Washington State sediment management standards.

Although the amphipod test based on *Leptocheirus plumulosus* is not currently identified as an indicator that can be used to determine compliance with the Washington State sediment management standards, that test was added to the suite of toxicity tests used to evaluate Ward Cove sediments because the test species appears to be tolerant to a wider range of physical sediment characteristics than *R. abronius* (Swartz 1996, pers. comm.). Given the unusual physical character of sediments in parts of Ward Cove, it was decided that the test based on *L. plumulosus* should be added to the test suite to attempt to factor out potential effects of the physical characteristics of test sediments on the toxicity results (PTI 1996). Although *L. plumulosus* is an estuarine amphipod from the east coast of the United States, it has been found to be as sensitive to chemical toxicity as two of the test species (*Ampelisca abdita* and *Eohaustorius estuarius*) used to evaluate compliance with the Washington State sediment management standards (Schlekat et al. 1995). In addition, the amphipod test based on *L. plumulosus* has been recommended for consideration for use in the sediment regulatory programs in Washington State (Ecology et al. 1995). This test, therefore, was considered acceptable for use in assessing the toxicity of Ward Cove sediments.

The information collected on sediment chemistry and sediment toxicity in Moser Bay during the Phase 1 investigation was used to place the chemical concentrations found in Ward Cove in the context of concentrations typically found in unaffected embayments of southeastern Alaska (Section 4.2) and to evaluate the statistical significance of the toxicity results found in Ward Cove. Section 7.1.2 presents a detailed description of how the data from Moser Bay were used in the statistical comparisons.

2.3.1.2 Modifications to the Work Plan

The following modifications were made to the Phase 1 sediment sampling strategy described in the work plan:

- Divers were used at six stations in Ward Cove (Stations 10, 12, 13, 14, 17, and 21) to guide the van Veen sampler through the logs and debris to the sediment bottom.
- Because large amounts of logs, wood debris, and cables were present in the northeast sector of Ward Cove, Station 10 was moved slightly to the northeast and Station 17 was moved slightly to the south to allow sample collection.

Laboratory personnel made substitutions for several methods specified in the work plan to accommodate their standard analytical procedures, as follows:

- EPA Method 7471 was used for the analysis of total mercury rather than the Contract Laboratory Program (CLP) method (EPA Method 245.5 CLP-M)
- EPA Method 350.1 was used for the analysis of total ammonia rather than EPA Method 350.3 and was modified to include sediment extraction with potassium chloride
- American Society for Testing and Materials Method D4129-82M was used for the analysis of TOC rather than Standard Method 5310B
- EPA Method 410.1 was used for the analysis of COD rather than EPA Method 410.2.

Because the substituted methods are similar to the methods specified in the work plan, the quality and usability of the data were not affected by any of the substitutions.

Toxicity tests were completed as described in the work plan (PTI 1996) without modification.

2.3.1.3 Phase 1 Field Methods

Table 2-2 provides a summary of the general characteristics of each station sampled in Ward Cove and Moser Bay. Sediments were sampled for chemical analysis and toxicity testing according to the field methods described in the work plan (PTI 1996, Appendix A). Sediment samples were collected using a stainless-steel, 0.06-m^2 modified van Veen bottom grab sampler from a boat equipped with a winch, davit, and pulley assembly. Station locations were established on the basis of station location coordinates from historical NPDES monitoring and the specifications provided in the work plan.

All sampling equipment was constructed of stainless steel and was decontaminated prior to sampling according to the procedures described in the work plan (PTI 1996, Appendix A). Although the target sediment horizon was 0–10 cm, samples were collected from shallower horizons at three stations after repeated sampling attempts were unsuccessful at obtaining samples from the 0–10 cm horizon. Surficial sediment samples were collected at each station and composited for chemical and toxicity testing. Based on EPA sediment sampling guidance (U.S. EPA 1991c), unrepresentative material was removed from the surficial sediment sample prior to sample compositing. Only materials (i.e., wood debris) that were large enough to be removed without contaminating the sample were removed in the field. Sediment samples were homogenized in a large stainless-steel bowl, and aliquots were collected from the homogenized samples for the individual analyses and toxicity tests. The samples were placed into appropriate chemically cleaned containers and held at 4°C during shipment and prior to testing. An additional aliquot of each sample was collected for potential future analysis. These archive samples were placed into frozen storage (−20°C) upon arrival at the laboratory.

2.3.1.4 Phase 1 Laboratory Methods

The sediment analyses were completed by three laboratories. Analyses for PCDDs/Fs were completed by Zenon Environmental Laboratories (Burlington, Ontario, Canada); analyses for methylmercury were completed by Frontier Geosciences (Seattle, Washington); and the remaining analyses were completed by Columbia Analytical Services, Inc. (Kelso, Washington). Puget Sound Estuary Program (PSEP) and EPA methods were used to complete the analyses whenever possible. A summary of analytical methods is provided in Table B1-2 in Appendix B1.

The compounds 3-methylphenol and 4-methylphenol commonly coelute from the chromatographic column under conditions prescribed by EPA Method 8270 (modified to include selected ion monitoring for optimal detection limits) and cannot be differentiated. During the 1995 sediment monitoring study, ENSR (1995b) analyzed 3- and 4-methylphenol separately in Ward Cove sediments and found that only 4-methylphenol is present at detectable concentrations. 3-Methylphenol was not detected in any sample. Consequently, separate analyses of 3- and 4-methylphenol were not conducted for the present study, and the laboratory reported results only for the sum of 3- and 4-methylphenol. Because 3-methylphenol was shown to be absent from the site, these

TABLE 2-2. STATION LOCATIONS, WATER DEPTHS, AND GENERAL SAMPLE CHARACTERISTICS FOR SEDIMENTS SAMPLED IN WARD COVE AND MOSER BAY IN 1996

Station	Location		Water Depth[a] (m)	Sample Depth (cm)	Sediment Characteristics[b]
	Easting	Northing			
Ward Cove					
1	3088118.99	1309529.74	26	10	Brownish, black color; very soft sediment with flocculent surface layer; minimal wood debris biological organisms (red worms); normal odor
2	3088356.16	1309823.81	16.5	10	Brown, black color; lots of wood debris (0–2 cm sediment, 3–10 cm mostly wood debris); slight sulfide
3	3088903.40	1310125.87	12	8–9	Black color; sheen on sediment surface; wood debris; shell debris; large rocks; sulfide odor
4	3089229.42	1310314.46	15.5	10	Black color; soft sediment; minimal wood debris; biological organisms (mussels); sulfide odor
5	3089517.53	1310659.94	7	8–9	Black color; lots of wood debris; sulfide odor
6	3088095.02	1309057.47	30.5	10	Dark brown, black color; soft sediment; sheen on surface; biological organisms (worms); moderate sulfide odor
7	3088547.27	1309391.16	28	10	Black color; soft sediment; minimal wood debris; fibers in overlying water; slight sulfide odor
8	3088913.69	1309653.57	30.5	10	Black color; soft sediment; minimal wood debris; fibers in overlying water; slight sulfide odor
9	3089219.71	1309908.03	18	10	Black, olive green color; wood debris and large piece of red bark; slight sulfide odor
10	3089940.72	1310317.03	13.5	9–10	Black color; wood debris; moderate sulfide odor
11	3085947.06	1307275.93	13	10	Dark brownish black; upper 2 cm of sediment mostly wood and shale; some shell debris; sulfide odor
12	3087182.08	1307837.25	20	10	Olive green color; soft sediment; wood debris; vegetative debris; shell debris; biological organisms (mussels and red worms); strong sulfide odor
13	3088170.84	1308361.30	40	8–10	Dark gray color; soft sediment; fibers in overlying water; wood debris; shell debris; biological organisms (red worms); sulfide odor
14	3088257.96	1308613.45	39	10	Black color; very soft sediment; wood debris; biological organisms (red worms)

TABLE 2-2. (cont.)

Station	Location		Water Depth[a] (m)	Sample Depth (cm)	Sediment Characteristics[b]
	Easting	Northing			
15	3088931.28	1309048.54	33.5	10	Dark gray color; wood debris (branches and bark); biological organisms (red worms); sulfide odor
16	3089340.48	1309431.21	15	10	Very black color; soft sediment; wood debris; biological organism (mussel); strong sulfide odor
17	3089840.31	1309500.83	11	10	Black color; lots of wood debris; strong odor (not sulfide)
18	3090361.67	1310249.61	4	6–8	Black gray sand; wood debris (chips only); shell debris; no noticeable odor
19	3087624.21	1307396.74	51	10	Black olive color; relatively firm sediment; wood debris (chips and a stick); sheen on sediment surface; moderate sulfide odor
20	3088674.36	1308149.38	44	10	Black olive color; minimal wood debris; very soft sediment; sulfide odor
21	3089426.75	1308762.52	34	10	Black olive color; wood debris (bark only); very soft sediment; sulfide odor
22	3086842.54	1305179.36	38	10	Dark brown color; wood debris (chips only) and slate chips; no noticeable odor
23	3087468.69	1305982.81	46	10	Brownish/blackish/green color; vegetative debris; wood debris (chips and twigs); biological organisms (crab, shrimp, clams, and worms); shell debris
24	3088267.26	1306829.19	36.5	10	Dark black, olive green color; black rivulets and spots of sheen on sediment surface; very soft sediment; shell debris; very strong sulfide odor
25	3088914.51	1307482.68	30	10	Dark olive, black color; some wood debris (bark and twigs); shell debris; very soft sediment; biological organisms (red worms); very slight sulfide odor
26	3089746.86	1308071.67	22.5	10	Dark olive green color; wood debris; strong sulfide odor
27	3090039.02	1308897.55	30.5	10	Black color; soft sediment in upper 4 cm, firmer sediment at 5–10 cm; wood debris (chips only); very strong sulfide odor
28	3090395.51	1309543.67	10.5	10	Dark greenish black color; very flocculent upper 4 cm, firmer sediment at 5–10 cm; wood debris; strong to moderate sulfide odor

TABLE 2-2. (cont.)

Station	Location		Water Depth[a] (m)	Sample Depth (cm)	Sediment Characteristics[b]
	Easting	Northing			
Moser Bay					
29	3102823.82	1360044.87	13	10	Brown black color with green surface layer (0–2 cm); firmer sediment; minimal wood debris; shell debris
30	3102253.34	1362163.39	62	10	Olive green color with a few black streaks; minimal shell debris

[a] Depths are presented to the nearest 0.5 m.

[b] wood debris - small wood chips and bark (unless otherwise noted)
shell debris - small shell fragments
vegetative debris - plant roots and leaves

results for the combined methylphenols were treated as concentrations of 4-methylphenol exclusively for all aspects of this study.

In addition to chemical analysis of the sediment, four toxicity tests were also performed. The laboratory methods used for the amphipod test based on *Rhepoxynius abronius*, the echinoderm embryo test based on *Dendraster excentricus* (sand dollar), and the juvenile polychaete test based on *Neanthes* sp. were the methods recommended by PSEP (1995). The laboratory methods used for the amphipod test based on *Leptocheirus plumulosus* were the methods recommended by ASTM (1992). Five replicate subsamples of each sediment sample collected in the field were tested in the laboratory. The following major laboratory specifications were used:

- A maximum sediment holding time of 14 days after field collection
- A photoperiod of 16 hours light and 8 hours dark
- Exposure periods as specified in the respective test protocol
- Water quality parameters (e.g., temperature, dissolved oxygen) as specified in the respective test protocol
- Aeration during testing
- Positive controls using cadmium chloride as the reference toxicant
- Negative controls using clean sediment.

The toxicity tests were completed by Northwestern Aquatic Sciences (Newport, Oregon).

2.3.1.5 Phase 1 Data Quality

The following sections describe the results of the quality assurance review of the Phase 1 data for chemical analyses and toxicity tests.

Phase 1 Chemical Analyses—A complete quality assurance report is provided in Appendix B1. Some of the results (Tables A1-1 through A1-5 in Appendix A1) were qualified as estimated (*J*) during the quality assurance review. As noted in U.S. EPA (1989d), "The *J*-qualifier is placed on CLP data to provide important information about an analysis to the data user or decision-maker, not to indicate low confidence in the analysis." Also noted in U.S. EPA (1989d), "The *J*-qualifier is a quantitative qualifier and can mean one or more of several things: 1) the target analyte is definitely present, 2) the sample was difficult to analyze, 3) the value may lie near the low end of the linear range of the instrument, and 4) the value should nearly always be seriously considered in decision-making."

Conventional Analytes: The laboratory reported a total of 466 results for conventional analytes. Samples from the 12 NPDES stations were analyzed for extractable organic halides, which were not detected in these samples. All other conventional analytes were present at concentrations above the detection limits in all samples. Results are provided in Table A1-1, Appendix A1.

No results for conventional analytes were qualified as estimated during the quality assurance review.

Metals: The laboratory reported a total of 125 results for metals. Total mercury was undetected in 12 samples. Arsenic, cadmium, methylmercury, and zinc were detected in all samples. Results are provided in Table A1-2, Appendix A1.

No results for metals were qualified as estimated during the quality assurance review.

Semivolatile Organic Compounds: The laboratory reported a total of 288 results for PAHs, phenol, 4-methylphenol, and benzoic acid. Of these results, 214 were reported at a concentration above the method detection limit, and 74 were reported as undetected. The detection limit for many of the samples was elevated because matrix interference necessitated sample dilution for analysis. Results are provided in Table A1-3, Appendix A1. Consistent with the approach recommended by EPA, concentrations of carcinogenic PAHs (i.e., benzo[a]pyrene, benz[a]anthracene, benzo[b]-fluoranthene, chrysene, benzo[k]fluoranthene, indeno[1,2,3-cd]pyrene, and dibenz[a,h]-anthracene) were calculated as the relative potency concentration (RPC) by adjusting their concentrations to reflect their carcinogenic potency relative to that of benzo[a]-pyrene. In calculating RPCs, undetected carcinogenic PAHs were included in calculations using one-half the detection limit.

During the quality assurance review, 184 results were qualified as estimated (*J*). Many of these results were qualified because the analyte was detected, but the analyte concentration was below the method quantification limit (the concentration equivalent to the lowest calibration standard) and could not be quantified reliably. Data were additionally qualified because quality control criteria were not met for one or more of the following procedures: surrogate recovery, analyte recovery from laboratory control samples, matrix spike recovery, or internal standards.

PCDDs/Fs: The laboratory reported a total of 255 results for PCDD/F congeners and 150 results for total homologs (total congeners at each chlorination level). For individual congeners, 153 results were above the detection limit and 102 were reported as undetected. Concentrations of total homologs for PCDDs/Fs were above the detection limit for all but 14 results. Results are provided in Table A1-4, Appendix A1. To be consistent with methods used by EPA in evaluating PCDD/F, where possible, PCDD/F concentrations were provided as TECs, wherein concentrations of PCDD/F

congeners that EPA considers to be carcinogenic (i.e., congeners substituted with chlorine at the 2, 3, 7, and 8 positions) were adjusted to reflect the assumed carcinogenic potency relative to that of 2,3,7,8-TCDD (U.S. EPA 1989c). In calculating TECs, one-half the detection limit was used for undetected relevant congeners.

No results were qualified as estimated during the quality assurance review.

Pulp Mill Compounds: The laboratory reported a total of 228 results for chlorinated phenols and related compounds and resin and fatty acids. Of these results, 196 were reported as undetected. The only detected pulp mill compounds were seven of the resin and fatty acids. Results are provided in Table A1-5, Appendix A1.

During the quality assurance review, 15 results for detected analytes and 50 results for undetected analytes (i.e., the reported detection limits) were qualified as estimated (*J*) because quality control criteria were not met for one or more of the following procedures: continuing calibration, surrogate recovery, analyte recovery from laboratory control samples, matrix spike recovery, or internal standards. Qualified compounds included the chlorinated catechols and the resin and fatty acids. Quality control results indicate that the chlorinated catechols were generally extracted with an efficiency of 50 percent or less. No catechols were detected in any sample; however, the reported detection limits are likely to exhibit a negative bias. No consistent bias could be determined for qualified results for the resin and fatty acids.

No results were qualified as estimated during the quality assurance review.

Phase 1 Toxicity Tests—A quality assurance review of the results of the four sediment toxicity test evaluations was performed. The results for the sediment toxicity study are summarized in Section 7. Details of the quality assurance review are provided in Appendix B3. The toxicity results and water quality data for each replicate sample are presented in Appendices A2 and A3, respectively. A summary of data quality is provided below.

Amphipod Toxicity Test Based on *Rhepoxynius abronius*: The recommended protocols were followed closely during testing. Water quality parameters were measured in the overlying water in all test replicates. There were no deviations from the specified salinity range of 28 ± 1 ppt during the test. The specified temperature range of 15 ± 1°C (i.e., 14–16°C) was exceeded by a small amount on one day of testing (exceedance of 16.5°C). There were no other deviations from the specified temperature. Concentrations of dissolved oxygen were greater than the recommended minimum level of 5.0 mg/L for all control and test sediment replicates. Values of pH ranged from 7.4 to 8.5 and were all within the recommended range of 7.0–9.0. The concentration of total ammonia ranged from less than 0.1 to 8.0 mg/L, and the concentration of total sulfide was less than 0.01 mg/L for all measurements.

The negative control consisted of sediment from West Beach, Washington. The mean survival value for the negative control sediment was 100 percent, which exceeds the performance criterion of 90 percent (Ecology 1995). The mean survival values for sediments from the two reference area samples were 91 and 93 percent, which exceed the performance criterion of 75 percent (Ecology 1995).

Because the amphipod test was conducted using appropriate protocols, water quality variables were generally within acceptable ranges, and performance criteria were achieved for the negative control and reference area samples, the results are considered acceptable for use in evaluating the toxicity of Ward Cove sediments.

Amphipod Toxicity Test Based on *Leptocheirus plumulosus*:
The recommended protocols were followed closely during testing. Water quality parameters were measured in the overlying water in all test replicates. There were no deviations from the specified salinity range of 28 ± 1 ppt during the test. The specified temperature range of 20 ± 1°C (i.e., 19–21°C) was maintained throughout the exposure period. Concentrations of dissolved oxygen were greater than the recommended minimum level of 5.0 mg/L for all control and test sediment replicates. Values of pH ranged from 7.5 to 8.6 and were all within the recommended range of 7.0–9.0. The concentration of total ammonia ranged from less than 0.1 to 10.5 mg/L, and the concentration of total sulfide was less than 0.01 mg/L for all measurements.

The negative control consisted of sediment from York River Marsh (culture media from the amphipod supplier). The mean survival value for the negative control sediment was 100 percent, which exceeds the performance criterion of 90 percent (Ecology 1995). The mean survival values for sediments from the two reference area samples were 97 and 99 percent, which exceed the performance criterion of 75 percent.

Because the amphipod test was conducted using appropriate protocols, water quality variables were within acceptable ranges, and performance criteria were achieved for the negative control and reference area samples, the results are considered acceptable for use in evaluating the toxicity of Ward Cove sediments.

Echinoderm Embryo Test Based on *Dendraster excentricus*:
The recommended protocols were closely followed during testing. Water quality parameters were measured daily in a designated water quality beaker. There were no deviations from the specified salinity range of 31 ± 1 ppt during the test. Temperatures measured during the testing period deviated slightly (minimum temperature of 13.5°C) from the specified temperature range of 15 ± 1°C (i.e., 14–16°C). Concentrations of dissolved oxygen were greater than the recommended minimum level of 5.0 mg/L. Values of pH ranged from 7.5 to 7.9 and were all within the recommended range of 7.0–9.0. The concentration of total ammonia ranged from less than 0.1 to 0.7 mg/L, and the concentration of total sulfide was less than 0.01 mg/L for all measurements.

The negative control consisted of seawater from Yaquina Bay, Oregon. Normal larvae were produced by 91 percent of the embryos in the negative seawater control, which exceeds the performance criterion of 70 percent (Ecology 1995). Normal survival values were 83 and 86 percent in the two reference area samples, which exceed the performance criterion of 65 percent (Michelsen 1996).

Because the echinoderm embryo test was conducted using appropriate protocols, water quality variables were generally within acceptable ranges, and performance criteria were achieved for the negative control and reference area samples, the results are considered acceptable for use in evaluating the toxicity of Ward Cove sediments.

Polychaete Toxicity Test Based on *Neanthes* sp.: The recommended protocols were followed closely during testing, and few methodological departures were made. There were no deviations from the specified salinity range of 28 ± 2 ppt during the test. There were no deviations from the specified temperature range of 20 ± 1°C (i.e., 19–21°C) during the test. Concentrations of dissolved oxygen were generally greater than the recommended minimum level of 5.0 mg/L. However, the dissolved oxygen concentration was less than 5.0 mg/L in three test replicates. The lowest dissolved oxygen concentration was 4.1 mg/L on Day 3. In addition, there were some procedural errors (i.e., air line misplaced, water renewal) in a single test replicate on one day of testing. This replicate was removed from further testing, and water quality monitoring was reassigned to another replicate for the same sample.

Values of pH ranged from 7.3 to 8.4 and were all within the recommended range of 7.0–9.0. The concentration of total ammonia ranged from less than 0.2 to 12.5 mg/L, and the concentration of total sulfide was less than 0.01 mg/L.

The negative control consisted of sediment from West Beach, Washington. The mean survival value for the control sediment was 80 percent, which is less than the minimum survival of 90 percent specified by PSDDA. PSEP does not specify a minimum control survival. However, mean individual growth rate for the negative control sediment met the performance criterion of a minimum growth rate of 0.38 mg/day proposed by Kendall (1996). The survival values for the reference area samples were 80 and 100 percent, and growth rates for the reference area samples (0.48 and 0.72 mg/day) were greater than the performance criterion of 0.40 mg/day (Ecology 1995).

Although the observed survival value of 80 percent in the negative control was less than the target level of 90 percent, results of the juvenile polychaete test were considered acceptable for use in assessing sediment toxicity in Ward Cove. Survival was high in most samples from Ward Cove and in both samples from the reference area (Moser Bay). Moreover, growth rates (the primary test endpoint) in the negative control and reference area were within the expected range relative to the initial size of the test organisms.

The primary use of the negative control in toxicity testing is to ensure that the test organisms are not unusually sensitive to testing as the result of handling stress, inadequate holding conditions, or other factors unrelated to sediment toxicity. If the test organisms are unusually sensitive, as demonstrated by reduced survival in the negative control, reduced survival in the test sediments could be wholly or partially an artifact of the unusual sensitivity of the test organisms. For the juvenile polychaete test used to test Ward Cove sediments, the high survival values observed in both the reference area and throughout most of Ward Cove indicate that the organisms were not unusually sensitive. For example, survival in Moser Bay (80–100 percent) achieved the target minimum level of 80 percent for reference areas (Table 2-3). For Ward Cove, survival at 27 of the 28 stations (97 percent) was greater than 80 percent, the target minimum acceptable level for a valid reference area (Table 2-3). These results indicate that the observed negative control survival of 80 percent was likely a random occurrence and did not indicate that the test organisms were unusually sensitive. The fact that the mean negative control value was based on only 25 individuals (i.e., 5 individuals per 5 replicate test chambers) also supports the likelihood that the observed control survival was a random event, because the difference between the observed level of 80 percent and the target minimum level of 90 percent was based on only three organisms.

The observed growth rates of the test organisms for the negative control and reference samples used in the juvenile polychaete test also indicate that the test results are acceptable. Because growth rate is the primary endpoint of this test, those results should have considerable influence on the acceptability of the test. The growth rates for both kinds of samples were within the expected range of growth rates based on the initial biomass (mean of 0.5 mg) of the test organisms (Figure 2-4). In addition, as previously stated, growth rate in the negative control (0.50 mg/day) was greater than the performance criterion of 0.38 mg/day, and growth rates for the reference area (0.48 and 0.72 mg/day) were greater than the performance criterion of 0.40 mg/day.

2.3.2 Phase 2

2.3.2.1 Phase 2 Study Design

The objectives of Phase 2 sampling activities were to provide a detailed characterization of the physical features of Ward Cove, refine the characterization of the horizontal extent of AOFs near the KPC facility, determine natural recovery rates for sediments, distinguish removal and/or capping areas from no-action areas, and characterize bulk chemical and engineering properties of sediments. The Phase 2 investigation included the following study elements:

- Bathymetric, geophysical, and hydrodynamic surveys
- Surface sediment characterization

TABLE 2-3. SUMMARY OF 1996 SURVIVAL RESULTS FOR THE SEDIMENT TOXICITY TEST BASED ON *Neanthes* sp.

Station	Survival (percent)
Ward Cove	
1	96
2	96
3	100
4	100
5	96
6	100
7	96
8	84
9	92
10	100
11	88
12	96
13	84
14	88
15	92
16	100
17	80
18	92
19	100
20	76
21	100
22	92
23	80
24	100
25	88
26	92
27	96
28	84
Moser Bay	
29	100
30	80

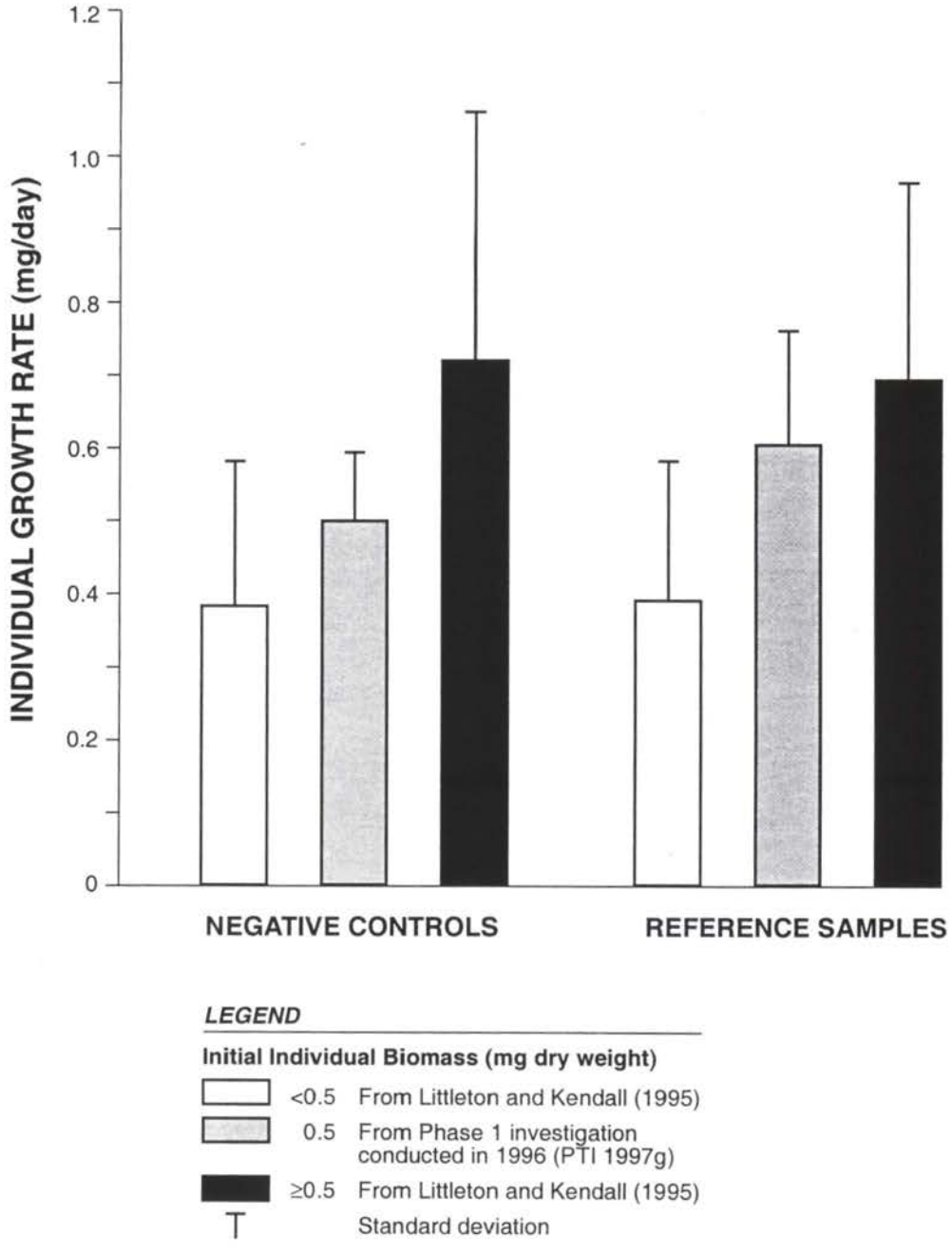

Figure 2-4. Growth rate of *Neanthes* sp. relative to initial biomass of test organisms.

- Sediment column characterization
- Sediment accumulation testing.

An overview of these study elements is provided in the following sections and in Figure 2-1. A more detailed description of the sampling and analytical methods is provided in the Phase 2 field sampling plan and quality assurance project plan (PTI 1997f).

Bathymetric, Geophysical, and Hydrodynamic Surveys—The bathymetric and geophysical surveys characterized the major physical features of Ward Cove and were conducted concurrently in May 1997. The surveys included simultaneous measurement of three kinds of information: precision bathymetry (i.e., depth distributions), physical characteristics of surface sediments (i.e., side-scan sonar), and characteristics of subsurface sediments (i.e., subbottom profiling and seismic reflections). In addition, video ground-truthing was performed during the geophysical survey to verify the results of the side-scan sonar and subbottom profiling data.

The bathymetric and geophysical surveys provided key information on physical features of Ward Cove prior to the more detailed Phase 2 sediment investigation. Information on water depth, shoreline configuration, and slope stability is also critical to the evaluation of sediment remedial alternatives. Finally, more detailed information on water depth and surface sediment characteristics provided an enhanced perspective on the kinds of benthic habitats found in Ward Cove.

The bathymetric survey was used to prepare a detailed map of water depth and bottom topography throughout Ward Cove. Side-scan sonar provided a detailed and continuous acoustic image of the bottom of Ward Cove. The images generated with this system were equivalent to an aerial photograph of the bottom of the Cove. Subbottom profiler and seismic reflection data provided information on the thickness of the sediment beneath the sediment surface.

A hydrodynamic survey was conducted in July and August 1997 to characterize the current patterns, tidal elevations, and salinity/temperature profiles within Ward Cove. As part of the hydrodynamic survey, current meters with salinity/temperature ports were placed at five locations in Ward Cove and at one location in Ward Creek (Figure 2-5). In addition, a digital tide gauge was placed on the northern shoreline of the Cove (Figure 2-5). The hydrodynamic data were used to assess the potential for sediment transport into Tongass Narrows, improve present knowledge of water circulation within the Cove, and support assessment of the potential for natural recovery of sediment.

Surface Sediment Characterization—As part of Phase 2, surface sediment (upper 10 cm) samples were collected from 33 stations in Ward Cove (Figure 2-2) and from 2 stations in Moser Bay, the selected reference area (Figure 2-3). The two stations in Moser Bay were sampled to estimate concentrations of CoPCs and toxicity levels in a

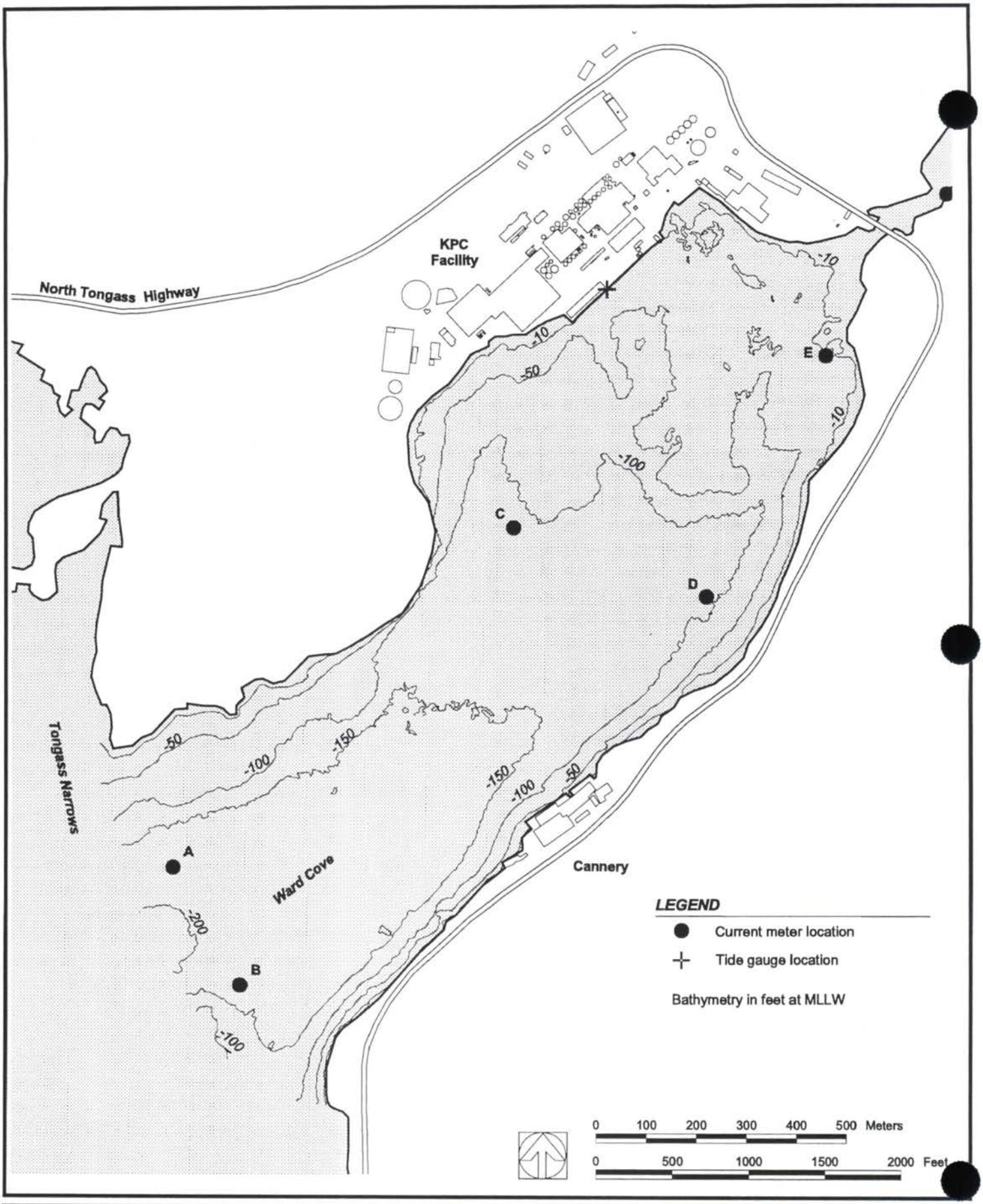

Figure 2-5. Locations of current meters placed in Ward Cove in 1997.

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representative reference area in southeastern Alaska. In addition, surface samples (upper 5 cm) were collected from two transects (5 stations per transect) at the mouth of Ward Creek (i.e., intertidal stations 50 and 51; Figure 2-2). Sampling was conducted in July and August 1997. Twelve of the 33 stations in Ward Cove were stations identified in the sediment monitoring component of the KPC NPDES permit (Figure 2-2). The surface sediment characterization refined the boundaries of the primary and secondary AOFs near the KPC facility identified during Phase 1 (described in Section 5 of PTI 1997g), filled gaps in the spatial coverage of stations within and adjacent to the AOFs, and evaluated potential cause and effect relationships between CoPCs and sediment toxicity.

The same suite of sediment toxicity tests (i.e., 10-day amphipod test using *Rhepoxynius abronius* and 96-hour echinoderm embryo test using *Dendraster excentricus*) was evaluated for all 33 Ward Cove stations and for the 2 Moser Bay stations (reference area), but the suites of chemical analytes differed among stations (Table 2-4). The following three major groups of analytes were measured:

- **Group 1 (Phase 1 CoPCs and associated conventional analytes):** Because intertidal sediment samples were not collected during Phase 1, surface sediment samples from two intertidal locations were analyzed for Phase 1 CoPCs.
- **Group 2 (Phase 2 CoPCs and associated conventional analytes):** Phase 2 analytes were limited to those Phase 1 CoPCs that were not screened out by the initial human health and ecological evaluations (PTI 1997g) and included TOC, total ammonia, total sulfide, BOD, COD, and 4-methylphenol, as well as additional conventional analytes (i.e., grain size and total solids) considered essential for interpreting results of the chemical analyses and toxicity tests.
- **Group 3 (additional NPDES analytes):** These analytes were measured at the 12 NPDES stations to satisfy the requirements of the KPC permit. They include the NPDES analytes that are not included in Group 1 (i.e., AVS, arsenic, methylmercury, benzoic acid, PAH compounds, and extractable organic halides). PCDDs/Fs were excluded from the 1997 NPDES sampling with the approval of EPA.

Specialized toxicity testing was also performed using surface sediment samples collected from eight stations during Phase 2. The specialized toxicity testing focused mainly on *R. abronius*. Specialized toxicity testing involved whole sediment and porewater manipulations (aeration and *Ulva* exposures) that wholly or partially removed ammonia or sulfide. The specialized toxicity testing was conducted to evaluate the role of ammonia and sulfide in causing sediment toxicity.

In addition, 12 surface sediment samples that were archived frozen since their date of collection in June 1996 were analyzed for PCDDs/Fs in August 1997. This information was used to fill gaps on the spatial extent of PCDDs/Fs in the Cove.

TABLE 2-4. SUMMARY OF ANALYTES EVALUATED AT EACH STATION IN WARD COVE AND MOSER BAY IN 1997

Station	Surface, Core Transect, or 1996 Archive	NPDES Number[a]	Conventional Analytes								Metals				
			Grain Size	Total Solids	TOC	Total Ammonia	Total Sulfide	AVS	BOD	COD	Arsenic	Cadmium	Methyl-mercury	Total Mercury	Zinc
Ward Cove-Subtidal															
1	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
	1996 Archive		--	--	--	--	--	--	--	--	--	--	--	--	--
2	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
	Surface	43	X	X	X	X	X	X	X	X	X	X	X	X	X
3	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
	Surface	40	X	X	X	X	X	X	X	X	X	X	X	X	X
4	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
	Surface	39	X	X	X	X	X	X	X	X	X	X	X	X	X
5	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
	Surface	41	X	X	X	X	X	X	X	X	X	X	X	X	X
6	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
	1996 Archive		--	--	--	--	--	--	--	--	--	--	--	--	--
7	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
8	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
9	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
11	Surface	48	X	X	X	X	X	X	X	X	X	X	X	X	X
12	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
	1996 Archive		--	--	--	--	--	--	--	--	--	--	--	--	--
13	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
	Surface	46	X	X	X	X	X	X	X	X	X	X	X	X	X
14	1996 Archive		--	--	--	--	--	--	--	--	--	--	--	--	--
15	1996 Archive		--	--	--	--	--	--	--	--	--	--	--	--	--
16	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
	Surface	44	X	X	X	X	X	X	X	X	X	X	X	X	X
17	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
	1996 Archive		--	--	--	--	--	--	--	--	--	--	--	--	--
18	Surface	42	X	X	X	X	X	X	X	X	X	X	X	X	X
19	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
	1996 Archive		--	--	--	--	--	--	--	--	--	--	--	--	--
20	1996 Archive		--	--	--	--	--	--	--	--	--	--	--	--	--
21	1996 Archive		--	--	--	--	--	--	--	--	--	--	--	--	--
22	Surface	51	X	X	X	X	X	X	X	X	X	X	X	X	X
23	Surface	49	X	X	X	X	X	X	X	X	X	X	X	X	X
24	1996 Archive		--	--	--	--	--	--	--	--	--	--	--	--	--
25	Surface	47	X	X	X	X	X	X	X	X	X	X	X	X	X
26	1996 Archive		--	--	--	--	--	--	--	--	--	--	--	--	--
27	Surface	45	X	X	X	X	X	X	X	X	X	X	X	X	X
28	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
31	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
32	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
33	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
34	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--

TABLE 2-4. (cont.)

Station	Surface, Core Transect, or 1996 Archive	NPDES Number[a]	Conventional Analytes								Metals				
			Grain Size	Total Solids	TOC	Total Ammonia	Total Sulfide	AVS	BOD	COD	Arsenic	Cadmium	Methyl-mercury	Total Mercury	Zinc
35	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
36	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
37	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
38	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
39	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
40	Core[f]		--	--	--	--	--	--	--	--	--	--	--	--	--
	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
41	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
42	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
43	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
44	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
45	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
47	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
48	Surface		X	X	X	X	X	--	X	X	--	--	--	--	--
49	Core		X	X	X	X	X	--	X	X	--	X	--	X	X
Moser Bay-Subtidal															
29	Surface		X	X	X	X	X	X	X	X	X	X	X	X	X
30	Surface		X	X	X	X	X	X	X	X	X	X	X	X	X
	1996 Archive		--	--	--	--	--	--	--	--	--	--	--	--	--
Ward Cove-Intertidal															
50	Transect		X	X	X	X	X	--	X	X	--	X	--	X	X
51	Transect		X	X	X	X	X	--	X	X	--	X	--	X	X

TABLE 2-4. (cont.)

Station	Surface, Core Transect, or 1996 Archive	NPDES Number[a]	Organic Compounds: Dioxins/ Furans[b]	Phenol	4-Methyl-phenol	Benzoic Acid	PAH Compounds	EOX	Lead-210 Cesium-137	Toxicity Tests: 10-Day Amphipod	96-Hour Echinoderm	Specialized Toxicity Tests: Sediment Analyses	Pore Water Analyses
Ward Cove-Subtidal													
1	Core		X[1]	X	X	--	--	--	--	--	--	--	--
	1996 Archive		X	--	--	--	--	--	--	--	--	--	--
2	Core		X[1]	X	X	--	--	--	--	--	--	--	--
	Surface	43	X	X	X	X	X	X	--	X	X	--	--
3	Core		X[2]	X	X	--	--	--	--	--	--	--	--
	Surface	40	X	X	X	X	X	X	--	X	X	--	--
4	Core		X[2]	X	X	--	--	--	--	--	--	--	--
	Surface	39	X	X	X	X	X	X	--	X	X	--	--
5	Core		X[2]	X	X	--	--	--	--	--	--	--	--
	Surface	41	X	X	X	X	X	X	--	X	X	--	--
6	Core		X[1]	X	X	--	--	--	--	--	--	--	--
	1996 Archive		X	--	--	--	--	--	--	--	--	--	--
7	Core		X[3]	X	X	--	--	--	--	--	--	--	--
	Surface		--	--	X	--	--	--	--	X	X	X	X
8	Core		X[3]	X	X	--	--	--	--	--	--	--	--
9	Core		X[3]	X	X	--	--	--	--	--	--	--	--
11	Surface	48	X	X	X	X	X	X	--	X	X	--	--
12	Core		X[4]	X	X	--	--	--	--	--	--	--	--
	Surface		--	--	X	--	--	--	--	X	X	X	X
	1996 Archive		X	--	--	--	--	--	--	--	--	--	--
13	Core		X[4]	X	X	--	--	--	--	--	--	--	--
	Surface	46	X	X	X	X	X	X	--	X	X	--	--
14	1996 Archive		X	--	--	--	--	--	--	--	--	--	--
15	1996 Archive		X	--	--	--	--	--	--	--	--	--	--
16	Core		X[5]	X	X	--	--	--	--	--	--	X	--
	Surface	44	X	X	X	X	X	X	--	X	X	X	X
17	Surface		--	--	X	--	--	--	--	X	X	X	X
	1996 Archive		X	--	--	--	--	--	--	--	--	--	--
18	Surface	42	X	X	X	X	X	X	--	X	X	--	--
19	Surface		--	--	X	--	--	--	--	X	X	--	--
	1996 Archive		X	--	--	--	--	--	--	--	--	--	--
20	1996 Archive		X	--	--	--	--	--	--	--	--	--	--
21	1996 Archive		X	--	--	--	--	--	--	--	--	--	--
22	Surface	51	X	X	X	X	X	X	--	X	X	--	--
23	Surface	49	X	X	X	X	X	X	--	X	X	--	--
24	1996 Archive		X	--	--	--	--	--	--	--	--	--	--
25	Surface	47	X	X	X	X	X	X	--	X	X	--	--
26	1996 Archive		X	--	--	--	--	--	--	--	--	--	--
27	Surface	45	X	X	X	X	X	X	--	X	X	--	--
28	Surface		--	--	X	--	--	--	--	X	X	--	--
31	Surface		--	--	X	--	--	--	--	X	X	--	--
32	Surface		--	--	X	--	--	--	--	X	X	--	--
33	Core		X[2]	X	X	--	--	--	--	--	--	--	--
	Surface		--	--	X	--	--	--	--	X	X	--	--
34	Surface		--	--	X	--	--	--	--	X	X	X	X

TABLE 2-4. (cont.)

Station	Surface, Core Transect, or 1996 Archive	NPDES Number[a]	Organic Compounds: Dioxins/ Furans[b]	Phenol	4-Methyl-phenol	Benzoic Acid	PAH Compounds	EOX	Lead-210 Cesium-137	Toxicity Tests: 10-Day Amphipod	96-Hour Echinoderm	Specialized Toxicity Tests: Sediment Analyses	Pore Water Analyses
35	Surface		--	--	X	--	--	--	--	X	X	X	X
36	Core		X[5]	X	X	--	--	--	--	--	--	--	--
37	Surface		--	--	X	--	--	--	--	X	X	--	--
38	Surface		--	--	X	--	--	--	--	X	X	--	--
39	Surface		--	--	X	--	--	--	--	X	X	--	--
40	Core[f]		--	--	--	--	--	--	X	--	--	--	--
	Surface		--	--	X	--	--	--	--	X	X	--	--
41	Core		X[5]	X	X	--	--	--	--	--	--	--	--
	Surface		--	--	X	--	--	--	--	X	X	--	--
42	Surface		--	--	X	--	--	--	--	X	X	--	--
43	Surface		--	--	X	--	--	--	--	X	X	--	--
44	Surface		--	--	X	--	--	--	--	X	X	X	X
45	Surface		--	--	X	--	--	--	--	X	X	--	--
47	Surface		--	--	X	--	--	--	--	X	X	--	--
48	Surface		--	--	X	--	--	--	--	X	X	--	--
49	Core		--	X	X	--	--	--	X	--	--	--	--
Moser Bay-Subtidal													
29	Surface		X	X	X	X	X	X	--	X	X	--	--
30	Surface		X	X	X	X	X	X	--	X	X	--	--
	1996 Archive		X	--	--	--	--	--	--	--	--	--	--
Ward Cove-Intertidal													
50	Transect		--	X	X	--	--	--	--	--	--	--	--
51	Transect		--	X	X	--	--	--	--	--	--	--	--

TABLE 2-4. (cont.)

Station	Surface, Core Transect, or 1996 Archive	NPDES Number[a]	Engineering Tests MET[b]	Engineering Tests DRET[b]	Engineering Tests Other[c]	Native Sediments[d]	Archive Sample[e]
Ward Cove-Subtidal							
1	Core		X[1]	X[1]	X[1]	A	--
	1996 Archive		--	--	--	--	--
2	Core		--	--	--	--	--
	Surface	43	--	--	--	--	A
3	Core		X[2]	X[2]	X[2]	A	--
	Surface	40	--	--	--	--	A
4	Core		--	--	--	A	--
	Surface	39	--	--	--	--	A
5	Core		X[2]	X[2]	X[2]	A	--
	Surface	41	--	--	--	--	A
6	Core		--	--	--	A	--
	1996 Archive		--	--	--	--	--
7	Core		X[1]	X[1]	X[1]	X	--
	Surface		--	--	--	--	A
8	Core		--	--	--	A	--
9	Core		--	--	--	A	--
11	Surface	48	--	--	--	--	A
12	Core		--	--	--	A	--
	Surface		--	--	--	--	A
	1996 Archive		--	--	--	--	--
13	Core		--	--	--	A	--
	Surface	46	--	--	--	--	A
14	1996 Archive		--	--	--	--	--
15	1996 Archive		--	--	--	--	--
16	Core		--	--	--	--	--
	Surface	44	--	--	--	--	A
17	Surface		--	--	--	--	A
	1996 Archive		--	--	--	--	--
18	Surface	42	--	--	--	--	A
19	Surface		--	--	--	--	A
	1996 Archive		--	--	--	--	--
20	1996 Archive		--	--	--	--	--
21	1996 Archive		--	--	--	--	--
22	Surface	51	--	--	--	--	A
23	Surface	49	--	--	--	--	A
24	1996 Archive		--	--	--	--	--
25	Surface	47	--	--	--	--	A
26	1996 Archive		--	--	--	--	--
27	Surface	45	--	--	--	--	A
28	Surface		--	--	--	--	A
31	Surface		--	--	--	--	A
32	Surface		--	--	--	--	A
33	Core		--	--	--	A	--
	Surface		--	--	--	--	A
34	Surface		--	--	--	--	A

TABLE 2-4. (cont.)

Station	Surface, Core Transect, or 1996 Archive	NPDES Number[a]	Engineering Tests MET[b]	DRET[b]	Other[c]	Native Sediments[d]	Archive Sample[e]
35	Surface		--	--	--	--	A
36	Core		--	--	--	A	-
37	Surface		--	--	--	--	A
38	Surface		--	--	--	--	A
39	Surface		--	--	--	--	A
40	Core[f]		--	--	--	--	--
	Surface		--	--	--	--	A
41	Core		--	--	--	X	--
	Surface		--	--	--	--	A
42	Surface		--	--	--	--	A
43	Surface		--	--	--	--	A
44	Surface		--	--	--	--	A
45	Surface		--	--	--	--	A
47	Surface		--	--	--	--	A
48	Surface		--	--	--	--	A
49	Core		--	--	--	X	--
Moser Bay-Subtidal							
29	Surface		--	--	--	--	A
30	Surface		--	--	--	--	A
	1996 Archive		--	--	--	--	--
Ward Cove-Intertidal							
50	Transect		--	--	--	--	A
51	Transect		--	--	--	--	A

Note: -- - analyte was not measured
X - analyte was measured
A - sample was archived for possible future analysis
AVS - acid-volatile sulfide
BOD - biochemical oxygen demand
COD - chemical oxygen demand
DRET - dredging elutriate test
EOX - extractable organic halides
KPC - Ketchikan Pulp Company
MET - modified elutirate test
NPDES - National Pollutant Discharge Elimination System
PAH - polycyclic aromatic hydrocarbon
TOC - total organic carbon

[a] Corresponding station identified in KPC's NPDES permit.

[b] Superscript numerals indicate which cores were composited into a single sample. Water for elutriate preparation was also collected at these stations.

[c] Other engineering properties tests are column settling and physical properties, including grain size, water content and void ratio, specific gravity, and Atterberg limits (liquid and plasticity limits).

[d] Native sediments are defined as sediments that existed in the Cove prior to the deposition of material potentially affected by KPC.

[e] If enough sediment was present at a specific sediment horizon in a sediment core, then an archive sample was collected.

[f] A core sample was collected at this station and analyzed for only lead-210, cesium-137, grain size, and total solids.

Sediment Column Characterization—As part of Phase 2, subsurface sediment samples were collected from 16 stations in Ward Cove (Figure 2-6). Sampling was conducted in August 1997. Characterization of the sediment column established the vertical extent of sediment contamination, wood debris, and other distinct sediment horizons and determined the bulk chemical and physical properties of sediments within the primary AOF near the KPC facility. In addition, during Phase 2, selected composite sediment samples were analyzed for PCDDs/Fs (Figure 2-7) and for engineering properties that affect remediation options.

Sediment Accumulation Testing—As part of Phase 2, two sediment cores were collected in Ward Cove (Figure 2-6). Sampling was conducted in August 1997. Sediment cores were collected from representative stations in Ward Cove to characterize sediment accumulation rates. These rates were then used to estimate the rate at which existing sediments will be buried by newly deposited clean sediments after shutdown of the KPC facility. This information is incorporated into natural recovery modeling to predict future sediment conditions in the absence of releases from the KPC facility (Section 9, *Natural Recovery*).

Lead-210 and cesium-137 were analyzed at multiple depths throughout the sediment cores to determine 1) the rate at which lead-210 decreases below the surface mixed layer, and 2) the depth horizon of the cesium-137 maximum. Sediments can be dated on the basis of lead-210 measurements by relating the time scale of lead-210 decay (22-year half-life) to the sediment depth over which a comparable decrease in lead-210 activity occurs (Carpenter et al. 1985). Sediments can be dated on the basis of the cesium-137 measurements because the worldwide subsurface maximum in cesium-137 can be related to the period of nuclear testing, with the peak corresponding to 1963 and the first appearance corresponding to 1955.

2.3.2.2 Modifications to the Field Sampling Plan

The following modifications were made to the sediment sampling strategy described in the field sampling plan (PTI 1997f):

- A video camera was used during surface and subsurface sediment sampling to guide the van Veen sampler and piston corer, respectively, through the logs and debris to the sediment bottom.
- Because the chemical testing laboratory did not receive samples from the overnight express shipper (i.e., Federal Express), Station 14 was resampled and the original sample, which was received by the toxicity testing laboratory, was discarded. Analyses were performed by all testing laboratories on the resampled sediment from this station.

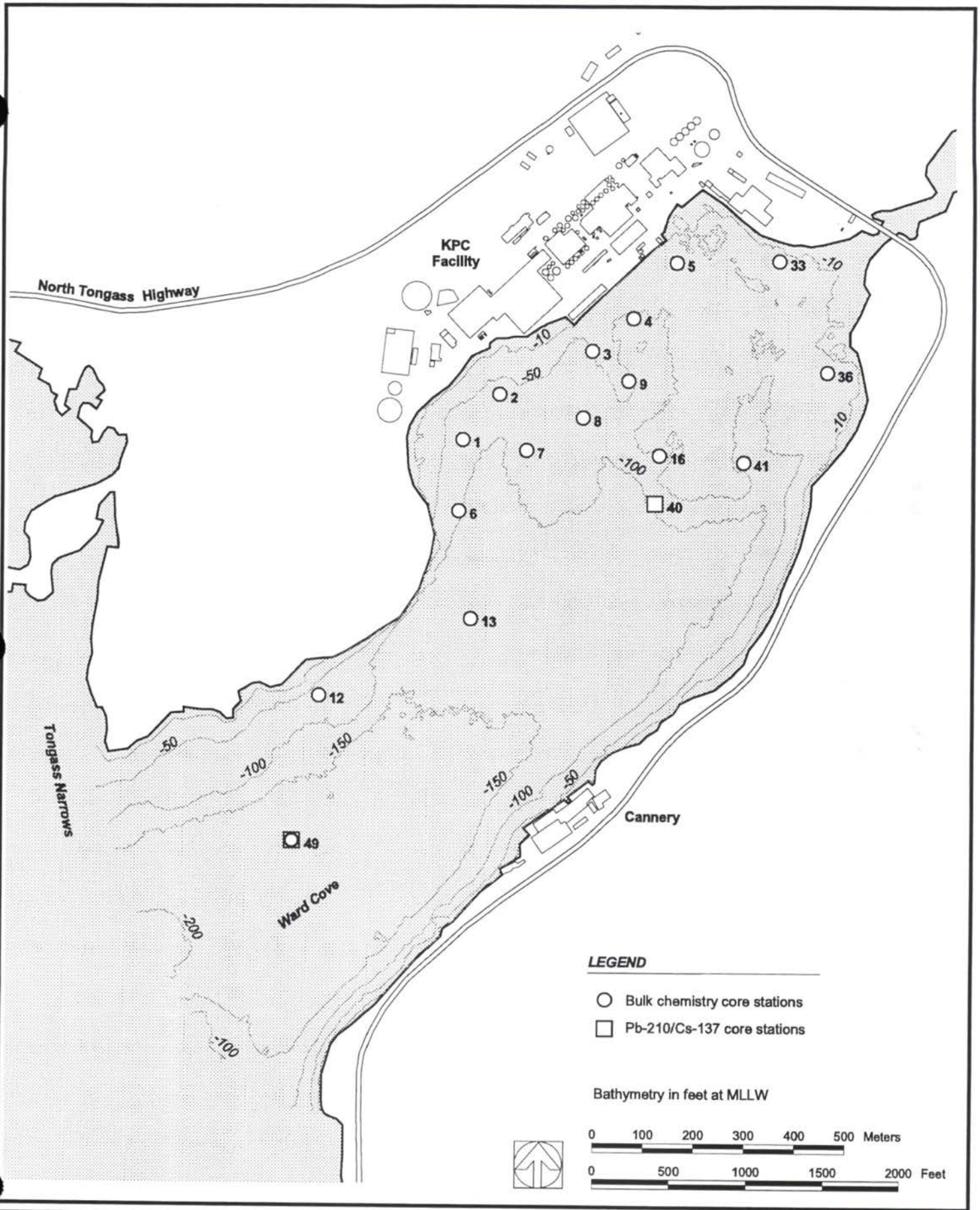

Figure 2-6. Station locations in Ward Cove at which sediment core samples were collected in 1997.

CB0W-16-01 06/15/98 GIS

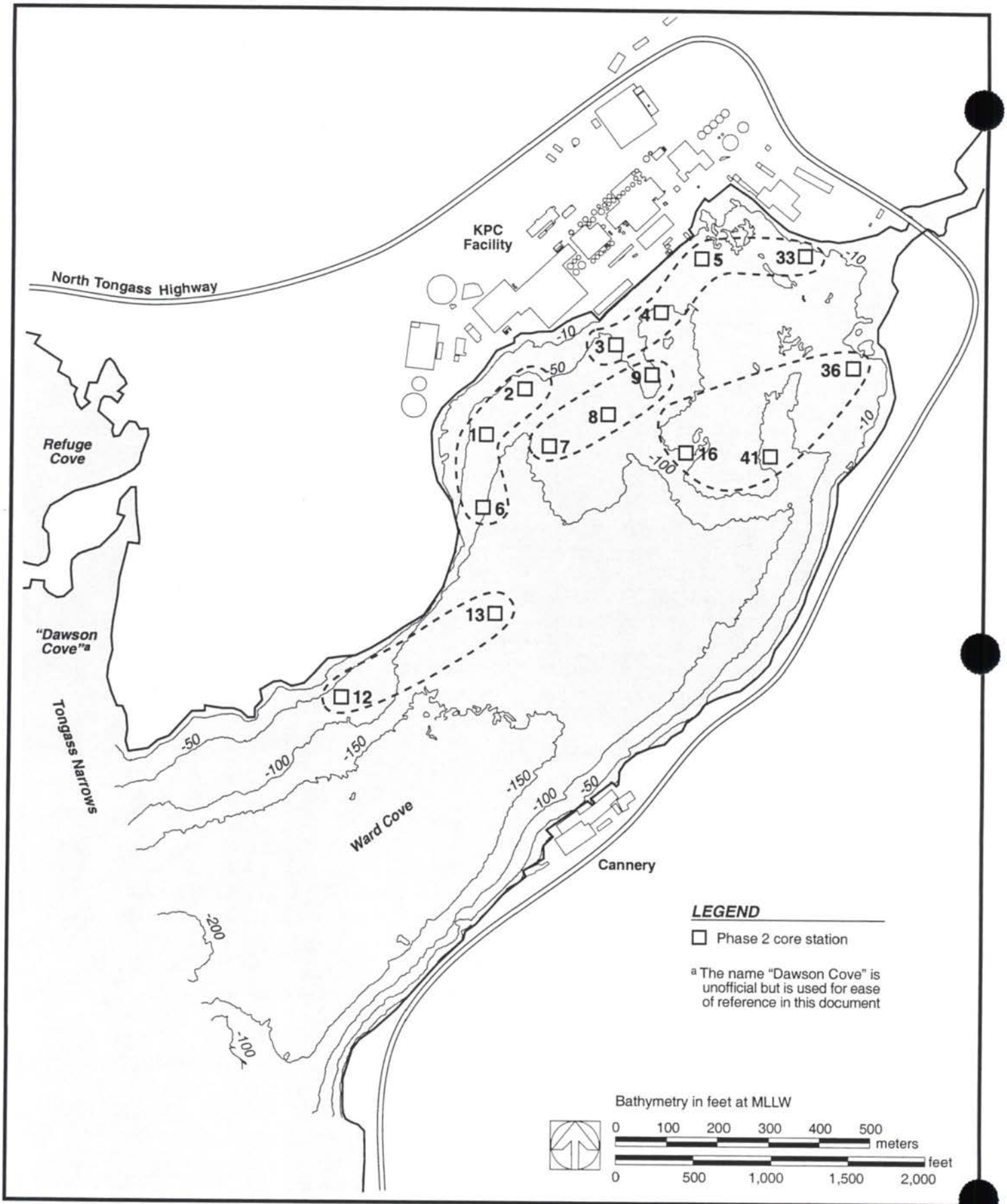

Figure 2-7. Station locations in Ward Cove at which sediment composites were collected for dioxin and furan analysis in 1997.

- No impacted sediments (i.e., an organic-rich surface horizon) were observed at Stations 46, 47, and 49. Therefore, no impacted sediment samples from these stations were submitted for analysis.
- Because of the minimal amounts of impacted sediments near the mouth and in the center of the Cove (i.e., Stations 46, 47, and 49), the stations specified in PTI (1997f) as locations for collection of composite samples for dioxin and furan analyses were changed (Figure 2-7).
- Although 6-ft cores were specified in PTI (1997f), in an attempt to reach native sediment, 10-ft cores were collected at Stations 1, 2, 4, 5, 6, 7, 8, 9, 12, 13, and 16. In addition, a 10-ft core tube was used at Station 36.
- Native sediments were not reached at Stations 1, 2, 6, 9, and 16 (note: native sediment was observed in the nose cone of the core at Station 16). Therefore, native sediments at these stations were not submitted for either analysis or archive.
- Three stations were specified for analysis of native sediments (Stations 2, 16, and 49). Native sediments were not reached at two of the three stations (Stations 2 and 16). Therefore, native sediments were collected at Stations 7 and 41 instead (Figure 2-8). In addition, review of core logs and chemistry results for the 31–59 in. horizon for Core S7 indicated that this horizon also represented native sediment.
- Additional surface sediments were collected at multiple stations for possible specialized toxicity tests if results from the targeted specialized toxicity testing stations proved to have no toxic responses (Becker 1997, pers. comm.).
- Surface sediment for specialized toxicity testing was collected from additional stations that were not originally intended for Phase 2 bulk sediment chemistry analyses and toxicity testing (i.e., Stations 7 and 12). The surface sediment at these stations was analyzed for bulk sediment chemistry, toxicity tests, and specialized toxicity tests.
- Because large amounts of logs, wood debris, and cables were present in the northeast sector of Ward Cove, Station 3 was moved slightly to the southeast to allow sample collection and Station 27 was moved slightly to the southwest. Station 23 was moved slightly to the west because a large immovable vessel was present.
- Photographs of sediment stratigraphy were not taken on Day 1 of the subsurface sediment investigation.
- Additional water samples were collected just above the sediment surface at Stations 2, 8, 16, and 41.

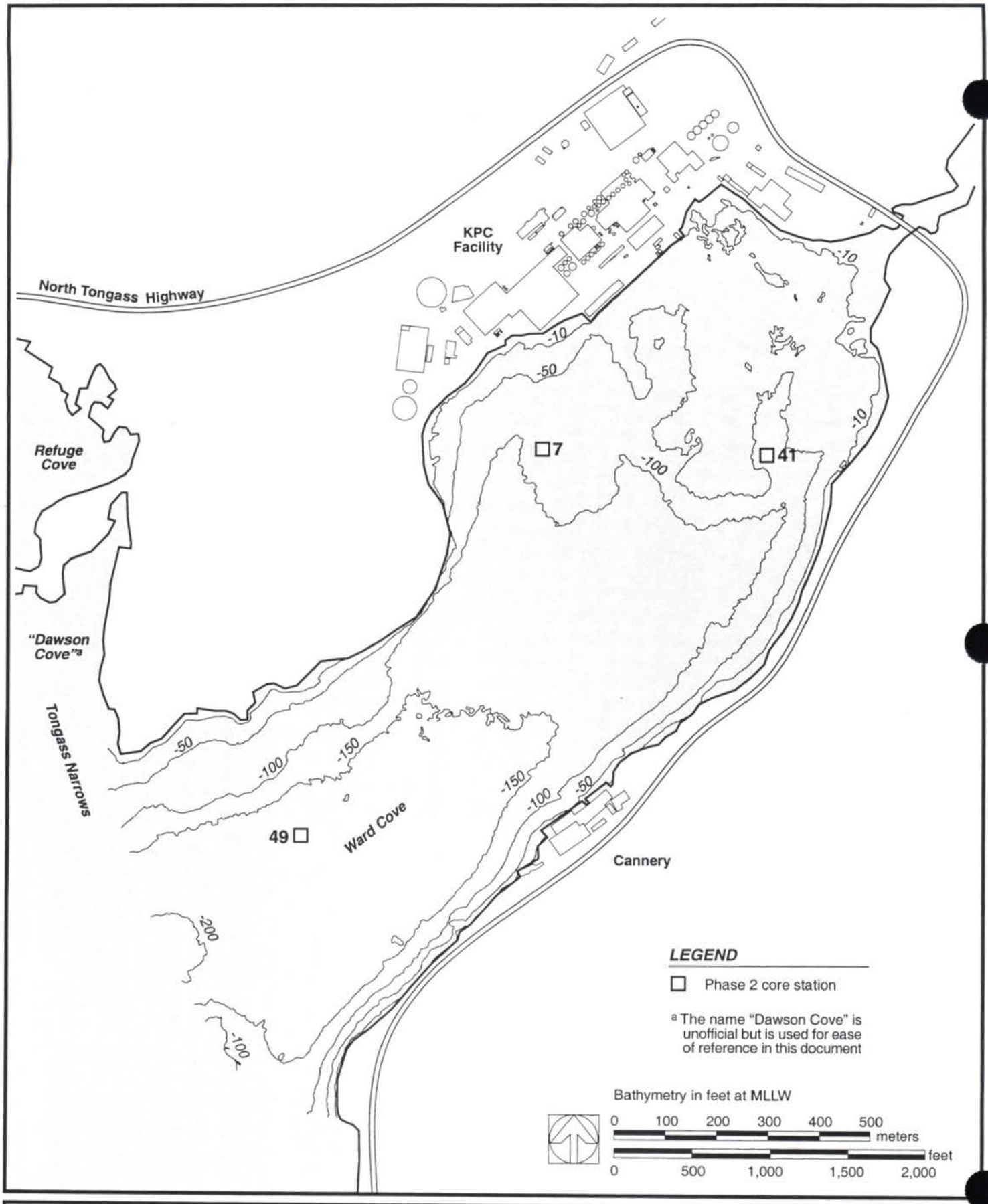

Figure 2-8. Station locations in Ward Cove at which native sediments were collected in 1997.

- Because of commercial activity at the cannery and the associated increased moorage requirements, access to the planned location for the tide gauge was denied. The tide gauge was moved to KPC property and attached to a piling on the northern shoreline of the Cove.
- During the hydrodynamic survey, a current meter, rather than a flow meter, was placed in Ward Creek.

Laboratory personnel made substitutions for several methods specified in the quality assurance plan (PTI 1997f, Appendix B) to accommodate their standard analytical procedures, as follows:

- PSEP method (PSEP 1986) was used for the analysis of TOC rather than Standard Method 5310B (APHA 1989)
- PSEP method (PSEP 1986) was used for the analysis of BOD rather than EPA Method 405.1M (U.S. EPA 1983)
- EPA Method 160.3 (U.S. EPA 1983) was used for the analysis of total solids rather than the PSEP method (PSEP 1986)
- EPA Method 8290 (U.S. EPA 1994g) for the analysis of PCDDs/Fs was modified to included some of the quality control criteria specified in EPA Method 1613B (U.S. EPA 1994c) and a greater number of isotopically labeled internal standards
- EPA Method 6010A (U.S. EPA 1992c) with inductively coupled plasma-atomic emission spectrometry (ICP-AES) was used for the analysis of metals in the elutriate samples rather than EPA Method 200.7 (U.S. EPA 1994f) using ICP-AES
- To achieve lower detection limits, EPA Method 200.8 (U.S. EPA 1994f) using inductively coupled plasma-mass spectrometry was used for the analysis of metals in the equipment rinsate blank samples rather than EPA Method 200.7 (U.S. EPA 1994f) using ICP-AES
- SW-846 Method 7470A (U.S. EPA 1994f) was used for the analysis of mercury in elutriate and equipment rinsate blank samples rather than EPA Method 245.1 (U.S. EPA 1994f)
- The procedure used by the toxicity testing laboratory for daily water quality monitoring of the amphipod test was modified from daily measurements in each replicate to daily measurements in all replicates at test initiation and test termination and in one test replicate per station daily.

Because the substituted methods are similar to the methods specified in the field sampling plan (PTI 1997f), the quality and usability of the data were not affected by any of the

substitutions. Consistent with the methods stated in the field sampling plan, organic material was not removed from sediment samples prior to the determination of grain size.

2.3.2.3 Phase 2 Field Methods

Table 2-5 provides a summary of the general characteristics of each surface sediment station sampled in Ward Cove, Ward Creek (i.e., intertidal stations), and Moser Bay. Table 2-6 provides a summary of the general characteristics of each subsurface sediment station sampled in Ward Cove. Sediments were sampled for chemical analysis and toxicity testing according to the field methods described in the field sampling plan (PTI 1997f, Appendix B). Station positioning for the bathymetric and geophysical surveys and for all sediment and water sampling was accomplished using a differential global positioning system (DGPS). Position data were used in real time to provide navigation information to the vessel operator. The planned station locations (or survey lines), and the actual station locations sampled (or survey lines traversed), were displayed in real time on a monitor, along with a left-right indicator to show the distance from the planned station location (or survey line). Station location coordinates are provided in Tables 2-2 and 2-6.

Bathymetric and Geophysical Surveys—The bathymetric and geophysical surveys were conducted concurrently. Positioning was accomplished with an accuracy of within 1 m for both surveys using a DGPS. Event marks were automatically triggered on a geophysical graphic recorder at 20-second intervals to correlate geophysical data with position data.

Bathymetric Survey: Bathymetric data were collected with a Reson Seabat 9001 multibeam sonar and an integrated suite of instruments for wide-area swath mapping of the seafloor. The Seabat 9001 multibeam system was used for this survey because of its narrow beam geometry (1.5 by 1.5 degrees) and swath coverage of 90 degrees (45 degrees to starboard and 45 degrees to port). These features result in 60 soundings over twice the water depth in a single pass, thereby increasing the resolution of the bathymetric survey. Sonar swaths were recorded at a rate of 8 per second as the survey vessel proceeded along the survey track line. Bathymetric data were collected by running transects parallel to the shoreline. The spacing between the adjacent transects was a function of water depth and varied depending on the swath width obtained on the previous transect. In general, the spacing of the transects was approximately 180 percent of the water depth (i.e., 20 percent overlap).

Water elevation was monitored during the survey by installing an automated tide gauge at the KPC facility. The automated gauge, a Stevens 420 Level Logger, recorded the water elevation at 1-minute intervals for the duration of the survey. Tide data were adjusted to mean lower low water (MLLW) by comparison of recorded tides with predicted tides for Ward Cove, measured tides in Ketchikan by the National Ocean Survey, and predicted tides for Ketchikan.

TABLE 2-5. SURFACE SEDIMENT STATION LOCATIONS, WATER DEPTHS, AND GENERAL SAMPLE CHARACTERISTICS FOR SEDIMENT SAMPLED IN WARD COVE AND MOSER BAY IN 1997

Station	Location		Water Depth[a] (m)	Sample Depth (cm)	Sediment Characteristics[b]
	Easting	Northing			
Ward Cove-Subtidal					
2	3088342.90	1309768.27	18.0	7–10	Dark brown to black color; soft fine grain sediment; wood debris (20–30 percent); evergreen tree needles; sulfide odor
3	3088945.07	1310050.62	15.0	5–10	Dark gray to black color; soft fine grain sediment; wood debris; sulfide odor
4	3089212.45	1310265.28	15.0	10	Dark gray to black color; soft fine grain sediment; red worms; sheen on surface; slight sulfide odor
5	3089490.58	1310629.58	6.0	4–9	Brown color; soft fine grain, sticky sediment; shell debris; mussel shell; sheen on surface; wood debris
7	3088518.59	1309402.55	25.0	5–10	Dark brown color; very soft fine grain to slightly sticky sediment; sheen and filamentous material on surface; mussel shells; shell debris; wood debris; sulfide odor
11	3085953.30	1307221.97	15.0	10	Dark brown to black color; soft fine grain sediment; lots of wood debris in surface layer (10–30 percent); small mussels; clam shell; shell debris; worm; seaweed; sulfide odor
12	3087176.91	1307799.35	22.0	10	Brown color; soft fine grain sediment; leaf on surface; copepods; shell debris; wood debris and large pieces of wood (removed); mussels; small stones; strong sulfide odor
13	3088162.63	1308299.75	40.0	10	Brown color; soft fine grain, sticky sediment; shell debris; large piece of wood (removed); white spherical objects (possibly eggs); mussel shells; sulfide odor
16	3089382.36	1309368.05	16.0	8–10	Brown to black color; soft fine grain, sticky sediment; wood debris (30–40 percent); sheen on surface; mussels; small shrimp; shell debris; rock (removed); slight sulfide odor
17	3089850.03	1309508.39	13.5	9–10	Dark brown color; soft fine grain, sticky sediment; wood debris and large pieces of wood (removed); shell debris; sulfide odor
18	3090336.11	1310216.65	4.0	9–10	Gray color; shell debris; rocks; several large clam shells; seaweed

TABLE 2-5. (cont.)

Station	Location		Water Depth[a] (m)	Sample Depth (cm)	Sediment Characteristics[b]
	Easting	Northing			
19	3087614.54	1307368.57	46.5	10	Dark brown color; soft fine grain sediment; wood debris; filamentous material on surface; starfish; vegetative debris; sulfide odor
22	3086804.83	1305129.66	33.0	6–10	Dark gray to black color; soft fine grain sediment mixed with some coarser grained sediment; shell debris; worm tubes; no odor; large rocks (removed from sample)
23	3087382.48	1305993.10	47.0	10	Dark gray to black color; soft fine grain sediment; slight sulfide odor
25	3088859.29	1307435.02	32.0	5–10	Dark gray to black color; soft fine grain sediment; worm; fish fin and bones; vegetative debris; sulfide odor
27	3089959.77	1308841.21	31.0	6–10	Dark gray to black color; soft fine grain sediment; slight sulfide odor
28	3090369.49	1309486.21	11.0	10	Dark gray to black color; soft fine grain sediment; wood debris and larger pieces of wood (removed); slight sulfide odor
31	3089694.41	1310885.00	5.0	7–10	Dark gray to black color; soft fine grain sediment; wood debris; sheen and filaments on surface; sulfide odor
32	3089857.34	1310772.26	4.0	6–10	Dark gray to black color; soft fine grain, sticky sediment; wood debris; mussel shell; evergreen tree needles; seaweed; sulfide odor
33	3090157.48	1310639.18	4.5	5–10	Dark gray to black color; soft fine grain, sticky sediment with some sand; wood debris and larger pieces of wood (removed); shell debris; large rocks (removed); sulfide odor; petroleum odor
34	3089508.32	1310358.76	10.5	10	Dark gray to black color; soft fine grain, sticky sediment; mussels; sheen on surface; wood debris and large pieces of wood on surface (removed); sulfie odor
35	3089555.30	1309934.28	12.0	5–10	Dark gray to black color; soft fine grain sediment; sea urchin; mussels; wood debris (40–50 percent) and; large pieces of wood on surface (removed); slight sulfide odor
37	3088691.88	1309841.65	16.0	10	Dark brown to blackish brown color; soft fine grain, sticky sediment; wood debris; sheen on surface; slight sulfide odor

TABLE 2-5. (cont.)

Station	Location		Water Depth[a] (m)	Sample Depth (cm)	Sediment Characteristics[b]
	Easting	Northing			
38	3088253.18	1309673.01	20.5	10	Dark gray color; soft fine grain sediment; wood debris; sheen on surface; sulfide odor
39	3088923.17	1309324.75	28.0	10	Dark brown to black color; soft fine grain sediment; wood debris and large piece of wood (removed); sheen and filamentous material on surface; sulfide odor
40	3089354.86	1309053.33	26.0	8–10	Dark brown color; soft fine grain, sticky sediment; wood debris and large piece of wood (removed); sheen on surface; sulfide odor
41	3089930.47	1309324.39	21.0	9–10	Dark brown color; soft fine grain, sticky sediment; bark on surface; wood debris; barnacle on wood; shell debris; copepod; eroded clam shell; terrestrial leaf; worms; no odor
42	3088553.80	1308975.07	30.0	7–10	Dark brown color; soft fine grain, sticky sediment; wood debris; sheen on surface; sulfide odor
43	3088920.59	1308623.03	37.0	5–10	Brown-gray to black color; soft fine grain sediment; wood debris; sheen and filaments on surface; sulfide odor
44	3087675.91	1308125.90	36.5	10	Dark brwon color; soft fine grain, sticky sediment; sheen and filamentous material on surface; white sherical objects (possibly eggs); little wood debris; mussels; sulfide odor
45	3087999.64	1308012.60	41.5	10	Dark brown color; soft fine grain sediment; wood debris; mussels; evergreen tree needles; shell debris; white spherical objects (possibly eggs); sulfide odor
47	3086397.86	1307513.35	13.0	6–10	Brown color; soft fine grain, sticky sediment; mussels; mussel shells; evergreen tree needles; wood debris; terrestrial leaf; copepod; shell debris; strong sulfide odor
48	3086949.72	1307551.53	30.5	10	Dark gray to black color; soft fine grain, slightly sticky sediment; copepod; mussels; worms; seaweed; wood debris; evergreen tree needles; sulfide odor
Ward Cove-Intertidal					
50	3090479.19	1310781.97		5	Gray color; sandy coarse grain sediment; small stones; lots of shell debris
51	3090797.26	1310494.45		5	Gray color; sandy coarse grain sediment; shell debris

TABLE 2-5. (cont.)

Station	Location		Water Depth[a] (m)	Sample Depth (cm)	Sediment Characteristics[b]
	Easting	Northing			
Moser Bay-Subtidal					
29	3102820.30	1360042.53	14.0	10	Gray brown color; soft fine grain sediment; seaweed; worm; shell debris; no odor
30	3102315.04	1362176.40	55.0	10	Gray brown color; soft fine grain, slightly sticky sediment; shell debris; pine cone

[a] Depths are presented to the nearest 0.5 m (mean lower low water).

[b] Wood debris: small wood chips and bark (unless otherwise noted)
Shell debris: small shell fragments
Vegetative debris: plant roots and leaves.

TABLE 2-6. STATION LOCATIONS, CORE SAMPLE DEPTHS, AND GENERAL SAMPLE CHARACTERISTICS FOR SUBSURFACE SEDIMENTS SAMPLED IN WARD COVE IN 1997

Station	Location		Recorded Core Depths (in.)[a]		Sediment Characteristics[b]
	Easting	Northing	Upper	Lower	
1	3088104.50	1309472.08	0.0	39.4	non-native organic material
			39.4	78.7	non-native organic material
			78.7	102.4	non-native organic material (no native)
2	3088342.90	1309768.27	0.0	39.4	non-native organic material
			39.4	78.7	non-native organic material
			78.7	102.0	non-native organic material (no native)
3	3088945.07	1310050.62	0.0	39.0	non-native organic material
			39.0	70.5	water break
			70.5	96.9	native clay/silt
4	3089212.45	1310265.28	0.0	39.4	non-native organic material
			39.4	72.4	non-native organic material
			72.4	89.8	water break
			89.8	108.7	native clay/silt
5	3089490.58	1310629.58	0.0	39.4	non-native organic material
			39.4	70.1	non-native organic material
			70.1	94.5	water break
			94.5	106.7	non-native organic material
			106.7	114.0	native clay/silt
6	3088081.46	1309004.41	0.0	39.4	non-native organic material
			39.4	78.7	non-native organic material
			78.7	105.1	non-native organic material (no native)
7	3088518.59	1309402.55	0.0	39.4	non-native organic material
			39.4	51.2	native clay/silt
			51.2	83.1	water break
			83.1	111.6	native clay/silt
8	3088887.43	1309613.82	0.0	39.4	non-native organic material
			39.4	47.6	non-native organic material
			47.6	116.1	native clay/silt
9	3089182.14	1309857.09	0.0	39.4	non-native organic material
			39.4	78.7	non-native organic material
			78.7	114.6	non-native organic material (no native)
12	3087176.91	1307799.35	0.0	39.4	non-native organic material
			39.4	56.3	non-native organic material
			56.3	75.2	water break
			75.2	92.1	native clay/silt

TABLE 2-6. (cont.)

Station	Location		Recorded Core Depths (in.)[a]		Sediment Characteristics[b]
	Easting	Northing	Upper	Lower	
13	3088162.63	1308299.75	0.0	39.4	non-native organic material
			39.4	57.5	non-native organic material
			57.5	87.4	native clay/silt
16	3089382.36	1309368.05	0.0	39.4	non-native organic material
			39.4	78.7	non-native organic material
			78.7	90.6	non-native organic material
			90.6	--[c]	native clay/silt[c]
33	3090157.48	1310639.18	0.0	39.4	non-native organic material
			39.4	56.7	non-native organic material
			56.7	67.7	native clay/silt
36	3090472.98	1309914.10	0.0	22.0	non-native organic material
			22.0	47.6	native clay/silt
40	3089354.86	1309053.33	--[d]	--[d]	--[d]
41	3089930.47	1309324.39	0.0	33.9	non-native organic material
			33.9	47.6	native clay/silt
46	3088218.89	1307836.05	0.0	4.7	non-native organic material
			4.7	67.7	native clay/silt
47	3086397.86	1307513.35	0.0	6.3	non-native organic material
			6.3	51.6	native clay/silt
49	3087008.55	1306845.93	0.0	3.9	non-native organic material
			3.9	63.8[e]	native clay/silt

[a] These depths were measured in field and recorded in logbook. At stations where more than one core was collected, the largest depth interval is used in this table.

[b] A more detailed description of the subsurface sediment characteristics is provided in Appendix C.

[c] Native materials were present only in nose cone of sampler; therefore, not enough material was available for analysis.

[d] The core sample collected at Station 40 was analyzed for lead-210 and cesium-137 only.

[e] The top 1 ft (30.5 cm) of native materials was collected for analysis.

Geophysical Survey: The geophysical survey coverage consisted of 25 track lines, oriented approximately north-south, spaced at an interval of approximately 40 m. The track lines ran from the head of the Cove to an area just west of Dawson Point, a distance of approximately 1,500 m. In addition, eight survey transects were run perpendicular to the main grid to validate the bathymetric data, and a transect was run parallel to the shoreline along the circumference of the Cove.

The surficial features of the seafloor were mapped using an EG&G Model 260 side-scan sonar. This system produces a plan view image of the seafloor to the left and right of the survey track line. Depending on the depth of water, the graphical display was set for a swath width that varied from 50 to 150 m.

Information on the shallow subsurface stratigraphy was obtained with a Datasonics Model 5000 SBP, interfaced with a 5-kHz transducer. The transducer was mounted on the port side of the vessel immediately below the navigation antenna. The data were processed using time-variable gain amplifiers and displayed on an EPC Model 1086 thermal graphic recorder. Position fixes were marked on the paper record at 20-second intervals.

A Datasonics Model 1200 Bubble Pulser, with a frequency band pass of 350 to 1 kHz, was used to obtain maximum subsurface penetration in the marine and glacial sediment. The acoustic energy source was towed on the port side of the vessel, and the hydrophone receiver was towed from the starboard side. The data were processed and filtered (band-pass 250 to 1200 Hz) and the graphic record was annotated in correlation to the navigation system at 20-second intervals.

To verify the results of the side-scan sonar and subbottom profiling data, video ground-truthing was performed using a Deepsea Power & Light Model SeaSnake 1000 high performance black and white video camera. Video images were obtained at five locations that were based on preliminary analysis of the side-scan sonar and subbottom profiler data. At each location, the camera was lowered to within approximately 1 m of the seabed, and video images were obtained as the survey vessel slowly drifted across the area. The data were recorded with voice annotation of the survey vessel position indicated by fixed numbers.

Surface Sediment Characterization—Surface sediment samples were collected for analysis of chemical concentrations, physical characteristics, and toxicity tests at 33 stations in Ward Cove (Figure 2-2) and at 2 reference stations in Moser Bay (Figure 2-3). In addition, surface sediment samples were collected for analysis of chemical concentrations and physical characteristics along two transects (five stations per transect) at the mouth of Ward Creek (i.e., intertidal stations; Figure 2-2). Phase 2 station locations were established on the basis of 1) coordinates from Phase 1 and historical NPDES monitoring stations, 2) bathymetry and log distribution data gathered during the

bathymetric and geophysical surveys, and 3) the specifications provided in the field sampling plan (PTI 1997f).

Surface sediment samples were collected for chemical analyses, toxicity testing, and specialized toxicity testing at eight stations in Ward Cove (Figure 2-9). The eight stations represent three subareas. Subarea 1 (Stations 12, 13, and 44) is the area along the northwest shoreline of the Cove, where ammonia was identified as the primary CoPC (PTI 1997g). Subarea 2 (Stations 16, 17, and 35) is the area in the center of the Cove offshore of the KPC facility, where sulfide was identified as the primary CoPC. Subarea 3 (Station 7 and 34) is the area immediately offshore of the KPC facility, where numerous CoPCs were identified. The rationale for the sampling strategy for specialized toxicity testing is described in greater detail in Appendix F of the Phase 2 field sampling plan (PTI 1997f).

Surface sediment samples were collected using a stainless-steel, 0.06-m^2 modified van Veen bottom grab sampler from a boat equipped with a winch, davit, and pulley assembly. All sampling equipment was constructed of stainless steel and was decontaminated prior to sampling according to the procedures described in the field sampling plan (PTI 1997f). Although the target sediment horizon was 0–10 cm, shallower horizons were collected at selected stations if the target horizon could not be sampled after repeated sampling attempts (Table 2-5). Surficial sediment samples were collected and composited for chemical and toxicity testing at all stations. Based on EPA sediment sampling guidance (U.S. EPA 1991c), unrepresentative material was removed from the sediment samples before samples were composited. Only materials (i.e., wood debris) that were large enough to be removed without contaminating the sample were removed in the field. Sediment samples were homogenized in a large stainless-steel bowl, and aliquots were collected from the homogenized samples for the individual analyses and toxicity tests. The samples were placed into appropriate chemically cleaned containers and held at 4°C during shipment and prior to testing. An additional aliquot of each sample was collected for potential future analysis. These archive samples were placed into frozen storage (−20°C) upon arrival at the laboratory.

Intertidal Sediment Characterization—Surface sediment samples (0–5 cm sediment horizon) were collected at two intertidal transects at the mouth of Ward Creek (Figure 2-2). Each transect comprised five stations.

Sediment samples were collected using stainless-steel spoons in accordance with standard methods used by U.S. EPA (1991c). Sediments from each station were composited to achieve a sample more representative of average surface sediment characteristics in the intertidal area. All sampling equipment was constructed of stainless steel and was decontaminated prior to sampling according to the procedures described in the field sampling plan (PTI 1997f). Sediment samples were homogenized in a large stainless-steel bowl, and aliquots were collected from the homogenized samples for the individual analyses. The samples were placed into appropriate chemically cleaned containers and held at 4°C during shipment and prior to testing. An additional aliquot of each sample

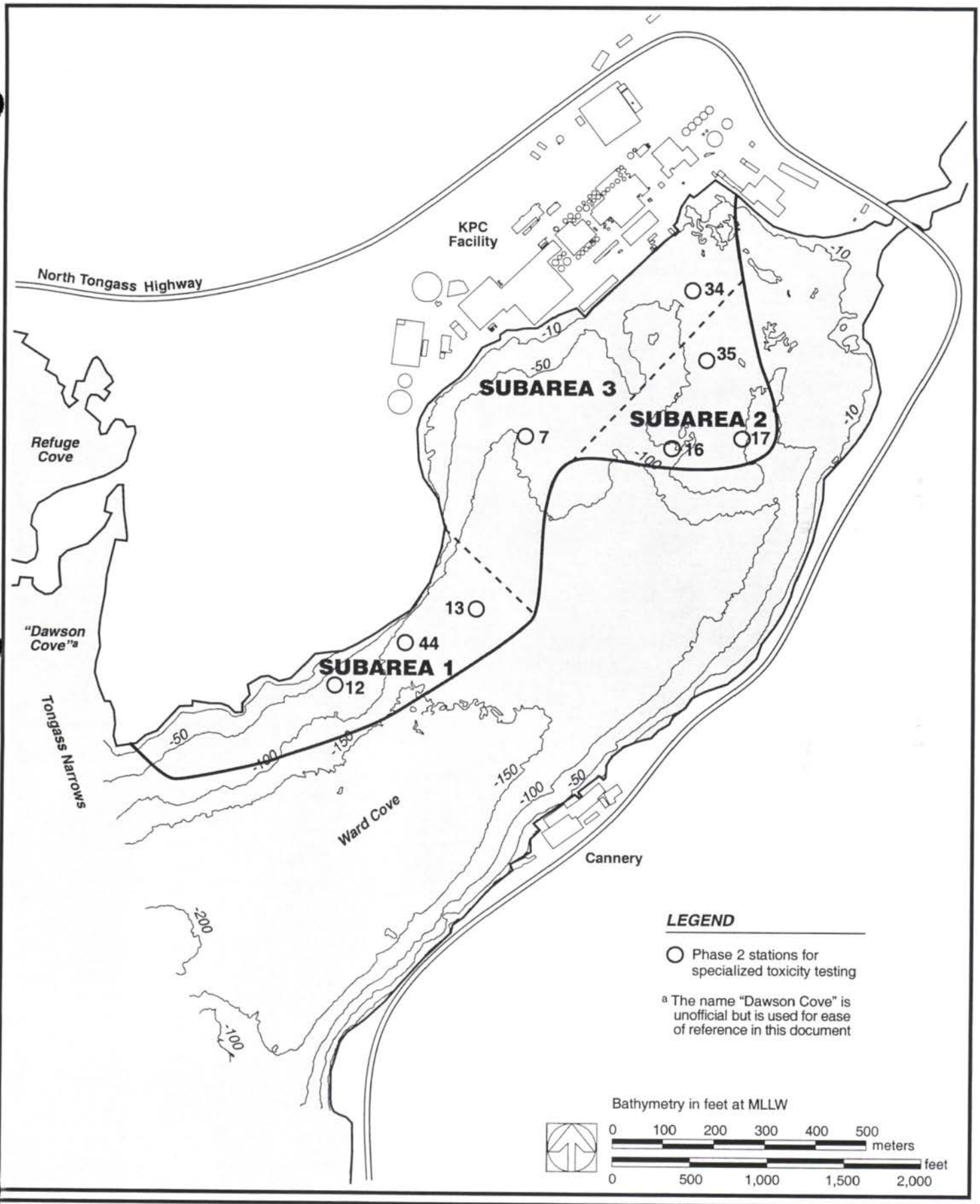

Figure 2-9. Station locations in Ward Cove at which surface sediments were collected for specialized toxicity testing in 1997.

was collected for potential future analysis. These archive samples were placed into frozen storage (−20°C) upon arrival at the laboratory.

Sediment Column Characterization—Subsurface sediment samples were collected for analysis of chemical concentrations and physical characteristics at 18 stations in Ward Cove (Figure 2-6). Phase 2 station locations were established on the basis of 1) coordinates of historical subsurface sampling stations (ENSR 1996b), 2) bathymetry and log distribution data gathered during the bathymetric and geophysical surveys, and 3) the specifications provided in the field sampling plan (PTI 1997f).

Subsurface sediment samples were collected using a piston corer from a boat equipped with a winch, davit, and pulley assembly. Polyethylene core liners were used in the corer. Prior to sampling, all core liners were decontaminated according to the procedures described in the field sampling plan (PTI 1997f). For most sediment cores, the depth of interest in the sediment is the depth at which the pulp mill compounds are above background concentrations. Therefore, core samples were collected and analyzed from the top down to the first stratum that appeared to represent native sediments. Sediment samples were collected for analysis from discrete horizons having a uniform appearance. Representative subsamples from each horizon over a depth not to exceed 3 ft were composited to create a single sample representing bulk chemical characteristics for that horizon. Where horizon thickness was greater than 3 ft, two composite samples were collected.

To characterize the sediment that will become surface sediment after any proposed dredging, the top 0–1 ft of native sediments was collected for analysis at the stations shown in Figure 2-8. An archive sample of the top 0–1 ft of native sediments was collected at all of the other Phase 2 subsurface sediment stations that reached to native sediment. A sediment sample was also collected and composited across discrete horizons of similar appearance from the station groups shown in Figure 2-7 for analysis of dioxins and furans.

For samples requiring compositing, each core section was placed in a separate stainless-steel bowl and covered with aluminum foil until all remaining sections were sampled. The sediment from each core section was then mixed in a stainless-steel bowl with stainless-steel spoons to achieve a uniform texture and color. The homogenized sediment was subsampled and transferred to sample containers. The samples were placed into appropriate chemically cleaned containers and held at 4°C during shipment and prior to testing. Samples for archive were placed into frozen storage (−20°C) upon arrival at the laboratory.

Sediment Accumulation Testing—Sediment cores were collected from two stations (Figure 2-6) in Ward Cove to characterize the accumulation rates of sediments. These stations were positioned outside of the area of influence of the historical and current outfalls from the KPC facility and outside the influence of the fish cannery.

Sediment samples were collected using a piston corer from a boat equipped with a winch, davit, and pulley assembly. Polyethylene core liners were used in the corer. Prior to sampling, all core liners were decontaminated according to the procedures described in the field sampling plan (PTI 1997f). Sediment cores of approximately 60–80 cm in length were collected. Depth horizons of 2 cm were extracted from the core and analyzed for lead-210 and cesium-137 at multiple depths throughout the core. The samples were placed into appropriate chemically cleaned containers and held at 4°C during shipment and prior to testing. Sample horizons that were not analyzed were placed into frozen storage (−20°C) upon arrival at the laboratory and archived.

Engineering Properties Testing—Sediment cores were collected from four stations (Figure 2-10) in Ward Cove and were composited into two sediment samples for testing of engineering properties. One sample represented a composite of core samples from Stations 1 and 7, and the other sample represented a composite of core samples from Stations 3 and 5 (Figure 2-10).

Sediment samples were collected using a piston corer from a boat equipped with a winch, davit, and pulley assembly. Polyethylene core liners were used in the corer. Prior to sampling, all core liners were decontaminated according to the procedures described in the field sampling plan (PTI 1997f). The compositing of samples for analysis of these properties followed PSDDA guidelines (1989). Cores were composited over a 4-ft depth interval, the interval considered to be representative of a typical dredge cut.

Water Sampling—Water samples to support engineering testing were collected from four stations (Figure 2-10) in Ward Cove and were composited into two water samples, with one sample representing a composite of water samples from Stations 1 and 7 and the other sample representing a composite of water samples from Stations 3 and 5 (Figure 2-10). Water samples were collected near the sediment surface in the vicinity of the core sampling stations where subsurface sediment was collected for analysis of the sediment's engineering properties.

Water samples to support engineering testing were collected using a peristaltic pump with Teflon® tubing. Samples were collected at the same time that sediment cores were being collected for the engineering tests. The Teflon® tubing was decontaminated with acid and rinsed with distilled/deionized water prior to sampling. The samples were placed into Teflon® bags and held at 4°C during shipment and prior to testing.

Bottom water samples to assess nutrient concentrations were collected in the vicinity of sediment Stations 2, 8, 16, and 41. These water samples were collected in the vicinity of the sediment sampling stations. These water samples were collected using a Niskin water-bottle sampler in accordance with standard methods used by U.S. EPA (1991c). The water-bottle sampler sampled a discrete parcel of water at the designated depth (i.e., 1 m above the sediment surface). The interior of each water-bottle sampler was

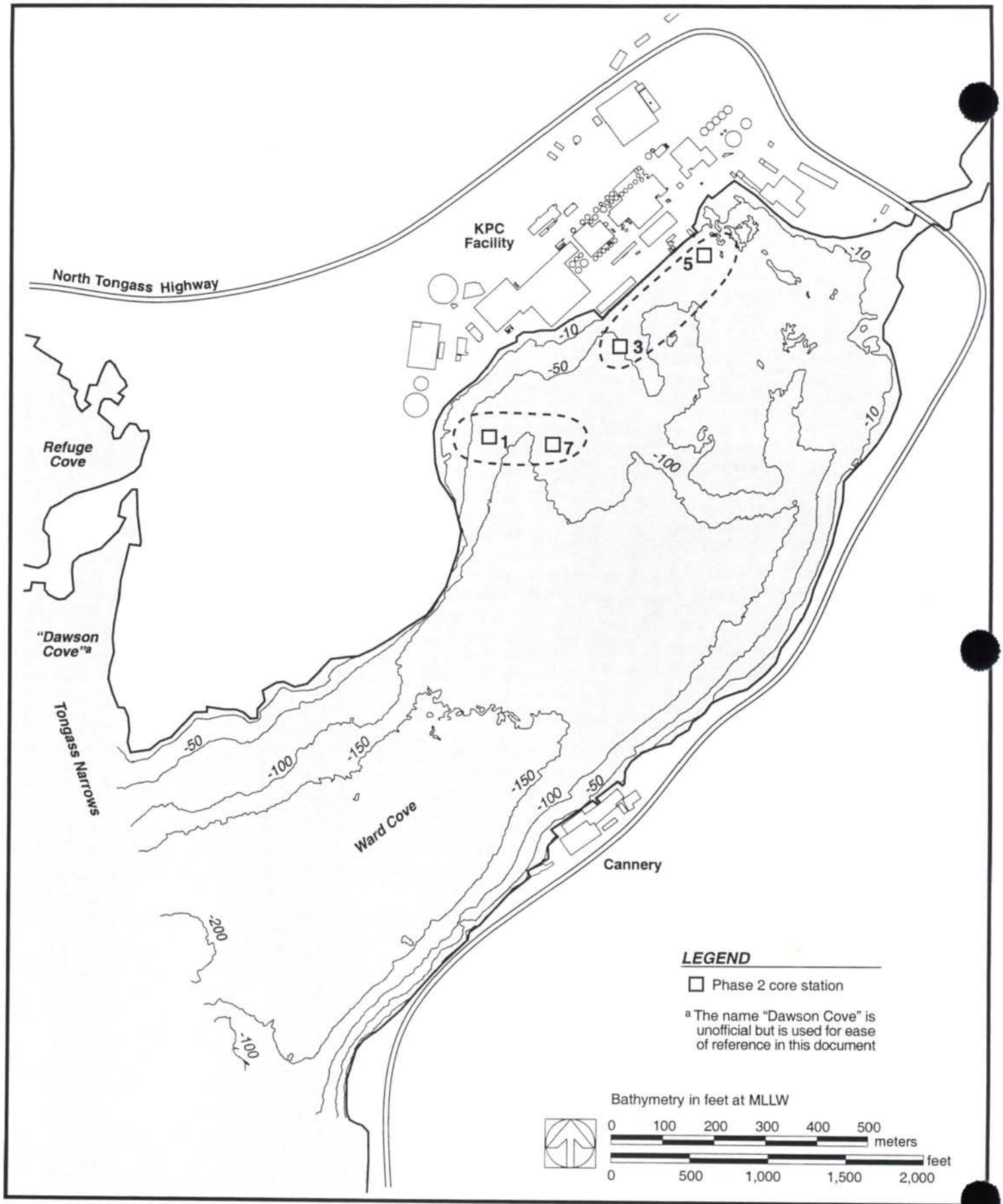

Figure 2-10. Station locations in Ward Cove at which sediment cores were collected and composited for sediment column characterization of engineering properties in 1997.

washed periodically with 10-percent hydrochloric acid. Immediately after sample containers were filled, all samples were preserved with sulfuric acid and placed on ice in a cooler at 4°C.

2.3.2.4 Phase 2 Laboratory Methods

The surface sediment, subsurface sediment, bottom water, elutriate, and equipment rinsate blank analyses were completed by five laboratories. Analyses for PCDDs/Fs were completed by Zenon Environmental Laboratories (Burlington, Ontario, Canada); analyses for methylmercury were completed by Frontier Geosciences (Seattle, Washington); analyses of engineering properties were completed by Soil Technology, Inc. (Bainbridge Island, Washington); analyses for cesiums-137 and lead-210 were completed by Battelle Marine Sciences Laboratory (Sequim, Washington); and the remaining analyses were completed by Columbia Analytical Services, Inc. (Kelso, Washington). PSEP methods, EPA methods, and laboratory-specific standard operating procedures were used to complete the analyses whenever possible. A summary of analytical methods is provided in Table B2-3 in Appendix B2.

The compounds 3-methylphenol and 4-methylphenol commonly coelute from the chromatographic column under conditions prescribed by EPA Method 8270 (modified to include selected ion monitoring for optimal detection limits) and cannot be differentiated. During the 1995 sediment monitoring study, ENSR (1995b) analyzed 3- and 4-methylphenol separately in Ward Cove sediments and found that only 4-methylphenol is present at detectable concentrations. 3-Methylphenol was not detected in any sample. Consequently, separate analyses of 3- and 4-methylphenol were not conducted for the present study, and the laboratory reported results only for the sum of 3- and 4-methylphenol. Because 3-methylphenol was shown to be absent from the site, these results for the combined methylphenols were treated as concentrations of 4-methylphenol exclusively for all aspects of this study.

In addition to chemical analysis of the sediment, standard toxicity tests and specialized toxicity tests were performed.

Two standard toxicity tests were performed: the amphipod test based on *Rhepoxynius abronius*, and the echinoderm embryo test based on *Dendraster excentricus* (sand dollar). The laboratory methods used for the amphipod test and the echinoderm embryo test were the methods recommended by PSEP (1995), as modified by the PSDDA program (PSDDA 1989), public workshops, and the annual review process. Five replicate subsamples of each sediment sample collected in the field were tested in the laboratory. The following major laboratory specifications were used for the standard toxicity tests:

- A maximum sediment holding time of 14 days after field collection
- Exposure periods as specified in the respective test protocol

- Water quality parameters (e.g., temperature, dissolved oxygen) as specified in the respective test protocol
- Aeration during testing
- Positive controls using cadmium chloride as a reference toxicant
- Negative controls using clean sediment or seawater.

In the Phase 2 investigation, specialized toxicity tests were performed in addition to the toxicity tests. The specialized toxicity tests were conducted primarily to evaluate the role of ammonia and sulfide in causing sediment toxicity. The four specialized toxicity tests included 1) a sediment purging procedure using the amphipod *Rhepoxynius abronius*, 2) a sediment *Ulva* procedure using *R. abronius*, 3) a porewater *Ulva* procedure using *R. abronius* and the echinoderm *Dendraster excentricus*, and 4) a porewater aeration procedure using *R. abronius*. The laboratory methods used for the specialized toxicity tests were based on modifications of the procedures used by EPA to conduct specialized toxicity testing of marine effluents and receiving waters (U.S. EPA 1996c) and testing with *D. excentricus* (U.S. EPA 1993c). Several of these procedures were modified for application to marine sediments (Ho et al. 1997, unpublished). In addition, another procedure recommended by U.S. EPA (1994e) for evaluating ammonia toxicity as part of dredged material testing was used in the Phase 2 study. The following major laboratory specifications were used for the specialized toxicity tests:

- Exposure periods as specified in the respective test protocol
- Water quality parameters (e.g., temperature, dissolved oxygen) as specified in the respective test protocol
- Aeration during testing (when appropriate)
- Positive controls using cadmium chloride as a reference toxicant
- Negative controls using clean sediment or seawater.

The toxicity tests and the specialized toxicity tests were completed by Northwestern Aquatic Sciences (Newport, Oregon).

2.3.2.5 Phase 2 Data Quality

The following sections describe the results of the quality assurance review of the Phase 2 data for chemical analyses and toxicity tests.

Phase 2 Chemical Analyses—A complete quality assurance report is provided in Appendix B2. Some of the results (Appendix A1) were qualified as estimated (*J*) during the quality assurance review. As noted in U.S. EPA (1989d): "The *J*-qualifier

is placed on CLP data to provide important information about an analysis to the data user or decision-maker, not to indicate low confidence in the analysis." Also noted in U.S. EPA (1989d), "The *J*-qualifier is a quantitative qualifier and can mean one or more of several things: 1) the target analyte is definitely present, 2) the sample was difficult to analyze, 3) the value may lie near the low end of the linear range of the instrument, and 4) the value should nearly always be seriously considered in decision-making."

Conventional Analytes in Surface Sediment Samples: The laboratory reported a total of 632 results for conventional analytes in surface sediment samples. Extractable organic halides were not detected in 12 of 16 samples. All other conventional analytes were present at concentrations above the detection limits in all samples, with the exception of one result for total sulfide and two results for grain size fraction. Results are provided in Table A1-1, Appendix A1.

During the quality assurance review, 14 total sulfide results reported as detected were qualified as estimated (*J*) because holding time constraints were not met.

Conventional Analytes in Subsurface Sediment Samples: The laboratory reported a total of 553 results for conventional analytes in subsurface sediment samples. All conventional analytes were present at concentrations above the detection limits in all samples, with the exception of one result reported for BOD. Results are provided in Table A1-6, Appendix A1.

Eight results were qualified as estimated (*J*) during the quality assurance review.

Conventional Analytes in Bottom Water Samples: The laboratory reported a total of four results for ammonia in bottom water samples. Ammonia was present at concentrations above the detection limit in all four samples. Results are provided in Table A1-11, Appendix A1.

No results were qualified as estimated during the quality assurance review.

Conventional Analytes in Elutriate Samples: The laboratory reported a total of 12 results for conventional analytes in elutriate samples. All conventional analytes were present at concentrations above the detection limits in all samples. Results are provided in Table A1-6, Appendix A1.

No results were qualified as estimated during the quality assurance review.

Metals in Surface Sediment Samples: The laboratory reported a total of 81 results for metals in surface sediment samples. Total mercury was undetected in 17 samples. Arsenic, cadmium, methylmercury, and zinc were detected in all samples. Results are provided in Table A1-2, Appendix A1.

No results were qualified as estimated during the quality assurance review.

Metals in Subsurface Sediment Samples: The laboratory reported a total of 111 results for metals in subsurface sediment samples. Of these results, 81 were reported at a concentration above the method detection limit and 30 results for total mercury were reported as undetected. Results are provided in Table A1-7, Appendix A1.

No results were qualified as estimated during the quality assurance review.

Metals in Elutriate Samples: The laboratory reported a total of 30 results for metals in elutriate samples. Of these results, 10 were reported at a concentration above the method detection limit and 20 were reported as undetected. Results are provided in Table A1-13, Appendix A1.

No results were qualified as estimated during the quality assurance review.

Metals in Equipment Rinsate Blank Samples: The laboratory reported a total of 16 results for metals in elutriate samples. Of these results, 8 were reported at a concentration above the method detection limit and 8 were reported as undetected. Results are provided in Table A1-16, Appendix A1.

No results were qualified as estimated during the quality assurance review.

Semivolatile Organic Compounds in Surface Sediment Samples: The laboratory reported a total of 420 results for PAHs, phenol, 4-methylphenol, benzoic acid, and dibenzofuran in surface sediment samples. Of these results, 298 were reported at a concentration above the method detection limit and 122 were reported as undetected. The detection limits for some of the samples were elevated because matrix interference necessitated sample dilution for analysis. Results are provided in Table A1-3, Appendix A1. Consistent with the approach recommended by EPA, concentrations of carcinogenic PAHs (i.e., benzo[a]pyrene, benz[a]anthracene, benzo[b]fluoranthene, chrysene, benzo[k]fluoranthene, indeno[1,2,3-cd]pyrene, and dibenz[a,h]anthracene) were calculated as the RPC by adjusting their concentrations to reflect their carcinogenic potency relative to that of benzo[a]pyrene. In calculating RPCs, undetected carcinogenic PAHs were included in calculations using one-half the detection limit.

During the quality assurance review, 295 results were qualified as estimated (*J*) for exceeding holding time constraints, and 22 results were qualified for exceeding control limits for matrix spikes or laboratory control samples.

Semivolatile Organic Compounds in Subsurface Sediment Samples: The laboratory reported a total of 74 results for phenol and 4-methylphenol in subsurface sediment samples. All results were reported at a concentration above the method detection limit, with the exception of one result for phenol and two results for 4-methylphenol. Results are provided in Table A1-8, Appendix A1.

During the quality assurance review, 16 results were qualified as estimated (*J*) because quality control criteria were not met for surrogate recovery.

Semivolatile Organic Compounds in Elutriate Samples: The laboratory reported a total of 20 results for phenol and 4-methylphenol in elutriate samples. Of these results, 6 were reported at a concentration above the method detection limit and 14 were reported as undetected. Results are provided in Table A1-14, Appendix A1.

During the quality assurance review, four results were qualified as estimated (*J*) for exceeding holding time constraints, and two results were qualified for surrogate compound recoveries below the lower control limit.

Semivolatile Organic Compounds in Equipment Rinsate Blank Samples: The laboratory reported a total of 84 results for PAHs, phenol, 4-methylphenol, benzoic acid, and dibenzofuran in equipment rinsate blank samples. Of these results, 5 were reported at a concentration above the method detection limit and 79 were reported as undetected. Results are provided in Table A1-17, Appendix A1. Consistent with the approach recommended by EPA, concentrations of carcinogenic PAHs (i.e., benzo[a]pyrene, benz[a]anthracene, benzo[b]fluoranthene, chrysene, benzo[k]-fluoranthene, indeno[1,2,3-cd]pyrene, and dibenz[a,h]anthracene) were calculated as the RPC by adjusting their concentrations to reflect their carcinogenic potency relative to that of benzo[a]pyrene. In calculating RPCs, undetected carcinogenic PAHs were included in calculations using one-half of the detection limit.

No results were qualified as estimated during the quality assurance review.

PCDDs/Fs in Phase 1 Archived Surface Sediment Samples: The laboratory reported a total of 204 results for PCDD/F congeners and 120 results for total homologs (total congeners at each chlorination level) in Phase 1 archived surface sediment samples. Of these results, 266 were reported at a concentration above the

method detection limit and 58 were reported as undetected. Results are provided in Table A1-4, Appendix A1. To be consistent with methods used by EPA in evaluating PCDDs/Fs, where possible, PCDD/F concentrations were provided as TECs, wherein concentrations of PCDD/F congeners that EPA considers to be carcinogenic (i.e., congeners substituted with chlorine at the 2, 3, 7, and 8 positions) were adjusted to reflect the assumed carcinogenic potency relative to that of 2,3,7,8-TCDD (U.S. EPA 1989c). In calculating TECs, one-half the detection limit was used for undetected relevant congeners.

During the quality assurance review, 54 results were qualified as estimated (*J*) and 41 results were restated as undetected (a *U* qualifier was assigned to the results reported by the laboratory). These results were qualified because the target analytes were detected in the associated method blank at a concentration above the action limit.

PCDDs/Fs in Subsurface Sediment Samples: The laboratory reported a total of 85 results for PCDD/F congeners and 50 results for total homologs (total congeners at each chlorination level) in subsurface sediment samples. Of these results, 105 were reported at a concentration above the method detection limit and 30 were reported as undetected. Results are provided in Table A1-9, Appendix A1. To be consistent with methods used by EPA in evaluating PCDDs/Fs, where possible, PCDD/F concentrations were provided as TECs, wherein concentrations of PCDD/F congeners that EPA considers to be carcinogenic (i.e., congeners substituted with chlorine at the 2, 3, 7, and 8 positions) were adjusted to reflect the assumed carcinogenic potency relative to that of 2,3,7,8-TCDD (U.S. EPA 1989c). In calculating TECs, one-half the detection limit was used for undetected relevant congeners.

During the quality assurance review, 3 results were qualified as estimated (*J*) and 10 results were restated as undetected (a *U* qualifier was assigned to the results reported by the laboratory). These results were qualified because the target analytes were detected in the associated method blank at a concentration above the action limit.

PCDDs/Fs in Elutriate Samples: The laboratory reported a total of 170 results for PCDD/F congeners and 100 results for total homologs (total congeners at each chlorination level) in elutriate samples. Of these results, 112 were reported at a concentration above the method detection limit and 158 were reported as undetected. Results are provided in Table A1-15, Appendix A1. To be consistent with methods used by EPA in evaluating PCDDs/Fs, where possible, PCDD/F concentrations were provided as TECs, wherein concentrations of PCDD/F congeners that EPA considers to be carcinogenic (i.e., congeners substituted with chlorine at the 2, 3, 7, and 8 positions) were adjusted to reflect the assumed carcinogenic potency relative to that of 2,3,7,8-TCDD (U.S. EPA 1989c). In calculating TECs, one-half the detection limit was used for undetected relevant congeners.

During the quality assurance review, 4 results were qualified as estimated (*J*) and 11 results were restated as undetected (a *U* qualifier was assigned to the results reported by the laboratory). These results were qualified because the affected target analytes were detected in the associated method blank at a concentration above the action limit.

Cesium-137 and Lead-210 in Subsurface Sediment Samples: The laboratory reported a total of 27 results for cesium-137 and a total of 35 results for lead-210 in subsurface sediment samples. Of these results, 12 results for cesium-137 and all 35 results for lead-210 were reported at a concentration above the method detection limit, and 15 results for cesium-137 were reported as undetected. Results are provided in Table A1-10, Appendix A1.

No results were qualified as estimated during the quality assurance review.

Engineering Properties of Sediment Samples: The laboratory reported results for engineering properties (i.e., geotechnical parameters) of sediment samples to provide information for selecting the type of dredge required, determining disposal site conditions, and identifying potential capping alternatives. Test results for engineering properties included the following:

- The modified elutriate test (MET) and dredging elutriate test (DRET) for chemical analyses of two composite sediment samples.
- Column settling testing of two composite sediment samples. Results were reported for specific gravity; total solids; total suspended solids (TSS) concentrations over time at specific heights of measurement; turbidity measurements over time at specific heights of measurement; interface heights; retention time vs. TSS; concentration profiles; retention time vs. averaged TSS; turbidity vs. TSS; and elapsed time vs. interface heights.
- For 12 sediment samples, results were reported for grain size, water content, void ratios, specific gravity, TVS, and Atterberg limits (i.e., liquid limits, plastic limits, plasticity index, and soil classification).
- Consolidation testing of one composite sediment sample.

Results for the engineering properties data are provided in Appendix A5. Results for the chemical analyses for the elutriate tests are provided in Appendix A1.

No results were qualified as estimated during the quality assurance review. Desiccation characteristics could not be determined by the laboratory because the samples were predominantly composed of organic matter and not cohesive sediment, which is required to determine desiccation characteristics. In addition, five samples were not analyzed for Atterberg limits and one sample did not undergo consolidation testing because the major

constituent of these samples was coarse wood fiber and the test data would not have reflected the true nature of the material. The lack of these data is not a reflection of poor laboratory performance, but is due to the physical nature of the material collected.

Phase 2 Toxicity Tests—A quality assurance review of the results of the two standard sediment toxicity tests was performed. The results for the sediment toxicity study are summarized in Section 7. Details of the quality assurance review are provided in Appendix B4. The results of the specialized toxicity tests and water quality data for each replicate sample are presented in Appendices A2 and A3, respectively. A summary of data quality is provided below.

Amphipod Toxicity Test Based on *Rhepoxynius abronius*: The recommended protocols were followed closely during testing. However, the specified holding time of 14 days was exceeded for five of the samples. Four samples exceeded the holding time by 1 day, and one sample exceeded the holding time by 2 days.

Water quality parameters were measured in the overlying water in all test replicates at test initiation and test termination and daily in one test replicate of each sample. The specified temperature range of 15 ± 1°C (i.e., 14–16°C) was exceeded during the acclimation period (exceedance of 1.4°C) and by a small amount (exceedance of 0.1–0.7°C) two days during the testing period in some of the test replicates. In addition, at test termination, one test replicate for samples collected at Stations 31 and 32, respectively, was siphoned prior to collection of the water quality measurements. There were no other deviations from the specified temperature. The specified salinity range of 28 ± 1 ppt was exceeded during the acclimation period (exceedance of 1.3 ppt) and was often exceeded during testing (exceedance of 0.5–2.0 ppt). Concentrations of dissolved oxygen were equal to or greater than the recommended minimum level of 5.0 mg/L for all control and test sediment replicates. Values of pH ranged from 7.4 to 8.5 and were all within the recommended range of 7.0–9.0. The concentration of ammonia in the overlying water ranged from less than 0.2 mg/L (detection limit) to 10.5 mg/L, and the concentration of ammonia in the pore water of the test sediments at test termination ranged from less than 1.0 to 14 mg/L. The concentration of total sulfide in the overlying water was less than 0.01 mg/L (detection limit), and the sulfide concentrations in the pore water of the test sediments at test termination ranged from less than 1.3 to 28.1 mg/L.

The negative control consisted of sediment from Yaquina Bay, Oregon. The mean survival value for the negative control sediment was 100 percent, which exceeds the performance criterion of 90 percent (Ecology 1995). The mean survival values for sediments from the two reference area samples were both 96 percent, which exceeds the performance criterion of 75 percent (Ecology 1995).

A positive control was tested using cadmium chloride as the reference toxicant. Because the supply of test organisms was limited, fewer test organisms and fewer replicates were used in the reference toxicant test for this study. The positive control exhibited a 96-hour

LC50 value of 0.61 mg/L, which is within the testing laboratory's control chart warning limits for this test. The observed LC50 value suggests that the test organisms were suitably sensitive for testing.

Because the amphipod test was conducted using appropriate protocols, water quality variables were generally within acceptable ranges, and performance criteria were achieved for the negative control and reference area samples, the results are considered acceptable for use in evaluating the toxicity of Ward Cove sediments.

Echinoderm Embryo Test Based on *Dendraster excentricus*: The recommended protocols were followed closely during testing. However, the specified holding time of 14 days was exceeded for five of the samples. Four samples exceeded the holding time by 1 day, and one sample exceeded the holding time by 2 days. In addition, as determined by the laboratory, the initial concentration of test organisms in the test chambers was 17.3 test organisms/mL. The protocol specifies a range of 20–30 test organisms/mL.

Water quality parameters were measured in the overlying water at test initiation and test termination in all test replicates and daily in one test replicate of each sample. The specified temperature range of 15 ± 1°C (i.e., 14–16°C) was exceeded during the acclimation period (exceedance of 2°C) and by a small amount (exceedance of 0.1–0.2°C) on one day during testing in some of the replicates. There were no other deviations from the specified temperature. There were no deviations from the specified salinity of 31 ± 1 ppt. Concentrations of dissolved oxygen were greater than the recommended minimum level of 5.0 mg/L for all control and test sediment replicates. Values of pH ranged from 7.6 to 8.0 and were all within the recommended range of 7.0–9.0. The concentration of ammonia in the overlying water ranged from less than 0.2 mg/L (detection limit) to 0.9 mg/L, and the concentration of sulfide in the overlying water was less than 0.01 mg/L (detection limit).

The negative control consisted of seawater from Yaquina Bay, Oregon. In the negative control (i.e., seawater), 73.8 percent of the inoculated embryos produced normal pluteus larvae. This value exceeds the test acceptance criterion of 70 percent (PSEP 1995).

A positive control was tested using cadmium chloride as the reference toxicant. The positive control exhibited an EC50 value of 11.2 mg/L, which is within the laboratory's control chart warning limits (4.66 to 11.9 mg/L). The observed EC50 value suggests that the test organisms were suitably sensitive for testing.

Because the echinoderm embryo test was conducted using appropriate protocols, water quality variables were generally within acceptable ranges, and performance criteria were achieved for the negative control, the results are considered acceptable for use in evaluating the toxicity of Ward Cove sediments.

Phase 2 Specialized Toxicity Tests—A quality assurance review of the results of the four specialized toxicity tests was performed. The results for the sediment toxicity evaluation are summarized in Section 7. Details of the quality assurance review are provided in Appendix B5. The toxicity results and water quality data for each replicate sample are presented in Appendices A2 and A3, respectively. A summary of data quality is provided below.

Sediment Purging Procedure: The recommended protocols were followed closely during testing; however, sediment holding times were exceeded prior to test initiation. In addition, purging of all sediment samples was conducted for 10 days, despite the fact that porewater ammonia concentrations after the first day of purging in all samples ranged from 4.0 to 16 mg/L, which is considerably less than the no-effect concentration of 30 mg/L for *Rhepoxynius abronius*. The laboratory continued purging all samples because elevated porewater sulfide concentrations persisted in all samples. Despite the departure from the protocols specified by U.S. EPA (1994e), the resulting information is considered useful because it provides relevant information on the effects of porewater ammonia and sulfide on amphipod toxicity at the concentrations present after purging was completed.

Water quality parameters (i.e., temperature, dissolved oxygen, salinity, and pH) were measured in the overlying water in all the replicates at test initiation and test termination. Ammonia and sulfide were measured in one replicate at test initiation and test termination (i.e., water quality beaker) of each test sample. In addition, temperature, dissolved oxygen, salinity, and pH were measured daily in one replicate of each test sample.

The specified temperature range of 15 ± 1°C (i.e., 14–16°C) was exceeded during the acclimation period (exceedance of 0.7°C), by a small amount during the purging period (exceedance of 0.9–1.0°C), and during the testing period (exceedance of 0.1–1.0°C) in some of the test replicates. There were no other deviations from the specified temperature. The specified salinity range of 28 ± 1 ppt was exceeded during the acclimation period (exceedance of 6.0 ppt). The specified salinity range was never exceeded during the purging period, but it was often exceeded during the testing period (exceedance of 0.5–1.0 ppt). Concentrations of dissolved oxygen were greater than the recommended minimum level of 5.0 mg/L for all control and test sediment replicates during the purging period and the testing period. During the purging period and the testing period, pH values ranged from 7.7 to 8.2 and were all within the recommended range of 7.0–9.0. Ammonia concentrations in overlying water declined during the initial phases of the purging period, but substantially increased in the overlying water 7 days after the end of the purging period. Ammonia concentrations in the pore water ranged from 2.0 to 16.0 mg/L. In general, the ammonia concentration in the overlying water appeared to correlate with concentrations in the sediment pore water. Sulfide in the overlying water was less than 0.01 mg/L (detection limit) for all samples. Porewater concentrations of sulfide at the beginning of the purging period ranged from 3.8 to 38.8 mg/L and gradually declined throughout the purging period (less than 2.5 to 22.5 mg/L).

The negative control consisted of sediment from Yaquina Bay, Oregon. The mean survival value for the negative control sediment was 98 percent, which exceeds the performance criterion of 90 percent (Ecology 1995).

Because the amphipod test was conducted using appropriate protocols, water quality variables were generally within acceptable ranges, and performance criteria were achieved for the negative control and reference area samples, the results are considered acceptable for use in evaluating the toxicity of Ward Cove sediments.

Sediment *Ulva* Procedure: The recommended protocols were followed closely during testing; however, sediment holding times were exceeded prior to test initiation. Water quality parameters (i.e., temperature, dissolved oxygen, salinity, and pH) were measured daily in the overlying water in one test replicate of each sample. In addition, ammonia and sulfide were measured in one replicate at test initiation and test termination (i.e., water quality beaker) of each test sample.

The specified temperature range of 15 ± 1°C (i.e., 14–16°C) was exceeded during the acclimation period (exceedance of 2.3°C). There were no deviations from the specified temperature range during testing. The specified salinity range of 28 ± 1 ppt was exceeded during the acclimation period (exceedance of 5.0 ppt) and was often exceeded during testing (exceedance of 1.0–2.0 ppt). Concentrations of dissolved oxygen were greater than the recommended minimum level of 5.0 mg/L for all control and test sediment replicates during both the purging period and the testing period. Values for pH ranged from 7.7 to 8.4 and were all within the recommended range of 7.0–9.0. Ammonia concentrations in sediment pore water in the untreated samples (i.e., no *Ulva*) ranged from 2.0 to 12.0 mg/L. Ammonia concentrations in sediment pore water in the treated samples were less than 0.5 mg/L, except for one sample, which had an ammonia concentration of 2.0 mg/L. Sulfide concentrations in the untreated samples ranged from less than 0.01 mg/L (undetected) to 5.3 mg/L and in the treated samples were all less than 0.01 mg/L (undetected).

The negative control consisted of sediment from Yaquina Bay, Oregon. The mean survival value for the negative control sediment was 100 percent, which exceeds the performance criterion of 90 percent (Ecology 1995).

Because the amphipod test was conducted using appropriate protocols, water quality variables were generally within acceptable ranges, and performance criteria were achieved for the negative control and reference area samples, the results are considered acceptable for use in evaluating the toxicity of Ward Cove sediments.

Porewater *Ulva* Procedure: The recommended protocols were followed closely during testing; however, sediment holding times were exceeded prior to test initiation. Water quality parameters (i.e., temperature, dissolved oxygen, salinity,

and pH) were measured daily in one test replicate of each sample; however, ammonia and sulfide were measured only at test initiation.

The specified temperature range of 15 ± 1°C was exceeded during the acclimation period (exceedance of 2.3°C). There were no exceedances of the specified temperature range during either the amphipod test or the echinoderm embryo test. The specified salinity range of 28 ± 1 ppt for the amphipod test was exceeded during the acclimation period (exceedance of 6.0 ppt) and was often exceeded during testing (exceedance of 0.5–2.5 ppt). The specified salinity range of 30 ± 1 ppt for the echinoderm test was exceeded only once (exceedance of 1.0 ppt). Dissolved oxygen concentrations were less than 5.0 mg/L in some of the *Ulva* treated samples. Dissolved oxygen concentrations in the *Ulva* treated samples ranged from 3.2 to 8.1 mg/L in the amphipod test and from 3.2 to 7.6 mg/L in the echinoderm embryo test. Values for pH ranged from 7.5 to 8.8 in the amphipod test and from 7.3 to 8.6 in the echinoderm embryo test and were all within the recommended range of 7.0–9.0. Ammonia concentrations in sediment pore water in the *Ulva* treated samples ranged from 0 to 36.7 mg/L in the amphipod test and from 4.0 to 16.0 mg/L in the echinoderm embryo test. Sulfide concentrations in sediment pore water in the *Ulva* treated samples ranged from 0 to 65.0 mg/L in the amphipod test and from less than 2.5 to 17.5 mg/L in the echinoderm embryo test.

The negative control consisted of seawater from Yaquina Bay, Oregon. The mean survival value for the negative control in the amphipod test was 100 percent, which exceeds the performance criterion of 90 percent (Ecology 1995). In the echinoderm embryo test, more than 80 percent of the inoculated embryos produced normal pluteus larvae in the seawater control (i.e., negative control).

Because the amphipod test and the echinoderm embryo test were conducted using appropriate protocols, water quality variables were generally within acceptable ranges, and performance criteria were achieved for the negative control and reference area samples, the results are considered acceptable for use in evaluating the toxicity of Ward Cove sediments.

Porewater Aeration Procedure: The recommended protocols were followed closely during testing; however, sediment holding times were exceeded prior to test initiation. Water quality parameters were measured daily in one test replicate of each sample; however, ammonia and sulfide were measured only at test initiation.

The specified temperature range of 15 ± 1°C was exceeded during the acclimation period (exceedance of 2.3°C) and in three replicate test chambers during testing (exceedance of 0.1°C). The specified salinity range of 28 ± 1 ppt for the amphipod test was often exceeded during testing (exceedance of 0.5–2.5 ppt). Concentrations of dissolved oxygen were greater than the recommended minimum level of 5.0 mg/L for all aerated samples, but dissolved oxygen concentrations were lower in the unaerated samples (3.3–8.1 mg/L). Values for pH ranged from 7.8 to 9.0 and were all within the recommended range of 7.0–9.0. Ammonia concentrations ranged from 7.5 to 57.5 mg/L in aerated sediment pore

water and from 7.5 to 62.5 mg/L in unaerated samples. Sulfide concentrations in aerated sediment pore water ranged from 0 to 11.3 mg/L and in unaerated samples from 0 to 130 mg/L.

The negative control consisted of seawater from Yaquina Bay, Oregon. The mean survival value for the negative control was 100 percent, which exceeds the performance criterion of 90 percent (Ecology 1995).

Because the amphipod test was conducted using appropriate protocols, water quality variables were generally within acceptable ranges, and performance criteria were achieved for the negative control and reference area samples, the results are considered acceptable for use in evaluating the toxicity of Ward Cove sediments.

3. PHYSICAL CHARACTERISTICS OF WARD COVE

In this section, the major physical characteristics of Ward Cove and vicinity are described.

3.1 BATHYMETRY AND MAJOR PHYSICAL FEATURES

Ward Cove is an estuary located on the north side of Tongass Narrows, approximately 5 miles (8 km) north of Ketchikan, Alaska. Ward Cove is approximately 1 mile (1.6 km) long and has a maximum width of 0.5 mile (0.8 km). Depths range from –10 ft below MLLW at the head of the Cove (i.e., the northeast portion of the Cove) to –200 ft below MLLW at the mouth (i.e., the southwest portion of the Cove, opening to Tongass Narrows). The shoreline of the Cove is mostly rocky (i.e., basalt) and relatively steep.

Detailed bathymetric and geophysical surveys of Ward Cove were conducted in May 1997. The bathymetric survey was conducted by David Evans and Associates, Inc. The precision bathymetric survey provided a detailed map of the seabed topography throughout Ward Cove. The bathymetric data were used to guide the location of stations in the Phase 2 sediment investigation. The bathymetric data were also used to assist with the remedial alternative analysis, which is discussed in Sections 10 and 11 of this report. Portions of the north shoreline of the Cove are very steep; some areas exceed a 25 percent slope (Figure 3-1), which is a consideration in evaluating potential remedial technologies.

Sediment accumulation is expected to be limited or absent in portions of the Cove that are very steep or are subjected to high current velocities or wave action (and that lack a nearby source of particulate material). As an example, surface sediment was difficult to collect by grab sampling along the steep areas of the north shore (see Figure 3-1), indicating that sediment is sparse or absent on the steep slopes. Several sampling stations along the north shore had to be relocated offshore to locations with accumulated sediment.

The geophysical survey was conducted by Golder Associates Inc. The geophysical survey provided a detailed characterization of the physical features of Ward Cove. The following types of geophysical information were collected concurrently with the bathymetric survey: side-scan sonar data, subbottom profiling data, and seismic reflection data. Video ground-truthing was performed during the geophysical survey to verify the results of the side-scan sonar and subbottom profiling data.

The side-scan sonar data clearly imaged the distribution of logs on the seafloor (Figure 3-2). The concentration of logs varied from more than 500 per 10,000 m^2 (in the center of the Cove) to fewer than 100 per 10,000 m^2 (near the mouth of the Cove). The

number of surface logs in Ward Cove estimated from the distribution shown in Figure 3-2 is as follows:

Log Density	Area (m^2)	Logs/10,000 m^2	Logs
Very high density	73,850 (approx. 18 acres)	500	3,693
High density	186,570 (approx. 46 acres)	400	7,463
Medium density	191,590 (approx. 47 acres)	200	3,832
Low density	290,800 (approx. 72 acres)	50	1,454
			16,442

Evidence from underwater video data indicates that there are multiple layers of logs in the area of the highest concentration. The side-scan sonar data also suggest that there may be numerous partially buried logs in the high concentration area. These partially buried logs, as well as the buried logs that cannot be detected, increase considerably the total number of logs present. There was no evidence of fiber mats on the side-scan sonar data. The only surficial features, other than logs, were exposed bedrock and a mound located offshore of the cannery.

The subbottom profiler and seismic reflection data provided only limited information on the thickness of the sediment. The presence of logs, organic debris, and gas-charged sediment over much of the study area prevented subsurface penetration of the acoustic signal. Where good subsurface information was obtained, the thickness of sediment overlying bedrock ranged from 20 to 30 m (66 to 98 ft).

Sediments in Ward Cove can be divided into two primary classifications: a surface horizon of non-native organic-rich material and a subsurface horizon of native clay. The upper organic-rich material ranges in thickness from undetected to greater than 10 ft, with a typical thickness of about 4 ft. The upper organic sediment layer generally consists of a watery, black, flocculent material with a strong sulfide odor. The upper organic zone also contains varying amounts of wood debris (e.g., wood chips, bark), with a higher percentage of wood present in cores collected near the KPC docks. The lower native sediments consist of olive-green to gray silty clays and clayey silts with imbedded roots, shells, and schist fragments. As noted above, the combined thickness of the organic-rich sediments and native clay sediments ranged from 66 to 98 ft. The physical properties and distribution of non-native sediments are described in greater detail in Section 4.3, *Subsurface Sediments*.

Aggregate of fibrous material originating from the pulping process (i.e., a fiber mat) was not observed during Exponent's Phase 1 and Phase 2 sampling efforts in 1996 and 1997, in ENSR's 1995 solids deposition study, or during 1994 and 1995 sediment sampling for KPC's NPDES monitoring program. A fiber mat was, however, documented by Jones & Stokes Associates, Inc., in their 1988 assessment of Ward Cove. A fiber mat has also been identified and characterized at APC, a similar pulp mill located in Sitka, Alaska. Little published information is available on the origin and conditions that lead to the

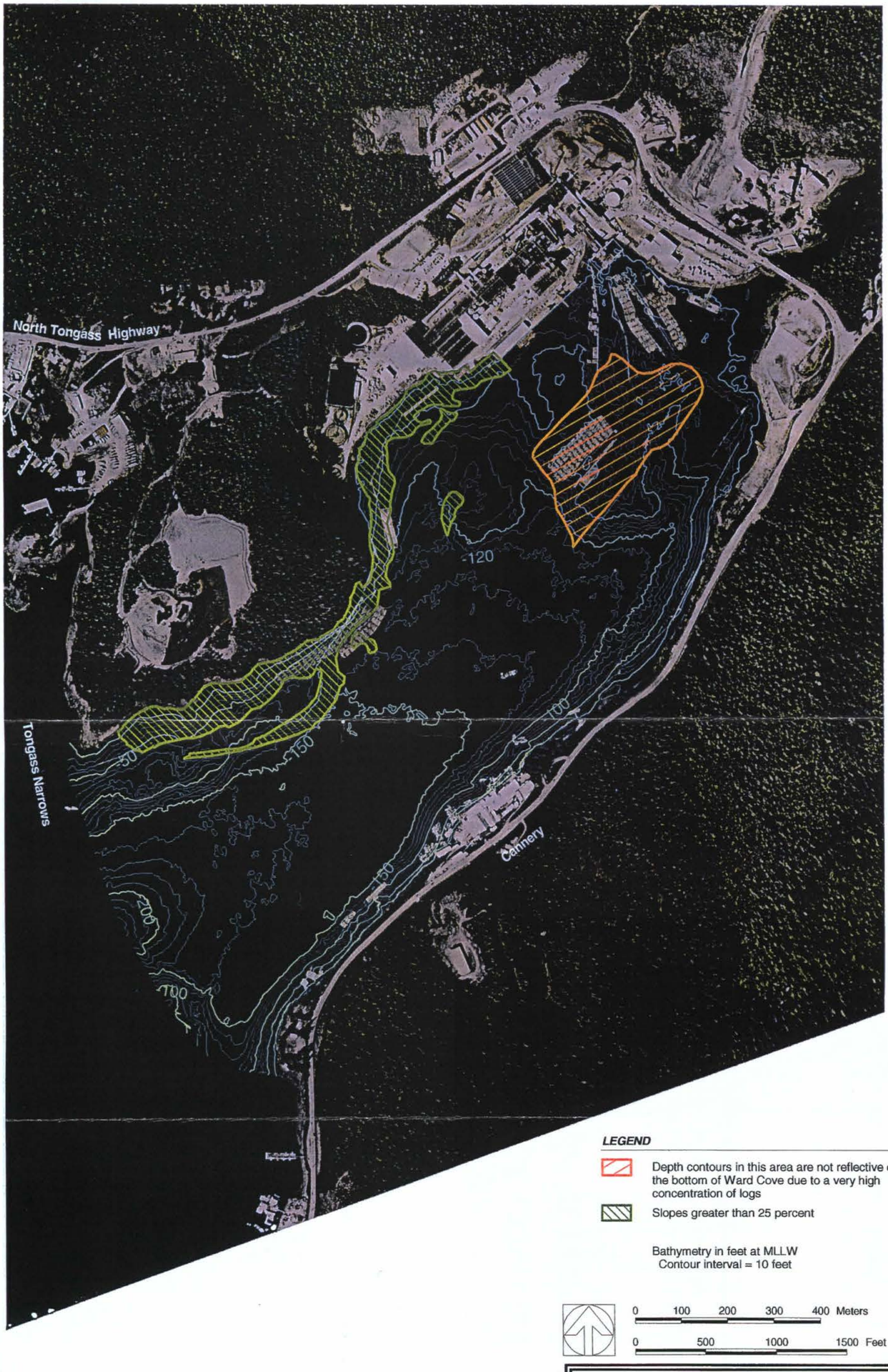

Figure 3-1. Ward Cove bathymetry.

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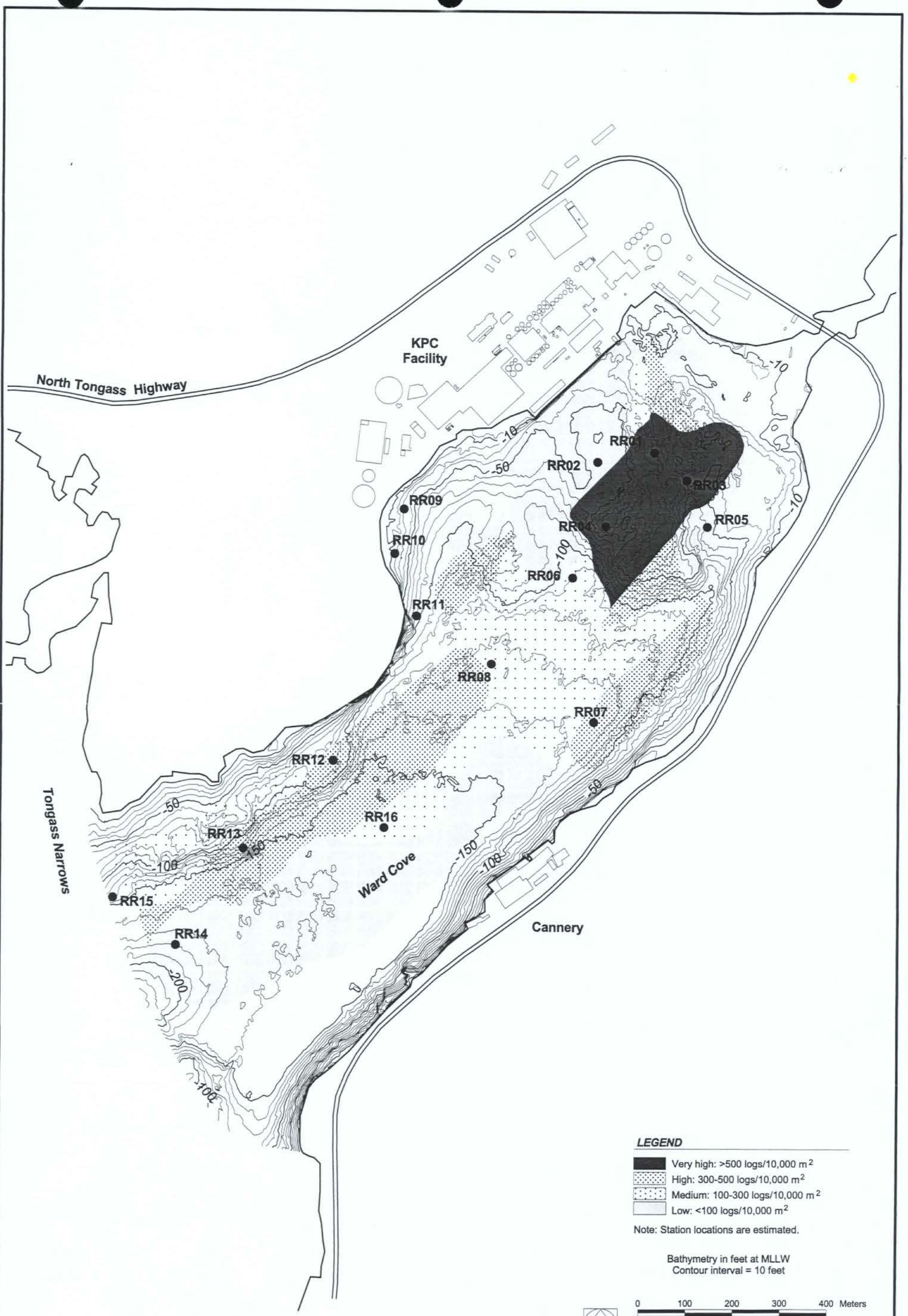


Figure 3-2. Ward Cove log distribution.

8600B0W.001 1602 | May 19,1999 | Figure 3-2. Log Distribution (view) | Figure 3-2. Ward Cove log distribution (layout) | g:\kpc\projects\kpc_rifs.apr

formation of fiber mats in the vicinity of pulp mills. Dredging activities may explain the absence of a fiber mat in Ward Cove. Dan Bodien, EPA's expert on pulp and paper, was questioned regarding potential differences between APC and KPC that could lead to a fiber mat at one facility (APC) and not the other (Keeley 1997c, pers. comm.). He indicated that the pulping processes used at the two facilities are essentially identical and that a fiber mat would be attributable to fibers in the effluent in addition to chips and bark and that conditions in the receiving environment may explain the presence of the fiber mat at APC and the absence of a fiber mat at KPC. Ward Cove is shallow near the mill and required routine dredging, which would have removed fiber prior to any significant mat accumulation. Propeller wash from the use of tug boats at low tide would also break up and disperse accumulated solids. In contrast, the receiving environment next to APC is deeper and dredging was not as frequent, possibly allowing a fiber mat to accumulate.

3.2 METEOROLOGY

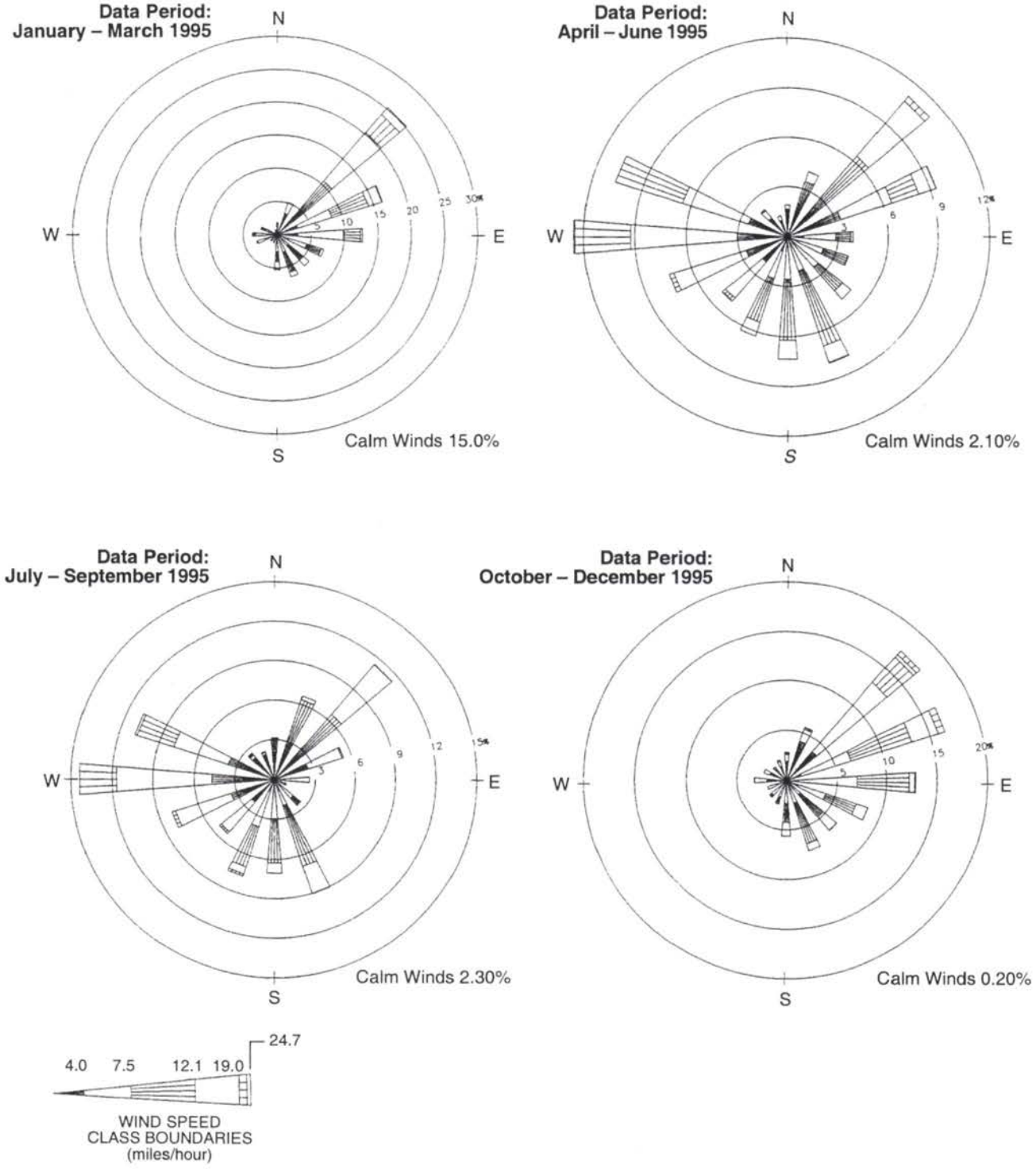
The Ketchikan area has a maritime climate, characterized by relatively mild, wet conditions. The average minimum/maximum January and July temperatures are 29/39°F and 51/65°F, respectively. Ketchikan is one of the wettest locations in the United States, receiving approximately 151 in. of precipitation annually. Actual evapotranspiration is approximately 24 in./year. Winds from the southeast predominate in the Ketchikan area. This direction results from the presence of low pressure cells to the northwest, which draw air in over the Ketchikan area from the Gulf of Alaska and funnel it through the Tongass Narrows. The Tongass Narrows also turns winds from the north into northwesterly winds (Martinson and Kuklok 1977). The meteorological station at the KPC facility provides site-specific information on wind speed and direction. The wind rose diagrams for the four quarters of 1995 (Figure 3-3) indicate that very little wind comes from the north or northwest directions. Depending on the season, local winds come from the west, south, east, and northeast directions. Average wind speeds are about 6 mph during the course of the year.

3.3 SURFACE WATER HYDROLOGY

This section summarizes the upland sources of surface water to Ward Cove and the current and flow of water within Ward Cove.

3.3.1 Upland Sources of Surface Water

Ward Creek is the primary source of fresh water to Ward Cove. Ward Creek, which drops quickly from the nearby mountains to the head of the Cove, is located at the east end of the Cove. Prior to entering the Cove, the creek drains or flows through three small lakes (Lake Perseverance, Connell Lake, and Ward Lake). Discharges from Ward Creek vary widely and respond relatively quickly to the large amount of rainfall that occurs in



Notes: Diagram of the frequency of occurrence for each wind direction. Wind direction is the direction from which the wind is blowing. Example: Wind is blowing from the South 5.2 percent of the time in January – March time period.

Figure 3-3. Wind direction and speed at KPC mill in 1995.

the region. The lower reach of Ward Creek is tidally influenced by currents in Ward Cove. The average flow velocity in the lower portion of the river is approximately 8.3 cm/s.

A small stream (Walsh Creek) flows into Ward Cove along its southeast shoreline. An intermittent stream originates on Dawson Point and discharges into Ward Cove. No perennial streams flow within the boundaries of the pulp mill area.

The three outfalls contributed from 30 to 40 mgd (Jones & Stokes and Kinnetic 1989). Since shutdown of the mill in March 1997, discharges from the outfalls have been greatly reduced.

Stormwater runoff from the pulp mill is another source of surface water to Ward Cove. Runoff from most areas of the pulp mill area is collected and treated in one of three oil–water separators prior to discharge to Ward Cove.

3.3.2 Circulation in Ward Cove

In addition to the geophysical data collected in May 1997, hydrodynamic data were collected in July and August 1997 during Phase 2 sampling. Currents, tidal elevations, and salinity/temperature profiles within Ward Cove were measured to provide data to 1) better assess the potential of sediment transport into Tongass Narrows, 2) improve present knowledge of water circulation within the Cove, and 3) support assessment of the potential for natural recovery of sediment. Current meter arrays (Figure 2-5 in Section 2) provided a continuous record of current velocities in surface water and in deep water at each location for a period of 1 month (July 23–August 23, 1997). The tide gauge records were used to interpret current flow data.

Current meter data indicate that a bilayer flow pattern is present in central and inner parts of Ward Cove. In the main section of the Cove, there is a net outflow in shallow water (to a depth of approximately 50 ft) and a net inflow in deeper water (below about 50 ft). This flow pattern is typical of fjords with a freshwater source at the head. Fresh water is less dense than salt water, and so remains on top when there is no strong mixing impetus. Some mixing occurs in the transition zone between salt and fresh water, resulting in the entrainment of some salt water into the outward flow. Deeper salt water migrates inward to maintain a mass balance of water. During the period of mill operation, when the outfalls discharged a larger volume of fresh water at the shoreline, the surface layer of fresher water is likely to have been deeper than 50 ft.

No bilayer flow or other dominant flow patterns are apparent near the entrance to Ward Cove, where currents were recorded to depths of approximately 150 ft. The lack of a dominant flow regime is likely a result of increased mixing with distance from Ward Cove. Mixing in the outer Cove may also be influenced by the relatively rapid currents in Tongass Narrows; eddies from these currents in the mouth of Ward Cove are likely to obscure weaker flow patterns.

Average flow velocities in Ward Cove vary with depth. In the main section of the Cove, average velocities decrease slightly with depth (2.4 cm/s near the surface to 1.2 cm/s near the bottom). Deep water moves through a greater cross-sectional area than surface water, and so moves more slowly to maintain water balance via bilayer flow. Near the entrance to the Cove, average velocities increase slightly with depth (3.0 cm/s near the surface to 3.4 cm/s near the bottom). Swift currents in Tongass Narrows may be the cause of higher velocities at depth.

The current data confirm that velocities in Ward Cove are low compared to Tongass Narrows. However, the data do not reveal the presence of any areas with little or no flow (i.e., stagnation zones). In combination with the bilayer flow, this observation indicates that dissolved or suspended material introduced into the surface water layer is likely to be transported directly to the mouth of the Cove and then mixed into the flow of Tongass Narrows.

The hydrodynamic modeling conducted by ENSR confirms the presence of bilayer flow and indicates that tidal currents flow through Ward Cove in a counterclockwise direction. During both flood and ebb tides, inward flow occurs fastest along the southern shore of the Cove, and there is a slight outward flow along the northern shore. This flow pattern is illustrated in Figures 3-4 and 3-5. There is a net influx of bottom water during both flood and ebb tides and a net outflow of surface water. The consequence of this flow pattern is that water or suspended solids introduced to the Cove at the surface at any point are expected to be transported out of the Cove along the northern shoreline.

3.4 TOPOGRAPHY AND LAND USE

This section summarizes the topography of Ward Cove and current and future land use. A more complete description of this information, including a comprehensive review of KPC facility operations and materials handling practices, can be found in the scoping document (PTI 1997c).

3.4.1 Topography

Ward Cove is a coastal valley bounded by Slide Ridge to the north and Ward Mountain to the south. The predominant orientation of the valley is southwest to northeast. The area surrounding the Cove is mountainous and largely forested. The pulp mill is located on the north shoreline of Ward Cove and covers approximately 70 acres. To the north of the pulp mill, the terrain slopes steeply upward to a peak at approximately 2,100 ft above mean sea level at a distance of approximately 1 mile from the shoreline. Ward Cove is approximately 1 mile long, has a maximum width of 0.5 mile, and connects to Tongass Narrows to the west. Ward Creek is located at the east end of Ward Cove.

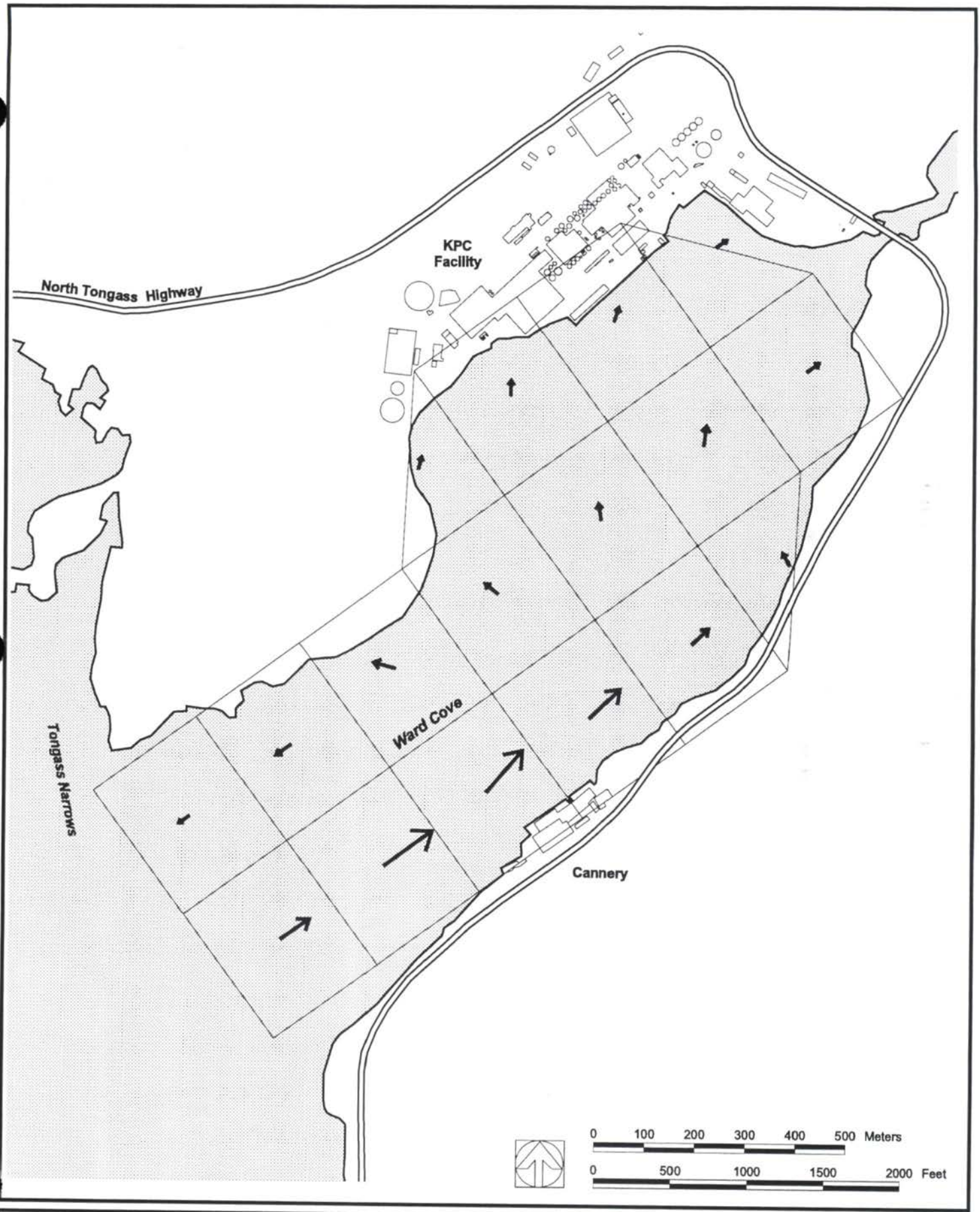


Figure 3-4. Flood tide flow pattern from hydrodynamic modeling.

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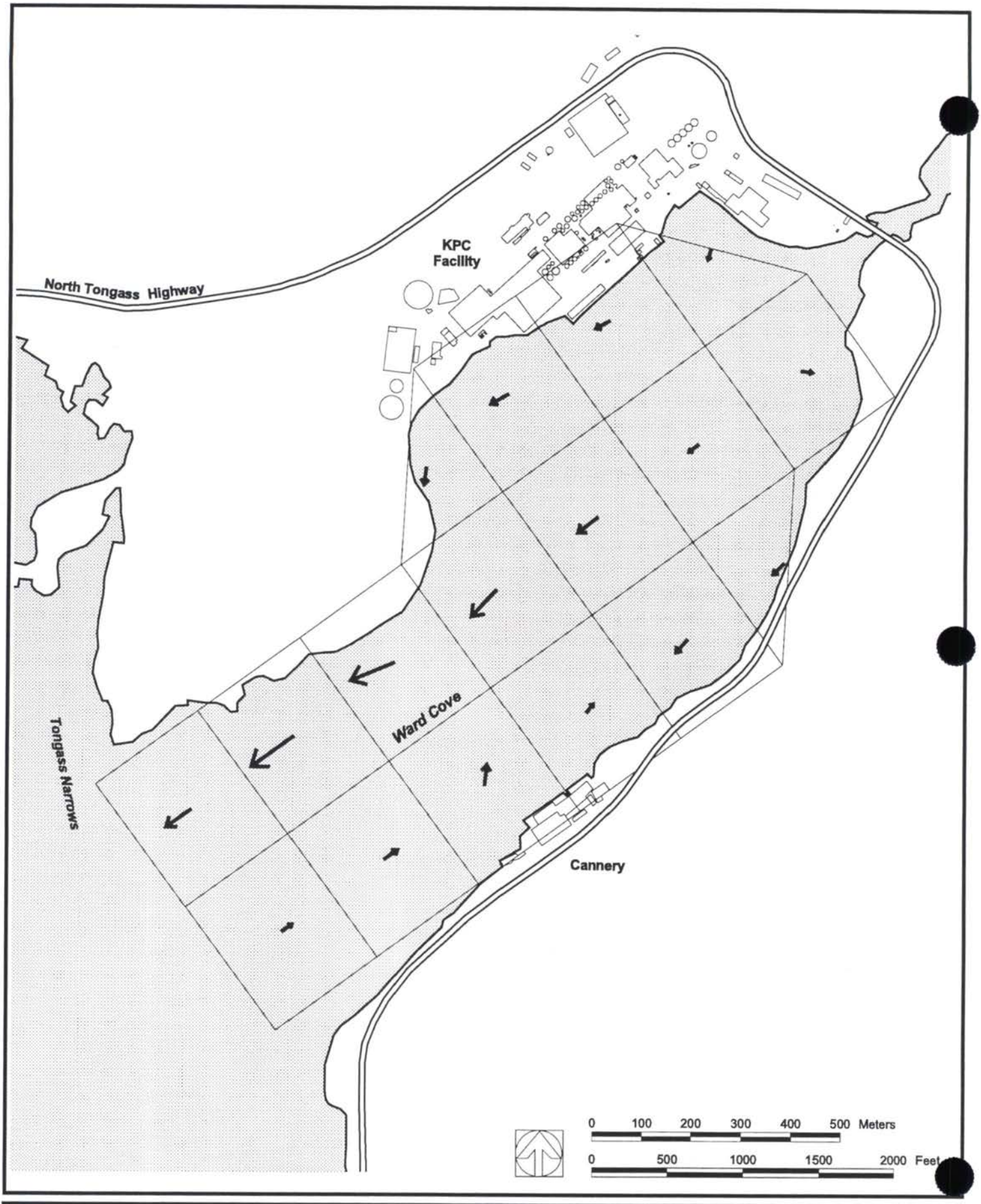


Figure 3-5. Ebb tide flow pattern from hydrodynamic modeling.

3.4.2 Land Use

The pulp mill area is currently used for industrial purposes, and such use is expected to continue in the future. Nearby areas are used for industrial/commercial, residential, and recreational purposes. The locations of residential and commercial buildings in the Ward Cove area are shown in Figure 3-6. More detailed land use maps are available from the Ketchikan Gateway Borough.

3.4.2.1 Industrial/Commercial

Approximately 12 businesses are located immediately across from the mill's water filtration plant north of the North Tongass Highway. These businesses include a tire store, construction company (yard and office), a refuse hauler, a self-service storage facility (mini storage), an auto body shop, and an auto wrecking yard. There are also small commercial properties located near the mouth of Ward Cove and adjacent to Refuge Cove. In addition to the industrial/commercial facilities on the northwest and northeast shorelines of Ward Cove, a fish cannery facility of the Ward Cove Packing Company is located on the southeast shoreline.

3.4.2.2 Residential

Approximately six residences are located immediately north of the pulp mill across the North Tongass Highway. One is located across the highway from the main plant entrance (north of the heliport). Several others are near the businesses located north of the mill's water filtration plant. Steep terrain limits the number of suitable building sites near Ward Cove. Several residences are located near the mouth of Ward Cove. Approximately 1 mile west of the entrance to the pulp mill area, there are additional residences on both sides of the North Tongass Highway (refer to Figure 3-6).

3.4.2.3 Recreational

The area near the mouth of Ward Creek (where the Tongass Highway crosses the creek) is a popular fishing location, especially during salmon season. The intertidal area around some parts of Ward Cove would be accessible during low-tide periods.

3.5 ECOLOGY

This section summarizes available information on the ecology of the aquatic habitats in both Ward Cove and Ward Creek and the terrestrial habitat in upland areas surrounding Ward Cove. The information is based on previous published studies of the area (ENSR 1995d; Jones & Stokes and Kinnetic 1989; Martinson and Kuklok 1977; Spannagel 1991), a preliminary ecological reconnaissance in February 1997 and a subsequent reconnaissance in July 1997 by an Exponent ecologist and representatives from EPA and

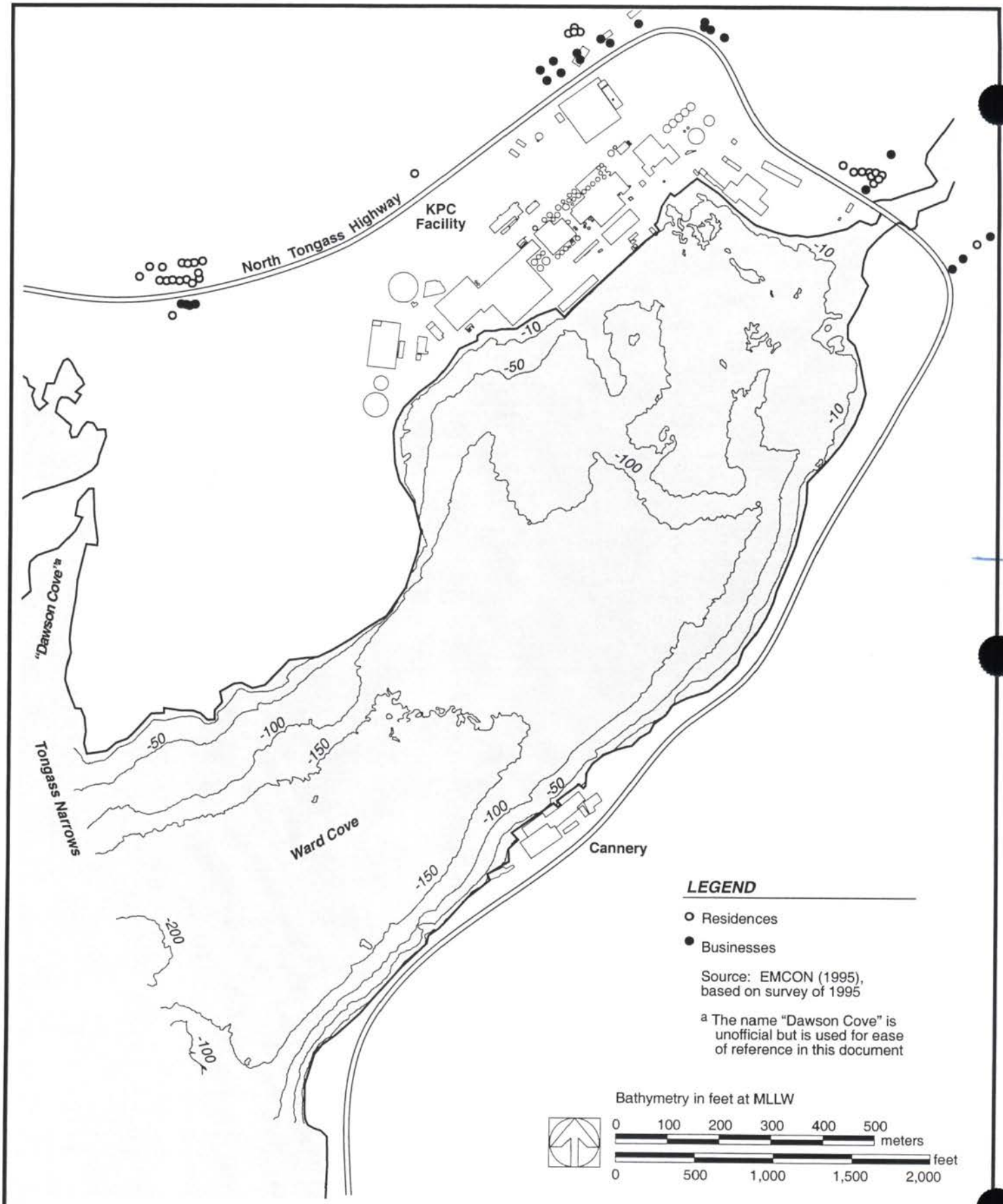


Figure 3-6. Locations of residential and commercial buildings in the Ward Cove area.

the USFWS, and personal communications with individuals having direct experience in the area (Crook 1995, pers. comm.).

3.5.1 Aquatic Habitat in Ward Cove

Ward Cove represents an embayment of Tongass Narrows and is characterized by an estuarine circulation pattern with saline waters at depth and less saline waters near the surface. The tidal range in Ward Cove is large and consequently currents are largely tidally driven. Benthic habitats of Ward Cove include both areas of soft substrate and areas of hard substrate (i.e., exposed rock). Soft substrates consist of varying percentages of sand, silt, and clay. No sensitive aquatic habitats have been identified in Ward Cove.

3.5.2 Ecological Receptors in Ward Cove

Although there have been no comprehensive surveys of ecological receptors in Ward Cove, ENSR (1995d) developed lists of potential resident species in the adjacent portion of Tongass Narrows as part of the ecological risk assessment for the proposed extension of KPC Outfall 001 to the center of Tongass Narrows. Because Ward Cove connects directly with Tongass Narrows, many of the species identified by ENSR (1995a) may be found in the Cove on either a continuous or intermittent basis.

As a result of the high rate of tidal exchange of water in Ward Cove with the adjacent Tongass Narrows, organisms inhabiting the water column (e.g., phytoplankton, zooplankton) are only transient residents of Ward Cove.

In undisturbed areas of soft benthic substrates in Tongass Narrows, benthic macroinvertebrate communities are dominated by filter-feeding and surface deposit-feeding polychaetes (e.g., *Prionospio* spp.) and bivalves (e.g., *Axinopsida* spp.), as well as by burrowing polychaetes (Lumbrineridae) (ENSR 1995d). Similar communities would be expected to occur in undisturbed, soft-bottom habitats within Ward Cove. Large epibenthic invertebrates likely to be found in such habitats include several species of crabs (notably Dungeness crabs, *Cancer magister*) and several species of shrimp.

Hard substrate benthic habitats of Ward Cove, especially along the shoreline, are characterized by the presence of mussels, barnacles, sea urchins, sea anemones, sea cucumbers, starfish, and brown and green algae (Sexton 1997, pers. comm.).

An extensive list of fish species potentially occurring in Tongass Narrows is presented in Table 15 of PTI (1996). From a commercial or recreational standpoint, the most important group of fishes is the salmonids, seven species of which are known to migrate through Ward Cove as adults to spawn in the Ward Creek watershed and then to outmigrate through the Cove as juveniles. These anadromous species include sea-run cutthroat trout (*Oncorhynchus clarki clarki*), pink salmon (*Oncorhynchus gorbuscha*), chum salmon (*Oncorhynchus keta*), coho salmon (*Oncorhynchus kisutch*), rainbow trout (*Oncorhynchus mykiss*), sockeye salmon (*Oncorhynchus nerka*), and Dolly Varden

(*Salvelinus malma*). Other important commercial or recreational fish species likely to occur in Ward Cove include Pacific herring (*Clupea harengus pallasi*) and various species of flatfishes and rockfishes.

Lists of the bird and marine mammal species potentially occurring in Tongass Narrows are presented in Tables 16 and 17, respectively, of PTI (1996). Although all of the bird species listed may occur in Ward Cove as well, frequent inhabitants are known to include bald eagles (*Haliaeetus leucocephalus*), common mergansers (*Mergus merganser*), gulls (*Laurus* spp.), pelagic cormorants (*Phalacrocorax pelagicus*), surf scoters (*Melanitta perspicillata*), belted kingfishers (*Ceryle alcyon*), and great blue herons (*Ardea herodias*). Similarly, all of the listed marine mammals may occur in Ward Cove, but the most common species observed in Ward Cove are harbor seals.

3.5.3 Aquatic Habitat in Ward Creek

Downcurrent portions of Ward Creek (i.e., the reach near the mouth of the creek) are considered by the ADFG to be sensitive or critical habitats for salmon spawning, rearing, and migration (PTI 1997h). However, data collected during the remedial investigation for the Uplands Operable Unit suggest that this area has not been affected by operations at the KPC site (PTI 1997c). Furthermore, observations made during the February 1997 reconnaissance suggest that the lower reaches of Ward Creek are unlikely to serve as salmon spawning habitat. Salmon spawning likely occurs in more upcurrent portions of the Ward Creek watershed.

The lower reaches of Ward Creek are tidally influenced, with water levels varying at least 8–10 ft. The waters of lower Ward Creek may at times be brackish or even saline depending on tide levels and flow conditions in the upper portions of Ward Creek. High tide lines are visible in the forest vegetation and trees surrounding the creek channel. Shorelines near the mouth of Ward Creek are rocky and covered with barnacles, mussels, and kelp. The initial slope of the shoreline (e.g., upper bank) is often steep and at times almost vertical. This characteristic is particularly true at the mouth of Ward Creek where the shorelines are often developed from dredge spoils and rip-rap.

About 220 yd upcurrent of the North Tongass Highway bridge is a riffle/pool area where the upcurrent portions of Ward Creek empty into a relatively flat, broad stream channel that is tidally influenced. Upcurrent of this riffle/pool area, Ward Creek flows through a narrow channel with steeply sloped (about 60 percent grade), heavily forested hillsides, resulting in a high volume, turbulent flow. Two small unnamed tributary streams enter Ward Creek in the riffle/pool area.

Several intermittent and permanent streams drain Slide Ridge, entering Ward Creek both upcurrent and downcurrent of Ward Lake. Most of the smaller streams are not indicated on the U.S. Geological Survey topographic maps. Nearly all of these smaller streams are undisturbed, having high-gradient flows of clear water, comprising numerous small pools interconnected by small cascades and waterfalls. The creek channel's substrates are

bedrock, often covered with various types of mosses and lichens and typically filled with downed limbs and branches.

Riparian zones are absent along the lower portions of Ward Creek (i.e., immediately upcurrent of Ward Cove) because this portion of the creek is subjected to substantial tidal fluctuation and the shorelines and stream bottom are extremely rocky and covered with barnacles, mussels, and kelp.

3.5.4 Ecological Receptors in Ward Creek

Likely the most important ecological receptors in Ward Creek are the seven species of anadromous salmonids whose adults move into the creek from marine waters during the breeding season to spawn in headwater portions of the creek, generally above Ward Lake. Depending on the species, the juvenile salmonids spend varying lengths of time in the creek before migrating to marine waters.

An important mammal species known to inhabit both Ward Creek and Ward Cove is the river otter (*Lutra canadensis*).

3.5.5 Terrestrial Habitat of the Upland Areas near Ward Cove

Ward Cove is a coastal valley bounded by Slide Ridge to the north and Ward Mountain to the south. The predominant orientation of the valley is southwest to northeast. To the north of the pulp mill area, the terrain slopes steeply upward to a peak at approximately 2,100 ft above mean sea level at a distance of approximately 1 mile from the shoreline. The area surrounding the pulp mill area is largely forested. The shoreline of Ward Cove on the south boundary of the pulp mill area is steep. Ward Creek, located at the east end of Ward Cove, is the primary source of fresh water to the Cove.

An ecological reconnaissance of terrestrial habitats of the Uplands Operable Unit was conducted by an Exponent ecologist on February 16 and 17, 1997. Observations during that reconnaissance formed the basis for the information summarized below.

The vegetation community of the upland areas surrounding Ward Cove is predominantly temperate, coniferous rainforest habitat, dominated by western hemlock (*Tsuga heterophylla*) and western red cedar (*Thuja plicata*), with components of Sitka spruce (*Picea sitchensis*) and shore pine (*Pinus contorta*). Forested habitats on Dawson Point and Slide Ridge are second growth forest, having regenerated after a fire approximately 50 to 70 years ago. This estimate is based on a visual inspection of the area during the February 1997 site visit, review of historical aerial photographs, and discussion with U.S. Forest Service personnel. Western red cedar is a very durable wood and thus dead trees and snags may remain standing for many years. There are many such snags and standing dead cedar trees on Slide Ridge that remained standing after the fire. These snags and trees provide excellent wildlife habitat.

Understory vegetation is primarily mosses, ferns, and fungi, with scattered vascular plants. Disturbed areas along roadways and small creeks are vegetated with early successional species such as red alder, Sitka alder, and shrub species. Moderately shallow muskeg soils are typical of the region. With the exception of those areas developed for commercial and industrial purposes, the Ward Creek watershed appears to offer relatively high quality habitat for wildlife, although limited in extent by the topography surrounding Ward Cove.

No sensitive habitats were observed in the areas surveyed, including the pulp mill area, the wood waste and ash disposal landfill, and adjacent forestland. The pulp mill area is a highly industrialized landscape, and adjacent areas are mixed commercial businesses and residences. Wildlife habitat is not present at the pulp mill area; only a few scattered, disjunct areas of vegetation (i.e., grasses, forbs) are found near storage tanks or along portions of the steeply sloped shorelines of Ward Cove. No sensitive or critical habitats were observed on either Dawson Point or Slide Ridge.

3.5.6 Terrestrial Ecological Receptors

Incidental observations during the February 1997 reconnaissance indicate the presence of a relatively diverse winter fauna. In addition to the bird species commonly observed in Ward Cove, species observed during the reconnaissance of the upland areas included black-billed magpies (*Pica pica*) and black-capped chickadees (*Ardea herodias*) along Ward Creek and its tributaries. KPC employees indicate that deer are relatively common in the area, and, although none were observed during the reconnaissance, deer tracks were present.

Plant, mammal, and bird species potentially occurring in terrestrial habitats of the Ward Cove area are listed in Tables 3-1 through 3-3, respectively. These lists were developed from the *Atlas of the Ketchikan Region* (Martinson and Kuklok 1977).

TABLE 3-1. VEGETATION POTENTIALLY OCCURRING ON THE UPLANDS AREA NEAR WARD COVE

Common Name
Trees
Alaska cedar
Douglas maple
Lodgepole pine
Mountain hemlock
Pacific silver fir
Pacific yew
Red alder
Sitka alder
Sitka spruce
Western crabapple
Western hemlock
Western redcedar
Willow
Yellow cedar
Understory Vegetation
Mosses
Club moss
Sphagnum moss
Spike mosses
Ferns
Beechfern
Bracken fern
Deer fern
Fragile fern
Lady fern
Maidenhair fern
Oak fern
Spreading fern
Triangular wood fern
Western swordfern

Common Name
Shrubs
Alaska blueberry
Baneberry or snakeberry
Bog blueberry
Bog cranberry
Bristly black currant
Crab apple
Devil's club
Douglas spiraea
Dwarf blueberry
Early blueberry
Fatty-leaved willow
Goatsbeard
High bush cranberry
Mountain cranberry
Nootka rose
Pacific ninebark
Pacific red elder
Pacific serviceberry
Red elderberry
Red huckleberry
Rusty menziesia
Salal
Salmonberry
Shore pine
Sitka willow
Stink currant
Thimbleberry
Trailing black currant
Monocots
Arrow grass
Beach ryegrass

TABLE 3-1. (cont.)

Common Name	Common Name
Blue eyed grass	Fescue grass
Red fescue	Fireweed
Reed canarygrass	Foamflower
Squirreltail grass	Goosetongue
Vanilla grass	Heart-leaved twyblade
Woodrushes	Hemlock parsley
Forbs	Horsetails
Alaska violet	Indian rice or black lily
Beach asparagus	Kruhsea
Beach greens	Lace flower
Beach lovage	Large-leaved avens
Beach pea	Locoweed
Beach strawberry	Lupine
Beach strawberry	Lyall
Bedstraw	Lyngbye sedge
Bent-leaved angelica	Manna grass
Bluejoint	Mare's tale
Broad leaved marigold	Monkshood
Bunchberry	Mountain hairgrass
Buttercups	Northern geranium
Chickweed	Orange hawkweed
Cleavers	Pacific twinflower
Copperbush	Piggyback
Coral root	Poison water hemlock
Cow parsnip	Reed bentgrass
Cow parsnip or Wild celery	Salad greens
Cowslip	Scurveygrass
Cranesbill	Sea milkwort
Dandelion	Seabeach sandwort
Deerberry	Seashore plantain
Dwarf fireweed	Shooting star
Eelgrass	Siberian spring beauty
Eschsholz false hellebore	Silverweed
False Lily of the Valley	Simple-stemmed twisted stalk

TABLE 3-1. (cont.)

Common Name	Common Name
Single-flowered clintonia	Wild cucumber or clasp leaf twisted stalk
Sitka burnet	Wild iris
Skunk cabbage	Wintercress
Small bedstraw	Yarrow
Starflower	Yellow marsh marigold
Stinging nettle	Yellow monkeyflower
Tiarella	Yellow paintbrush
Vetch	Yellow skunk cabbage
Villous cinquefoil	Yellow violet
Western columbine	Youth-on-age
White water crowfoot	

Note: The data on potential species occurrences presented in this table were developed from Martinson and Kuklok (1977).

TABLE 3-2. MAMMALS POTENTIALLY OCCURRING ON THE UPLANDS AREA NEAR WARD COVE

Common Name	Common Name
Beaver	Mink
Black bear	Muskrat
Deer mouse	Northern water shrew
Dusky shrew	Porcupine
Flying squirrel	Red squirrel
Land otter	Redback vole
Little brown myotis	River otter
Long-legged myotis	Short-tailed weasel
Long-tailed vole	Sitka black-tailed deer
Marten	Tundra vole
Masked shrew	Wolf
Meadow vole	

Note: The data on potential species occurrences presented in this table were developed from Martinson and Kuklok (1977).

TABLE 3-3. BIRDS POTENTIALLY OCCURRING ON THE UPLANDS AREA NEAR WARD COVE

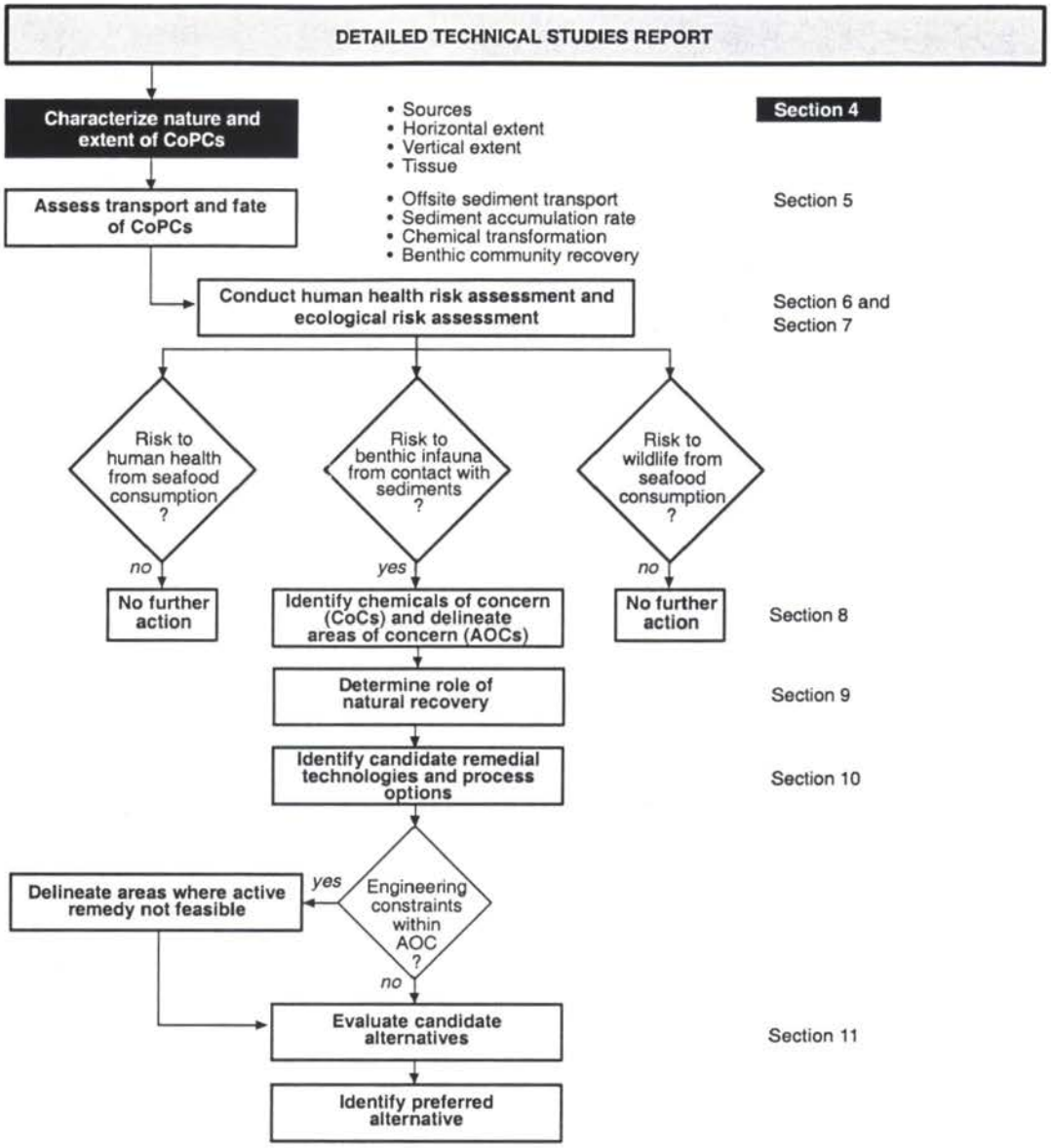
Common Name	Common Name
American robin	Great gray owl
American wigeon	Greater scaup
Arctic loon	Greater yellowlegs
Arctic tern	Great-horned owl
Bald eagle	Green-winged teal
Barrow's goldeneye	Hairy woodpecker
Belted kingfisher	Harlequin duck
Black scoter	Hermit thrush
Black turnstone	Herring gull
Blue grouse	Killdeer
Blue-winged teal	Lesser yellowlegs
Bonapart's gull	Lincoln's sparrow
Bufflehead	Mallard
Canada goose	Marbled murrelet
Chestnut-backed chickadee	Marsh hawk
Common goldeneye	Mew gull
Common loon	Northern phalarope
Common merganser	Northwestern crow
Common murre	Oldsquaw
Common raven	Orange-crowned warbler
Common snipe	Oregon junco
Dark-eyed junco	Pectoral sandpiper
Dipper or water ouzel	Pelagic cormorant
Double-crested cormorant	Pigeon guillemot
Downy woodpecker	Pine grosbeak
Dunlin	Pine siskin
Fox sparrow	Pintail
Glaucous-winged gull	Red breasted merganser
Golden-crowned kinglet	Red-necked grebe
Goshawk	Red-throated loon
Gray-crowned rosy finch	Ree crossbill
Great blue heron	Rock sandpiper

TABLE 3-3. (cont.)

Common Name	Common Name
Rock sandpiper	Swainson's thrush
Ruby-crowned kinglet	Thayer's gull
Rufous hummingbird	Tree sparrow
Rusty blackbird	Tree swallow
Savannah sparrow	Varied thrush
Semipalmated plover	Violet-green swallow
Sharp-shinned hawk	Water pipit
Short-billed dowitcher	Western flycatcher
Short-eared owl	Western sandpiper
Shoveler	Whistling swan
Snow goose	White-fronted goose
Sparrow hawk	White-winged scoter
Spotted sandpiper	Wilson's warbler
Steller's jay	Winter wren
Surf scoter	Yellow-rumped or Myrtle warbler

Note: The data on potential species occurrences presented in this table were developed from Martinson and Kuklok (1977).

4. NATURE AND EXTENT OF CHEMICALS OF POTENTIAL CONCERN



The distribution of CoPCs in Ward Cove sediments is the primary focus of this section. An initial discussion of potential sources is provided to better establish potential linkages between activities at the KPC facility and the distribution of chemicals in sediments. Tissue data for selected chemicals are also described. A synthesis of NPDES water column data is provided to address concerns related to potential oxygen depletion in surface water.

The following CoPCs have been identified for sediments:

- **Substances Associated with Organic Matter and Organic Matter Degradation**—TOC, ammonia, sulfide, BOD, COD, phenol, and 4-methylphenol
- **Metals**—arsenic, cadmium, mercury, and zinc
- **Organic Compounds**—PAHs and dioxins and furans.

TOC, BOD, and COD are included on the list because they are useful measurements of the magnitude and nature of organic matter content; ammonia and sulfide are considered to be the causative agents for sediment toxicity (Section 7). All of these substances were eliminated as CoPCs for human health and food-web concerns as part of the initial human health risk and ecological evaluations (PTI 1997g). All of these chemicals are included in the assessment of the nature and extent of CoPCs.

4.1 POTENTIAL SOURCES OF CHEMICALS TO WARD COVE

Potential CoPC sources from KPC activities to Ward Cove have been investigated in several previous studies (PTI 1997c, Kennedy/Jenks 1997). The following potential sources have been identified:

- KPC wastewater treatment discharges
- Log handling practices (in-water log rafting)
- Wood waste and ash disposal landfill
- Near-shore fill subarea (including surface water runoff and groundwater discharge)
- Wood waste and sludge disposal subarea (including surface water runoff and groundwater discharge)
- Groundwater seeps
- Dredge spoil subarea
- Storm water discharges
- Release of airborne contaminants from the power boilers
- Spills and accidental releases.

In addition to the above potential sources associated with KPC activities, a fish cannery located on the south side of Ward Cove to the southwest of KPC has also been identified as a source of CoPCs to the Cove. All of these sources except storm water discharges,

aerial deposition, and spills are shown in Figure 4-1. A chemical-specific synthesis of source information is provided in Section 1.2.

Of the potential sources associated with the KPC facility, the wastewater treatment discharges and wood handling (in-water rafting) are considered the most significant sources of chemicals and organic material to Ward Cove. Furthermore, it is likely that historical discharges from the wastewater outfalls are responsible for the high concentrations of fine-grained organic matter, which in turn are associated with the high concentrations of ammonia, sulfide, and 4-methylphenol. Fine-grained organic matter has a much higher surface-to-volume ratio than larger pieces of bark or wood, creating a much more reactive surface for microbial activity. Microbial activity is the source of these by-products (i.e., ammonia, sulfide, and 4-methylphenol) of organic matter degradation. Each of the sources and the status of source characterization investigations in relation to them are described in this section. Potential sources and transport mechanisms that may impact upland soils, but that are not potentially significant sources of contaminants to Ward Cove, have also been evaluated (PTI 1997c; Kennedy/Jenks 1997) but are not included in this discussion. These sources will be further evaluated in the uplands remedial investigation.

4.1.1 Wastewater Discharges

Historically, KPC has discharged an average of 38–45 mgd of wastewater to Ward Cove through several outfalls: 001, 002, 003, and 004 (Figure 1-2; Jones & Stokes and Kinnetic 1989; Hayes 1998, pers. comm.). From 1954 until 1972, wastewater was discharged at the shoreline to Ward Cove through four separate outfalls. Untreated wastewater primarily from the acid plant, wash plant, bleach plant, and machine room was discharged through the main outfall (001), which was located west of the No. 1 warehouse. Partially treated wastewater from the boiler house was discharged through Outfall 002. Wastewater generated in the wood rooms passed through North rotary screens and was discharged, via the hog house, through Outfall 003. Sediments and filter backwash from the water treatment plant were discharged through Outfall 004. The outfalls were located progressively from west to east. The primary treatment facility, which included a vacuum filter, a "V" press, and a grit chamber, was constructed in 1971 to reduce discharges of suspended solids. As a result, Outfall 003 was eliminated in 1972 and routed to primary treatment. Wastewater from the primary treatment facility was discharged from a separate outfall. In 1972, Outfall 002 was eliminated by rerouting to the main outfall. At this time, outfall numbers were redesignated. The main outfall (001) remained the same. The primary discharge was designated 002 and the water treatment plant became 003. The secondary activated sludge treatment system was installed in 1980 to reduce BOD discharges and included an aeration basin and a secondary clarifier. Primary and secondary effluents were combined and discharged through a newly constructed outfall separate from the main outfall. In 1993, the effluent neutralization system was installed to combine all process discharges and control pH of the combined discharge. This discharge was designated as Outfall 001 and the water treatment plant outfall became 002. Discharge of pulping waste ended with the shutdown of the pulp mill; however, the powerhouse remained active until March 1998, and the sawmill is still

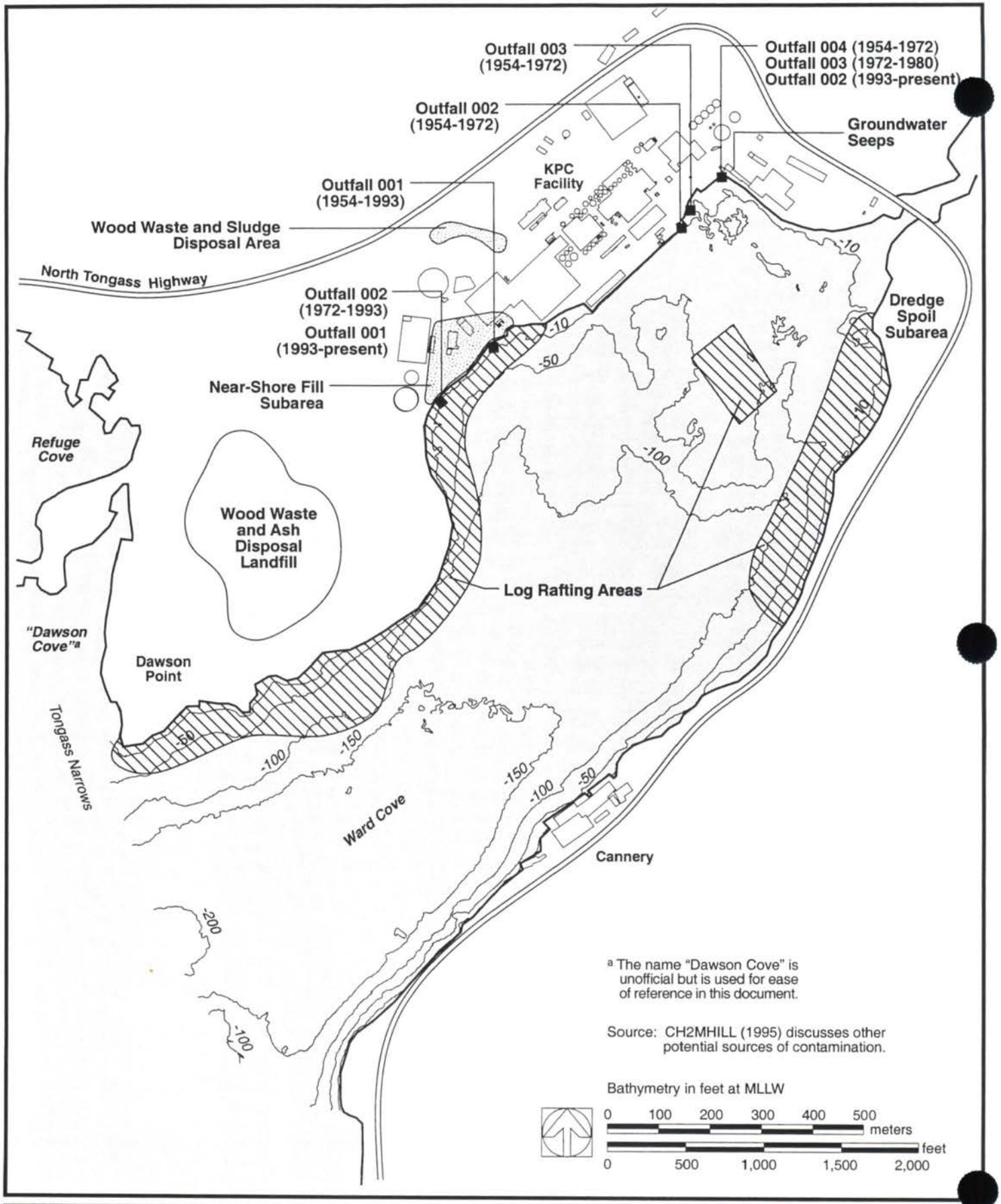


Figure 4-1. Potential sources of CoPCs to Ward Cove.

active. Outfall 001 currently discharges approximately 2–3 mgd of water to preserve a pipeline constructed of wood staves. This pipeline formerly provided process water to the pulp mill. Outfall 002 discharges a small amount of natural influent water from the water supply plant. Impacts of these wastewater discharges on sediments and biota are being evaluated in this Ward Cove investigation.

4.1.2 Log Rafting

Rafting of logs in the Cove has been used for many years as a method for storing logs prior to processing in the mill. Over the years, log rafting has been a source of wood debris to the sediments, including bark and whole logs. Log rafting continues as part of routine operation of the sawmill. Log rafting was conducted primarily at the locations shown in Figure 4-1. Logs were not rafted north of Dawson Point and cannot occur in this area because of the absence of pilings and anchors (Maloy 1998, pers. comm.). In general, pilings and anchors are used on the northern and southern shorelines of Ward Cove for log rafting. The natural degradation of organic matter by microorganisms consumes oxygen and produces ammonia and sulfide, which have been detected at elevated levels in Cove sediments. The degradation of wood-derived organic matter is also known to produce phenols and methylated phenols (Sjöström 1981), also identified as CoPCs in Cove sediments. The impacts of this organic-rich matter on sediments and biota are being evaluated in this Ward Cove investigation.

4.1.3 Remaining Potential Sources

Additional potential sources of CoPCs to Ward Cove are described below. Several of these potential sources are being further evaluated at this time to confirm that they are not significant sources of CoPCs to Ward Cove. These ongoing activities are discussed where applicable.

4.1.3.1 Wood Waste and Ash Disposal Landfill

The wood waste and ash disposal landfill was constructed in 1988 in an area west of the mill on Dawson Point (Figure 4-1). The landfill received boiler ash, flyash, wood waste, and small quantities of other materials including calcium filtrate, primary and secondary sludge, and dredge spoils from construction projects at the mill. The landfill is constructed without a bottom liner. The ash and wood waste were placed in separate areas, and the ash disposal and wood waste disposal areas were expanded in 1991 and 1992, respectively. KPC closed the wood waste and ash disposal landfill in the fall of 1997 under a landfill closure plan submitted to ADEC in May 1997 (Woodward-Clyde 1997). A new lined landfill was constructed and became operational in October 1997.

In the summer of 1997, Exponent, EPA, and ADEC conducted a site visit at the landfill to determine if ecological habitat was present in any streams that could potentially be affected by the landfill. It was determined that the only area that may have been affected,

could provide habitat, and can reasonably be accessed by human visitors is the intertidal area in Dawson Cove, which is the discharge point of a stream that has received runoff from the leachate pond.

Leachate from the landfill drains to a leachate treatment lagoon located on the south side of the landfill. During high rainfall periods in the past, surface water and sediment runoff could have washed CoPCs from the landfill to Ward Cove via small streams on Dawson Point. Since 1990, however, leachate has been collected and pumped to the secondary treatment plant and thus there is no ongoing release to Ward Cove. Leachate from both the wood waste and ash landfill cells, as well surface water from six stations in the ditches around the landfill, is periodically monitored for metals, organic compounds, and conventional parameters. As would be expected from a wood waste landfill, analytes such as temperature, COD, color, manganese, and iron are found at levels above background; however, these parameters are within landfill permit guidelines. Toluene was occasionally detected in surface water in the past at one station and is believed to have been the result of cleaning the backhoe used to move the landfill contents; this practice has been discontinued. 2,3,7,8-TCDD has been detected in 2 of 50 sampling events at concentrations well below the drinking water maximum contaminant level of 50 pg/L. In addition to routine monitoring data, sediment samples were collected during the Uplands Operable Unit remedial investigation from an intertidal area at the mouth of one of the ditches that drain the landfill area. Concentrations of all target analytes were below their respective risk-based concentrations. Current and future surface water discharges from the landfill area will be prevented as a condition of landfill closure.

4.1.3.2 Near-Shore Fill Subarea

The near-shore fill subarea is located west of the No. 3 warehouse adjacent to Ward Cove (Figure 4-1). This area was used for disposal of all solid waste generated at the plant (except food waste, which was hauled to a municipal landfill) and bottom ash prior to the late 1970s. Waste was deposited along the bank or burned in this area. Most of the fill used in this area is soil and rock excavated for the construction of the wastewater treatment plant. Shipping pallets and other wood debris were burned in this area until the early 1980s. The near-shore fill subarea is now covered with clean soil.

As described in the uplands work plan (PTI 1997h), seven test pits were excavated in this area, and 11 soil samples and 3 water samples were collected and analyzed for a wide range of chemicals. Several metals were detected above background concentrations in both surface and subsurface soil samples; however, arsenic was the only metal to exceed risk-based concentrations. As discussed in detail in the remedial investigation report for the Uplands Operable Unit, the elevated concentrations of arsenic found at the near-shore fill subarea (and at other locations both at the facility and in local residential and commercial areas) are most likely due to the extensive use of arsenic-rich crushed rock. Concentrations of all metals (dissolved) were at or below background. Low concentrations of several organic compounds were detected in the soil and water samples, but only Aroclor® 1254 was detected above its risk-based concentration, in one subsurface soil sample and in one unfiltered water sample. Because of the high affinity of PCBs to sorb onto

particulate matter, very little of the Aroclor® 1254 detected in the unfiltered water sample would be present in the dissolved phase. Particulate transport through the fill to Ward Cove is not likely to occur, and thus the expected concentrations of Aroclor® 1254 at the groundwater interface to Ward Cove are not considered to be significant. During high rainfall events, surface water runoff from limited portions of the near-shore fill subarea to Ward Cove has been observed, the result of some of the storm water grates being located above grade. Because surface soil concentrations of both metals and organic compounds are not elevated (with the exception of arsenic), this surface water runoff is not of significance to Ward Cove. As part of facility redevelopment, however, the storm water grates will be lowered to the soil surface so that surface water will be captured.

4.1.3.3 Wood Waste and Sludge Disposal Subarea

The wood waste and sludge disposal subarea is located down slope from the west parking area (Figure 4-1). Primary wastewater treatment plant sludge and hog fuel were placed in this area. Except for the wooded slope on the southern edge of the area, the remainder of the area was covered with rock (clean fill) and is now used for parking.

A composite soil sample was collected from the wood waste and sludge disposal subarea during the uplands remedial investigation. Although arsenic was detected at a slightly elevated concentration, the distance and topography from the subarea to Ward Cove and the relatively low concentration of arsenic in the small area of exposed sludge suggests that the subarea is not a source of contamination to Ward Cove.

4.1.3.4 Groundwater Seeps

During the remedial investigation for the Uplands Operable Unit, a survey of the shoreline of the facility and Dawson Point was conducted to identify intertidal seeps that discharge groundwater to Ward Cove. Three seeps were observed near the log deck (Figure 4-1). One of these seeps appears to be the result of a void in the backfill that fills with water during high tide. The two other seeps, located just east of the log deck approximately 30 ft apart, discharge groundwater at a rate of 2–10 gpm. One of the seeps was sampled during the remedial investigation. The only target analyte detected above marine background concentrations was manganese, which is most likely due to the mobilization of soil manganese by organic acids that have leached from the large piles of wood waste and hog fuel that were stored in that area of the facility. With the removal of the hog fuel and wood waste piles as part of mill shutdown, the manganese concentration in the seep is expected to decrease to ambient levels.

4.1.3.5 Dredge Spoil Subarea

The dredge spoil subarea is on the east/southeast shoreline of Ward Cove across from the mill, just south of the mouth of Ward Creek (Figure 4-1). This area has been used historically to drain and dispose dredge spoils. Dredge spoils could contain CoPCs from

process effluent, surface runoff, and other discharges to the Cove. CoPC binding to solids is expected to be significant in the dredge spoil subarea. Downward leaching of CoPCs to the shallow water table, which would, in most of the area, rise and fall with the tides, would occur for the portion of the CoPCs that is not bound to solids and could result in transport of CoPCs to Ward Cove. Biological degradation of organic compounds is likely in the area because of conditions favorable to microbial growth (e.g., higher organic matter). Transport of contaminants on particles by erosion or surface runoff to Ward Cove may also occur.

As part of the uplands remedial investigation, four surface composite and four subsurface sediment samples were collected and analyzed for metals, volatile and semivolatile organic compounds, PCDDs/Fs, petroleum hydrocarbons, and conventional parameters. Sodium and calcium were the only chemicals that were detected at concentrations above background levels. As part of its 1992 dredging permit, KPC monitors conventional water quality parameters twice a month in three wells installed near the dike of the dredge spoil area. The monitoring data indicate the presence of periodic depressed levels of dissolved oxygen and elevations in COD, BOD, and sulfate. Water quality in Ward Cove is discussed in greater detail in Section 4.5. In addition, as part of the 1997 dredging permit, leachate samples were collected at three times of the day for 15 days at seven locations adjacent to the dredge spoil area in January 1997 (concurrent with placement of dredge material) and monitored for conventional water quality parameters, including dissolved oxygen, sulfide, ammonia, and phenols. No elevated concentrations of any parameter (or low dissolved oxygen levels) were found.

4.1.3.6 Storm Water Discharge

Storm water runoff from the site is collected, treated in four oil–water separators, and discharged to Ward Cove through storm water outfalls or through the facility's process water outfalls. Prior to installation of the oil–water separators, some storm water ran directly into the Cove. In limited portions of the mill where storm water is not captured by the storm water and process water collection systems, it can run directly into Ward Cove; however, any current loading to Ward Cove from surface water runoff would be expected to be minimal. During storm events, surface water discharge through the storm water outfalls is 3.85 mgd.

As part of KPC's current NPDES permit (No. AK-00092-2) sampling of storm water discharges from both the pulp mill and the wood waste landfill is conducted during three storm events per year. Samples from 10 discharges are analyzed for total recoverable arsenic, copper, manganese, selenium, and zinc, total aqueous and aromatic hydrocarbons (EPA methods 602 and 610), 5-day BOD, COD, color, oil and grease, pH, and TSS. Samples from selected stormwater discharges associated with the landfill are analyzed for total chromium, total recoverable lead, mercury, and silver, and hardness. In addition, TCDDs/Fs and hardness are monitored quarterly in selected discharges associated with the landfill.

Samples are collected monthly from the main outfall (001) and analyzed for total recoverable cadmium and manganese, 5-day BOD, chlorine, color, oil and grease, pH, turbidity, and acute and chronic whole effluent toxicity. In addition, the sanitary sewer is monitored monthly for 5-day BOD, TSS, and fecal coliform; landfill leachate is monitored quarterly at two stations for PCDDs/Fs and total copper, manganese, selenium and zinc; and ambient monitoring is conducted monthly at 12 stations in Ward Cove for temperature, salinity, dissolved oxygen, turbidity, and pH.

4.1.3.7 Releases of Airborne Contaminants from Power Boilers

The primary combustion air emission units from the mill were the two multi-fuel power boilers and four chemical (red liquor) recovery boilers. The most recent of the emission sources to be added to the mill was an oil-fired package boiler in 1988. Shutdown of the power boilers was completed in March 1998.

Potential risks to human health and the environment from aerial deposition (primarily residual risk) was evaluated through aerial deposition modeling and soil sampling conducted as part of the uplands remedial investigation. PCDDs/Fs were detected in soil samples at levels that were not of concern for human health, but were above background concentrations. Because aerial deposition of current stack emissions is controlled by wind direction, which is predominantly from the south (onshore) in the vicinity of the mill, impacts to Ward Cove from aerial deposition are expected to be minimal.

4.1.3.8 Spills and Accidental Releases

All spills at the facility are currently reported and cataloged as part of the Spill Prevention, Containment, and Countermeasure plan implemented in 1995 (KPC 1995). As part of the plan, information on spills at the facility from 1979 to 1995 was compiled. Spills prior to 1979 are not documented. Major spills (greater than 100 gallons) that have been reported at the facility since 1979 are listed in Table 4-1. CH2M HILL (1995) provides a discussion of other potential sources of contamination. Chronic, small quantity petroleum releases have also occurred along the railroad tracks (primarily lubrication oil from the locomotive), in the process subarea (mainly hydraulic oil), and beneath the wood conveyor system (hydraulic oil and antifreeze). In addition to petroleum products, spills of process chemicals, such as red liquor, have occurred periodically in the process subarea (including the access road on the north side of the area), and spills of small volumes of solvents may have occurred at the paint shop, electrical shop, and steam-cleaning areas. Water used to wash out tank cars containing magnesium oxide was routinely discharged to the ground at the railroad sidings south of the boiler house. Petroleum or chemicals released during these spills may have reached Ward Cove via surface runoff.

In the 1970s, a 55-gallon drum of calcium hypochlorite, used as a cleaner for the pipeline from Connell Lake to the mill, spilled adjacent to the pipeline and flowed into Ward Creek, resulting in a fish kill. No residual effects of the kill are evident at present.

TABLE 4-1. MAJOR SPILLS AT THE PULP MILL AREA

Date	Location	Quantity (gallons)	Material
January 1979	Oil line pump	35,800	No. 6 fuel oil
July 1987	Powerhouse supply line	425–975	No. 6 fuel oil
August 1987	Bulk storage tank	150	No. 6 fuel oil
June 1989	Powerhouse supply line	200	No. 6 fuel oil
January 1992	Fuel oil strainer	9,000	No. 6 fuel oil
July 1992	Main sewer floor trench	100	Turbine oil
May 1994	Power boiler feed line	100	No. 6 fuel oil
November 1994	Bowser system	200–300	Hydraulic oil
November 1994	Diesel defoamer	130	Diesel
December 1994	Diesel defoamer	240	Diesel
January 1996	Log deck area	200	Diesel
September 1996	Wastewater treatment plant	3,500	Cation flocculant

4.1.4 Cannery

Although sources associated with past activities at KPC are the focus of this investigation, site data suggest that the fish cannery located on the southeastern shore of Ward Cove may have contributed CoPCs to the Cove. This facility is known to have at least one outflow that discharges into Ward Cove. A pile of fish processing waste of undetermined size is directly offshore from the cannery near Station 25, and elevated concentrations of several CoPCs were detected in a sediment sample collected from this station (described below). The outflow and waste pile are the two most probable potential sources of CoPCs from this fish processing facility. KPC is not currently planning further investigation of this potential source.

4.2 SURFACE SEDIMENTS

Surface sediments were sampled for CoPCs in 1996 and 1997 at a total of 44 different locations in Ward Cove and 2 locations in the Moser Bay reference area. Twenty-eight stations were sampled in Ward Cove during the 1996 (Phase 1) sampling effort, and 33 stations were sampled in Ward Cove during the 1997 (Phase 2) sampling effort. Seventeen of the 1997 Ward Cove stations were sampled at Phase 1 locations, and 16 new stations were sampled in Ward Cove in 1997. Two samples were collected at Moser Bay in both investigations. In addition, two intertidal samples were included in the 1997 sampling effort.

Surface sediment results for both 1996 and 1997 are summarized in Table 4-2. The complete data tables are provided in Appendix A1. Conventional analytes and grain size results are listed in Table A1-1. Metals data are presented in Table A1-2. Table A1-3 lists results for semivolatile organic compounds. Dioxin and furan results are provided in Table A1-4. Results for pulp mill compounds, analyzed only in 1996, are presented in Table A1-5.

The distribution of CoPCs in surface sediments is discussed in the following sections. The spatial distributions of chemicals that may be sensitive to seasonal changes or that appeared to be different for the two years are shown separately for 1996 and 1997 because of the different sampling times for the two years. Sampling in 1996 was conducted in late May and early June, and sampling in 1997 was conducted in late July and early August. Data for constituents that had consistent distributions for the two years were combined into a single figure and average values for the two years were used to represent chemical concentrations. Potential analytical variability and small scale spatial variability (due to reoccupation of the same station in two successive years) were also taken into consideration in determining whether it was appropriate to portray data on a single figure or two separate figures.

TABLE 4-2. SUMMARY OF SURFACE SEDIMENT DATA COLLECTED IN WARD COVE AND MOSER BAY IN 1996 AND 1997

Analyte	Concentration Range	Median	Number of Detected Values	Number of Samples	Frequency of Detection (percent)	Station with Maximum Concentration	Year in Which Maximum Value Was Detected	
							1996	1997
Conventional Analytes								
Acid-volatile sulfide (mg/kg)	240 - 17,000	2,450	28	28	100	16		X
Total ammonia (mg/kg)	3.2 - 690	83	72	72	100	44	X	
Biochemical oxygen demand 5-day test (g/kg)	0.72 - 65	9.2	72	72	100	38		X
Chemical oxygen demand (g/kg)	1.3 - 2,400	17	72	72	100	8	X	
Total sulfide (mg/kg)	20 *U* - 27,000	2,500	71	72	99	17	X	
Total organic carbon (percent)	1.1 - 41	23	72	72	100	2	X	
Gravel (percent)[a]	0 *U* - 61	2.0	71	72	99	50		X
Sand (percent)								
1.0–2.0 mm	0.27 - 20	2.7	72	72	100	18	X	
0.50–1.0 mm	0.53 - 20	5.3	72	72	100	33		X
0.25–0.50 mm	0.8 - 17	9.0	72	72	100	33		X
0.125–0.25 mm	0.79 - 16	10	72	72	100	16	X	
0.062–0.125 mm	1.9 - 35	9.5	72	72	100	29		X
Silt (percent)	4.5 - 78	37	72	72	100	30		X
Clay (percent)	1.5 - 34	21	72	72	100	44		X
Total solids (percent of wet weight)	12 - 80	19	72	72	100	50		X
Extractable organic halides (mg/kg)	10 *U* - 79	44	4	29	14	25		X
Metals								
Arsenic (mg/kg)	2.7 - 39	21	31	31	100	7	X	
Cadmium (mg/kg)	0.14 - 7.3	3.5	49	49	100	7	X	
Methylmercury (µg/kg)	0.22 - 14.3	0.90	28	28	100	23		X
Total mercury (mg/kg)	0.1 *U* - 0.7	0.20	20	49	41	3	X	
Zinc (mg/kg)	39 - 530	159	49	49	100	25		X
Semivolatile Organic Compounds (µg/kg)								
Low molecular weight PAHs								
Naphthalene	1 - 440	50	26	32	81	3	X	
2-Methylnaphthalene	10 *U* - 280	53	25	32	78	3	X	
Acenaphthylene	10 *U* - 110	20	7	32	22	23	X	
Acenaphthene	10 *U* - 500	40	19	32	59	3	X	
Fluorene	10 *U* - 470	46	25	32	78	3	X	
Phenanthrene	6 - 1,100	230	30	32	94	3	X	
Anthracene	3 - 380	57	27	32	84	25	X	
Total	10 *U* - 2,800	470	32	32	100	3	X	

TABLE 4-2. (cont.)

Analyte	Concentration Range	Median	Number of Detected Values	Number of Samples	Frequency of Detection (percent)	Station with Maximum Concentration	Year in Which Maximum Value Was Detected	
							1996	1997
High molecular weight PAHs								
Fluoranthene	10 *U* - 2,200	390	30	32	94	4		X
Pyrene	8 - 1,800	270	30	32	94	4		X
Benz[a]anthracene	3 - 990	120	29	32	91	25	X	
Chrysene	4 - 1,300	130	30	32	94	25	X	
Benzo[b]fluoranthene	3 - 740	100	28	32	88	25		X
Benzo[k]fluoranthene	10 *U* - 530	52	26	32	81	25	X	
Benzo[a]pyrene	10 *U* - 750	63	27	32	84	25	X	
Indeno[1,2,3-cd]pyrene	1 - 520	37	25	32	78	25	X	
Dibenz[a,h]anthracene	6 - 73[b]	20	7	32	22	25	X	
Benzo[ghi]perylene	1 - 290	32	24	32	75	25	X	
Total	10 *U* - 8,100	1,300	30	32	94	25	X	
Sum of carcinogenic PAH compounds	10 *U* - 4,900	540	30	32	94	25	X	
Sum of carcinogenic PAH, RPC[c]	11 - 1,100	140	30	32	94	25	X	
Sum of carcinogenic PAH, RPC[d]	0 *U* - 1,100	96	30	32	94	25	X	
Phenols and miscellaneous compounds								
Phenol	10 *U* - 990	200	28	51	55	25		X
4-Methylphenol[e]	10 *U* - 17,000	990	55	72	76	31		X
Benzoic acid	100 *U* - 1,600	500	16	32	50	4	X	
Dibenzofuran[f]	10 *U* - 180	20	8	19	42	4		X
Pulp Mill Compounds (mg/kg)[g]								
Individual chlorinated phenols	0.8 *U* - 2.3 *U*	1.8 *U*	0	6	0	--		
Individual chlorinated guaiacols	0.8 *U* - 2.3 *U*	1.8 *U*	0	6	0	--		
Individual chlorinated catechols	0.8 *U* - 2.3 *U*	1.8 *U*	0	6	0	--		
Individual chlorinated vanillins	0.8 *U* - 2.3 *U*	1.8 *U*	0	6	0	--		
Individual chlorinated syringaldehydes	0.8 *U* - 2.3 *U*	1.8 *U*	0	6	0	--		
Trichlorosyringol	0.8 *U* - 2.3 *U*	1.8 *U*	0	6	0	--		
Abietic acid	18 - 150	55	6	6	100	7	X	
Dehydroabietic acid	12 - 150	36	6	6	100	7	X	
12-Chlorodehydroabietic acid	2.9 - 22	5.0	5	6	83	7	X	
14-Chlorodehydroabietic acid	1.5 *U* - 23	1.8	2	6	33	7	X	
Dichlorodehydroabietic acid	1.5 *U* - 14	2.0	3	6	50	7	X	
9,10-Dichlorostearic acid	1.5 *U* - 7.2 *U*	1.8 *U*	0	6	0	--		
Pimaric acid	1.5 *U* - 7.2 *U*	1.8 *U*	0	6	0	--		
Isopimaric acid	4.3 - 22	6.9	5	6	83	7	X	
Linoleic acid	1.5 *U* - 7.2 *U*	1.8 *U*	0	6	0	--		
Oleic/linolenic acids	7.2 *U* - 79	14	5	6	83	7	X	

TABLE 4-2. (cont.)

Analyte	Concentration Range	Median	Number of Detected Values	Number of Samples	Frequency of Detection (percent)	Station with Maximum Concentration	Year in Which Maximum Value Was Detected	
							1996	1997
Dioxins and Furans (ng/kg)								
2,3,7,8-Tetrachlorodibenzodioxin	0.65 *U* - 2.6[b]	1.3	12	42	29	4	X	
1,2,3,7,8-Pentachlorodibenzo-*p*-dioxin	0.66 *U* - 12	3.6	25	41	61	4		X
1,2,3,4,7,8-Hexachlorodibenzo-*p*-dioxin	0.72 *U* - 11[b]	4.4	13	42	31	4	X	X
1,2,3,6,7,8-Hexachlorodibenzo-*p*-dioxin	0.72 *U* - 44	14	35	42	83	4		X
1,2,3,7,8,9-Hexachlorodibenzo-*p*-dioxin	0.73 *U* - 30	8.7	31	42	74	4		X
1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin	2 *U* - 920	290	38	42	90	4	X	
Octachlorodibenzo-*p*-dioxin	11 - 6,300	2,100	41	42	98	4		X
Total tetrachlorodibenzo-*p*-dioxins	0.66 *U* - 290	66	37	42	88	4	X	
Total pentachlorodibenzo-*p*-dioxins	0.66 *U* - 160	37	35	42	83	4		X
Total hexachlorodibenzo-*p*-dioxins	0.86 *U* - 390	120	37	42	88	4	X	
Total heptachlorodibenzo-*p*-dioxins	4.3 - 3,100	800	42	42	100	4	X	
2,3,7,8-Tetrachlorodibenzofuran	0.58 *U* - 36	9.1	9	42	21	7	X	
1,2,3,7,8-Pentachlorodibenzofuran	0.55 *U* - 9.7	3.0	21	42	50	4	X	
2,3,4,7,8-Pentachlorodibenzofuran	0.58 *U* - 20	3.7	25	42	60	7	X	
1,2,3,4,7,8-Hexachlorodibenzofuran	0.66 *U* - 85	5.7	8	42	19	7	X	
1,2,3,6,7,8-Hexachlorodibenzofuran	0.61 *U* - 39	4.0	24	42	57	7	X	
1,2,3,7,8,9-Hexachlorodibenzofuran	1.0 *U* - 4.5 *U*	2.1	0	42	0	--		
2,3,4,6,7,8-Hexachlorodibenzofuran	0.73 *U* - 30	4.0	17	42	40	7	X	
1,2,3,4,6,7,8-Heptachlorodibenzofuran	0.78 *U* - 310	48	39	42	93	24	X	
1,2,3,4,7,8,9-Heptachlorodibenzofuran	0.98 *U* - 27	3.6	11	42	26	7	X	
Octachlorodibenzofuran	2.6 *U* - 390	145	38	42	90	4		X
Total tetrachlorodibenzofurans	0.58 *U* - 230	52	36	42	86	4		X
Total pentachlorodibenzofurans	0.6 *U* - 170	35	34	42	81	7	X	
Total hexachlorodibenzofurans	0.86 *U* - 370	69	36	42	86	7	X	
Total heptachlorodibenzofurans	0.87 *U* - 640	155	39	42	93	24	X	
Dioxin and furan toxic equivalent concentration[c]	1.1 - 46	15	42	42	100	7	X	
Dioxin and furan toxic equivalent concentration[d]	0 *U* - 45	12	42	42	100	7	X	

Note: Results are presented on a dry weight basis unless noted otherwise.

See Tables A1-2 to A1-4 in Appendix A.

Concentrations for conventional analytes and organic compounds are rounded to two significant figures. Concentrations for metals are rounded to three significant figures if over 10 and two significant figures if less than 10.

Field replicates were treated as unique data points and the results were not averaged.

Medians were calculated using the detection limits for those congeners that were undetected.

TABLE 4-2. (cont.)

-- - not applicable; the analyte was not detected at any station
PAH - polycyclic aromatic hydrocarbon
RPC - relative potency concentration
U - undetected at concentration listed

[a] When grain-size distribution is determined by the analytical laboratory, the term "gravel" is a designation for a specific size fraction in the sediment. This verbiage does not mean that the sediment is gravel. In some shallower parts of the Cove, the "gravel" size fraction could consist of wood debris and probably includes organic material.

[b] At least one detection limit exceeded the concentration of the indicated maximum detected value.

[c] Detection limits are included in the sum at half their value.

[d] Detection limits are excluded from the sum.

[e] 3- and 4-Methylphenol results were quantified as 4-methylphenol.

[f] Dibenzofuran was analyzed only in 1997.

[g] Pulp mill compounds were analyzed only in 1996.

4.2.1 Grain Size

Grain size measurements were taken at Ward Cove and Moser Bay in 1996 and 1997. Grain size measurements provide information on the size distribution of organic and inorganic particles in surface sediment. Grain size distributions can provide information on the energetics of the deposition environment and the particle transport processes that act upon sediments. Organic matter and fine-grained material frequently covary and are associated with relatively quiescent, low energy environments.

Sediments that can pass through a screen size of 0.062 mm or less are termed fines and represent silt or clay particles. Isopleths for the percent fines at Ward Cove are portrayed in Figure 4-2. Percent fines ranged from 7 to 84 percent throughout Ward Cove, whereas in Moser Bay fines ranged from 53 to 91 percent. Most of Ward Cove is covered with at least 50 percent fine-grained material; the deeper portion of the Cove is covered with more than 75 percent of fine-grained material (Figure 4-2). The coarsest material is found near the mouth of Ward Creek. Settling of fine material in the deeper water of the Cove is consistent with a quiescent hydrodynamic environment (i.e., no strong currents) in the deeper and outer parts of Ward Cove. It is also likely that wood debris, which is coarser than silt and clay, is responsible for the lower concentrations of fine-grained material in the shallower parts of the Cove.

4.2.2 Total Organic Carbon

TOC is a measurement of all forms of organic matter in sediments, including bark and wood debris, fish waste, and naturally occurring organic matter provided by surface runoff and water column productivity. TOC concentrations exceeded 10 percent in most of the inner half of Ward Cove and along the northern shoreline. The exception to this pattern occurs in the northeastern portion of the Cove at the mouth of Ward Creek (Figure 4-3). At Moser Bay, TOC values were in the 3 to 5 percent range. The highest TOC values in Ward Cove were greater than 30 percent and were found immediately offshore from the KPC facility near Outfall 001 (Stations 1, 6, and 37) and Outfall 002 (Station 5). Two additional values above 30 percent were measured at Stations 26 and 35. Most of the deeper and outer portions of Ward Cove had TOC concentrations under 10 percent.

The observation that deeper parts of Ward Cove do not have elevated TOC despite the presence of fine-grained sediment indicates that currents are inadequate to move organic solids out of the inner portion of the Cove. The relatively large size of much of the organic material (i.e., bark and wood debris) may be responsible for this transport limitation.

4.2.3 Total Ammonia

Total ammonia is a measure of all forms of ammonia nitrogen in sediments, including ammonia associated with dissolved or particulate organic matter and dissolved ammonia in pore water. Ammonia is produced in anoxic sediments (i.e., sediments with no

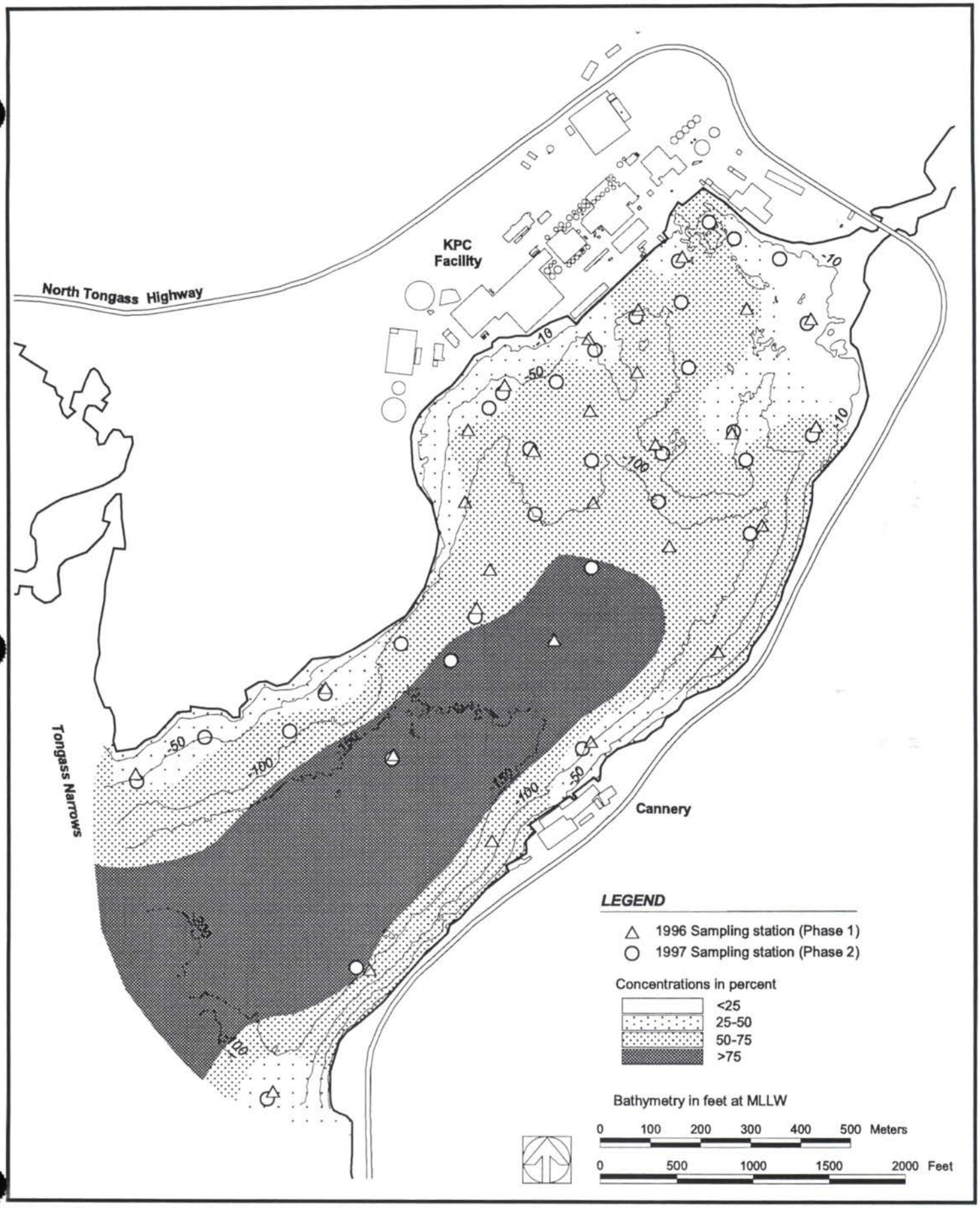


Figure 4-2. Distribution of percent fines (particles <0.062 mm) in Ward Cove sediments in 1996 and 1997.

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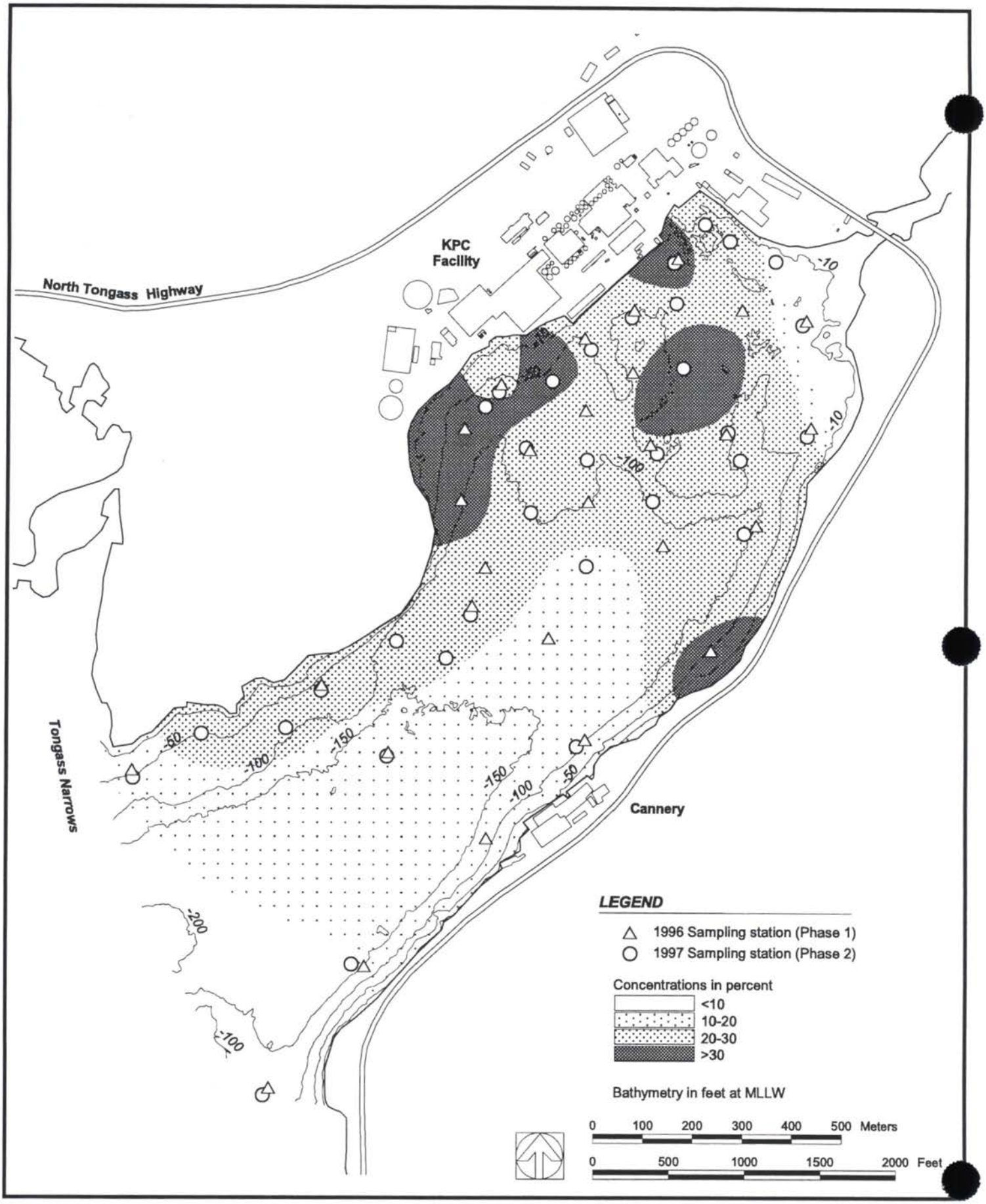

Figure 4-3. Distribution of total organic carbon in Ward Cove sediments in 1996 and 1997.

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oxygen) during the microbially mediated degradation of organic matter. Total ammonia exceeded 50 mg/kg over much of Ward Cove during both 1996 and 1997 (Figures 4-4 and 4-5) and exceeded 150 mg/kg along much of the northern shoreline during both years. In contrast, the total ammonia concentration in Moser Bay ranged from 11 to 18 mg/kg in both years. The highest ammonia values in Ward Cove during 1996 (360 mg/kg at Station 6) and 1997 (690 and 540 mg/kg at Station 44) were measured directly offshore from the KPC facility along the north shoreline west of Outfall 001.

The greater spatial extent of elevated ammonia concentrations in 1997 relative to 1996 may be a consequence of the different sampling times. Microbial action is likely to have been greater during the warmer summer months during which the 1997 samples were collected, leading to the production of more ammonia and a general increase in sediment ammonia concentrations. However, in general, ammonia concentration trends are consistent for the two years.

4.2.4 Sulfide (Acid-Volatile and Total)

Interpretation of sulfide data is complicated by the variety of forms of sulfide present in sediment and the different analytical methods used to quantify sulfide. Sulfide in sediment is present in several forms, including dissolved sulfide (which is present in three forms: hydrogen sulfide, bisulfide, and sulfide) and particulate metal sulfide (which is primarily present as iron sulfide). Iron sulfide is generally separated into two operational categories: AVS (which includes "amorphous" iron sulfide, mackinawite, greigite, and pyrrhotite) and pyrite (Cornwell and Morse 1987). Pyrite resists dissolution by acids and is typically analyzed using methods that include an oxidative step.

In the present study, bulk sediment was analyzed for AVS and total sulfide. AVS was analyzed to allow for comparison to total metals to provide a conservative estimate of metal bioavailability.[1] Total sulfide (i.e., dissolved sulfide plus some categories of particulate metal sulfide) was analyzed to gain a general sense of the total sulfide present in Ward Cove sediment. Neither of these metals has an oxidative step and therefore do not quantify pyrite. Given their similarity, they might be expected to provide comparable results; however, a sample-by-sample comparison indicates that AVS results are sometimes higher than total sulfide results (Table A1-1 in Appendix A1). This is likely due to limitations in the analytical method for total sulfide that tends to bias results low.[2]

---

[1] The molar ratio of simultaneously extracted metals (SEM) to AVS is considered to be an index of bioavailability; if it is less than 1, metals are believed to be unavailable and nontoxic to benthic organisms. If total metals are used instead of SEM, the analysis is highly conservative because SEM is always a fraction of the total metals in sediment.

[2] The method used to determine total sulfide is biased low because acid is added to an open flask, where hydrogen sulfide can be lost to the atmosphere, and because "bumps" during distillation can result in incomplete digestion (Payfair 1998, pers. comm.).

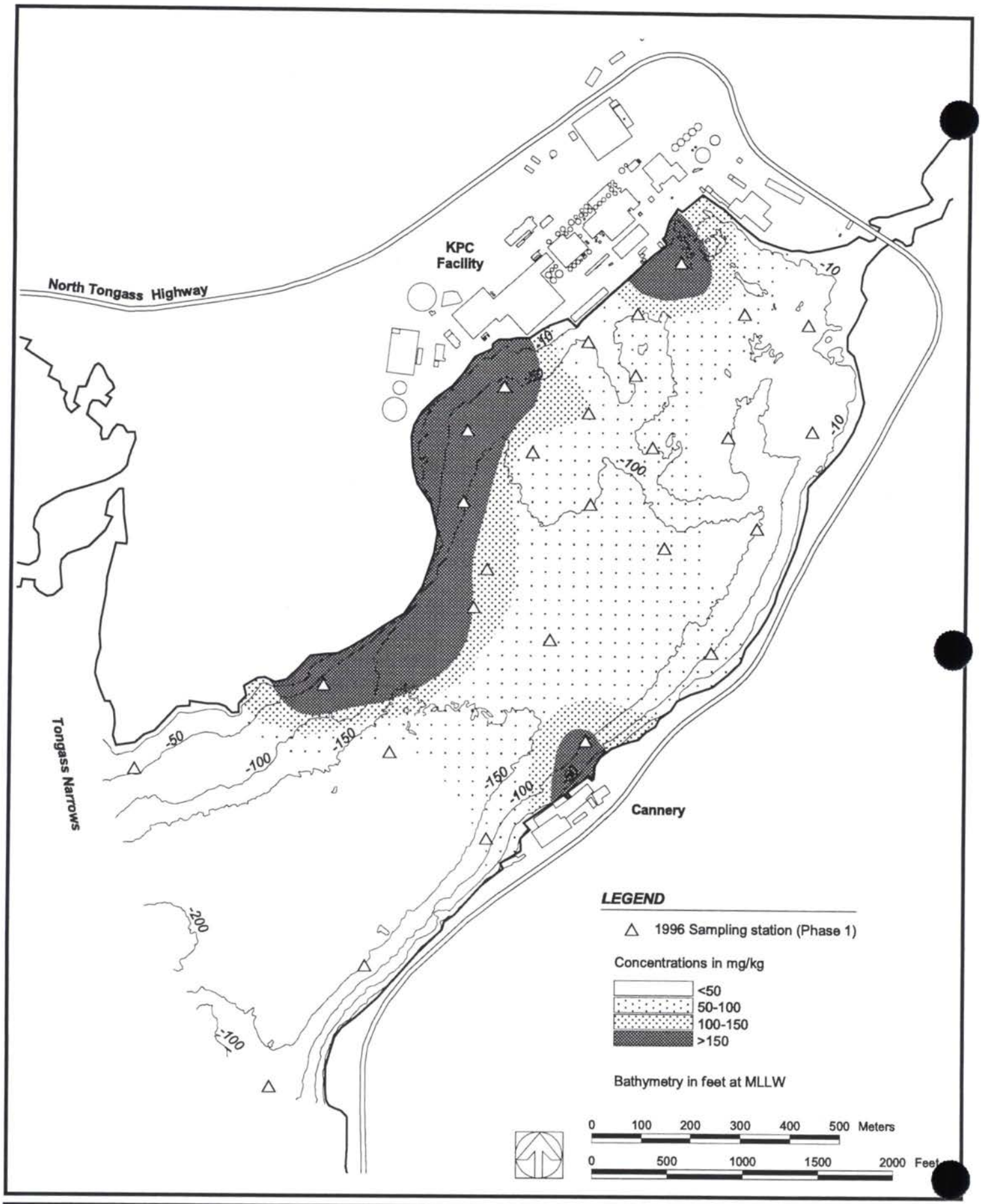

Figure 4-4. Distribution of total ammonia in Ward Cove sediments in May and June 1996.

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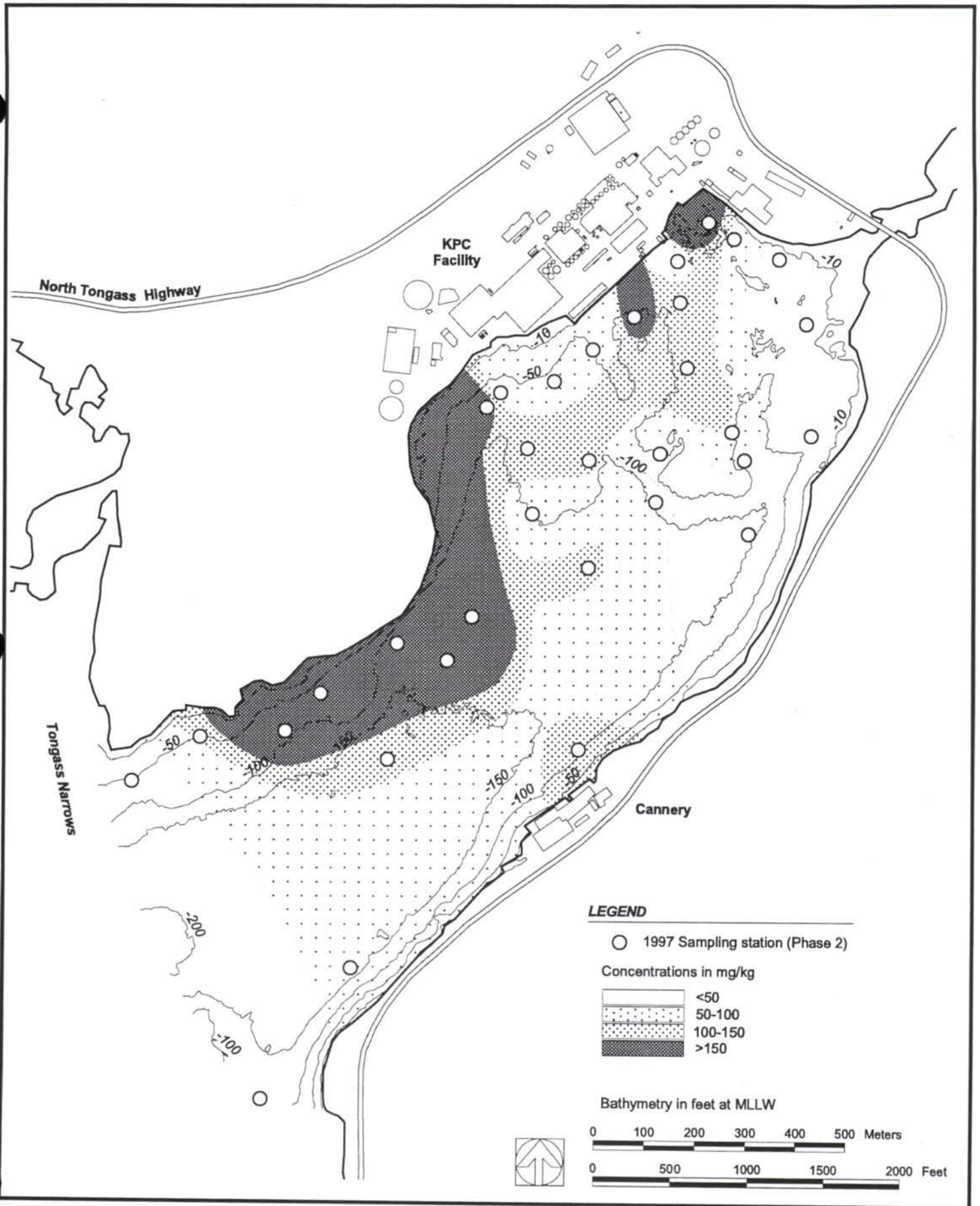

Figure 4-5. Distribution of total ammonia in Ward Cove sediments in July and August 1997.

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Dissolved sulfide was not specifically analyzed in the present study and is not usually analyzed in environmental assessments of sediments because of the complex handling steps required.[3] However, areas with elevated total sulfide are likely to have elevated levels of dissolved sulfide. Dissolved sulfide was measured as part of the specialized toxicity testing and is discussed in greater detail in Section 7.1.4.

The spatial pattern of total sulfide concentration in sediments varied considerably from 1996 to 1997 (Figures 4-6 and 4-7). Thus, although sulfide concentrations in Ward Cove were generally higher than in Moser Bay (240 to 590 mg/kg), sulfide production rates (due to microbial degradation of organic matter) or loss rates (e.g., by diffusion to the water column) in Ward Cove can evidently change relatively rapidly. Seasonal changes may be responsible in part for the marked changes observed from 1996 to 1997. However, unlike ammonia, which showed the same general spatial pattern in 1996 and 1997, the spatial pattern of total sulfide concentration differs between the two years. It is also possible that small-scale spatial variability contributes to the different distribution patterns observed in 1996 and 1997.

4.2.5 Biochemical and Chemical Oxygen Demand

BOD is an indicator of the amount of oxygen required by aerobic microorganisms in metabolizing organic material. There was a marked change in the distribution of BOD in Ward Cove sediments from 1996 to 1997 (Figures 4-8 and 4-9), with a much larger area of BOD greater than 15 g/kg in 1997. This increased BOD extent is adjacent to the KPC facility and along most of the southern shoreline of Ward Cove. Like sulfide, the spatial patterns of BOD differ considerably from 1996 to 1997. The most striking difference is the higher BOD values along the southern shore of Ward Cove in 1997. BOD shows no distinct spatial relationship with ammonia or TOC. The lack of correspondence suggests that the organic material in Ward Cove is heterogeneous spatially and chemically.

COD is an indicator of the amount of oxygen that can be consumed by inorganic reactions that occur in sediment. Like BOD, there was a marked change in COD in Ward Cove sediments from 1996 to 1997 (Figures 4-10 and 4-11). In contrast to BOD, which increased in concentration, COD concentrations generally decreased by over an order of magnitude from 1996 to 1997. COD concentrations in Moser Bay sediments showed the same pattern. The quality of COD data for 1996 and 1997 was reevaluated to determine if an analytical or dilution error had occurred, but no error was found. NPDES data for COD samples collected in 1994, 1995, and 1996 are similar, suggesting that the 1997 data are anomalous. (Moser Bay samples were not collected in 1994 and 1995.) The timing of sample collection in 1996 and 1997 (May/June and July/August, respectively)

[3] Sediment needs to be sampled, sectioned, and centrifuged to separate pore water from the solid phase. Frequently, at least a portion of this separation is conducted in a glove bag under a nitrogen atmosphere.

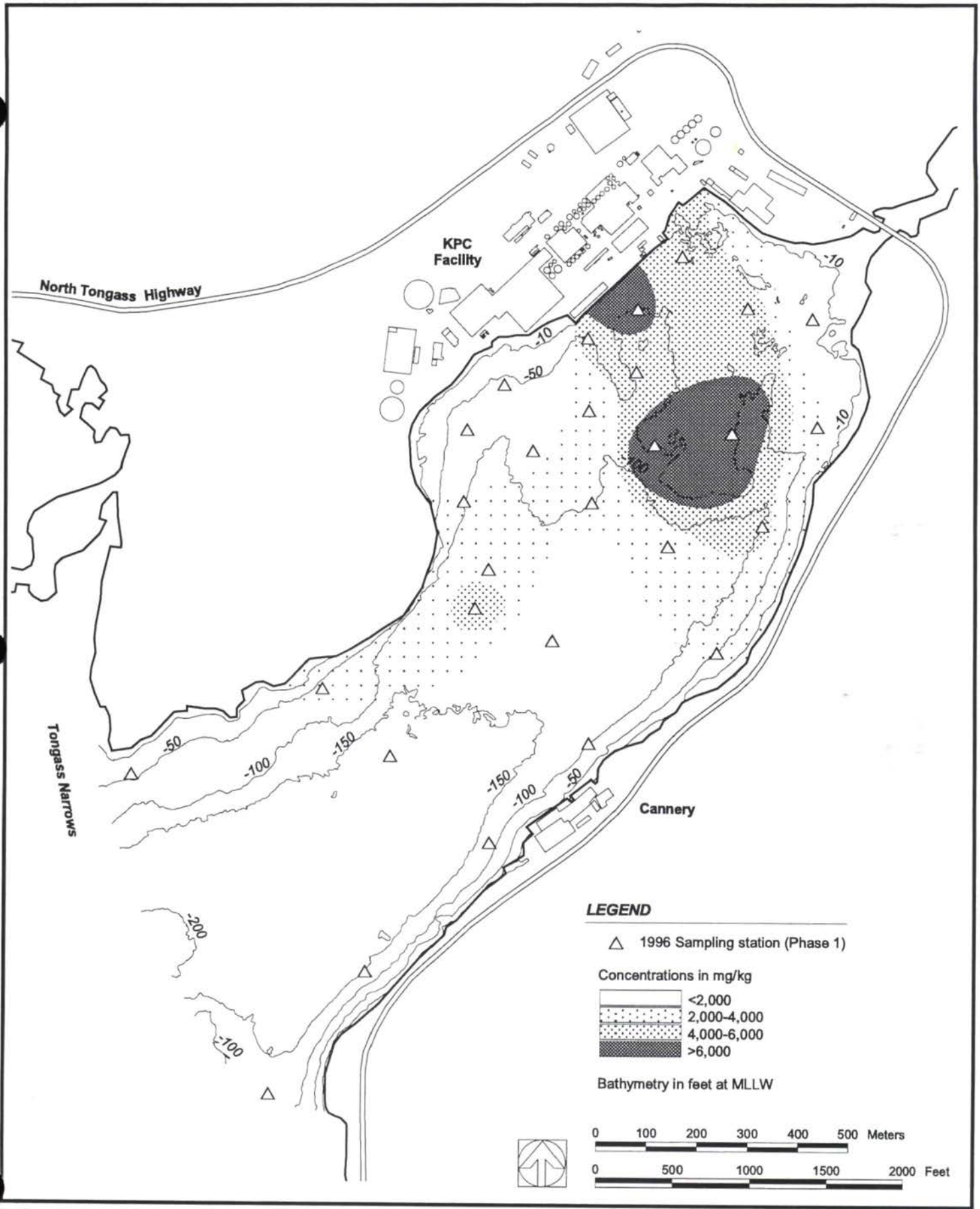

Figure 4-6. Distribution of total sulfide in Ward Cove sediments in May and June 1996.

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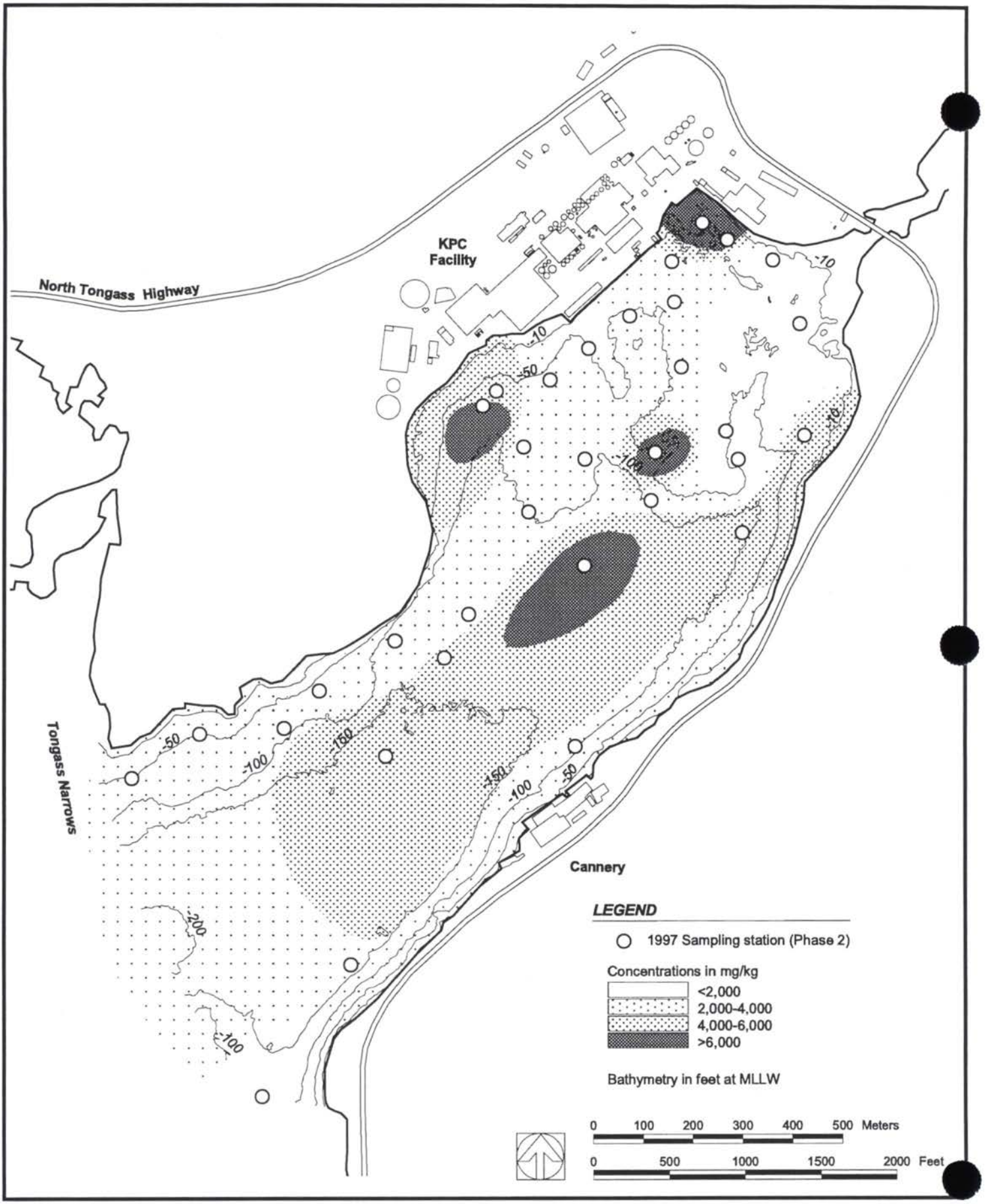

Figure 4-7. Distribution of total sulfide in Ward Cove sediments in July and August 1997.

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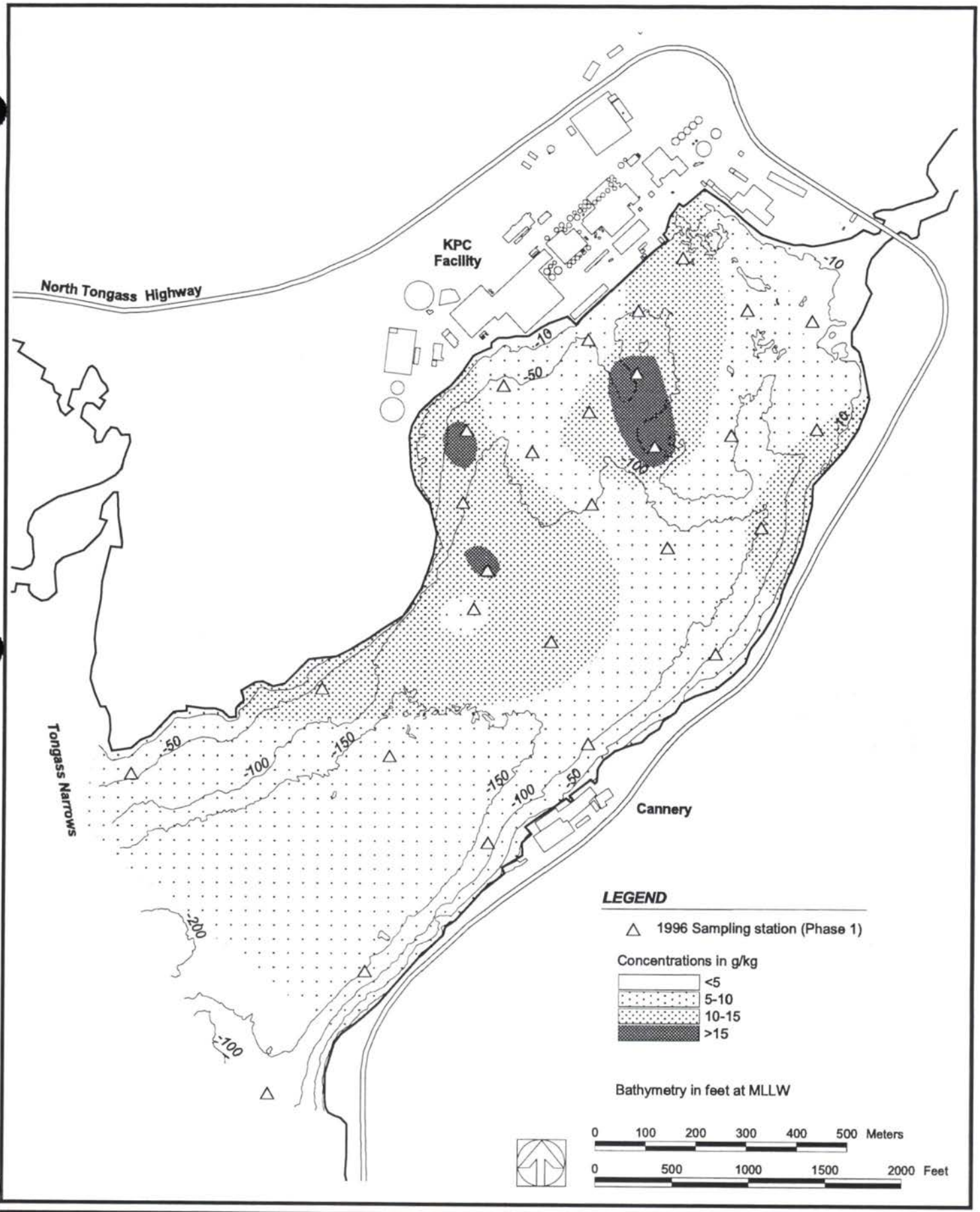

Figure 4-8. Distribution of BOD in Ward Cove sediments in May and June 1996.

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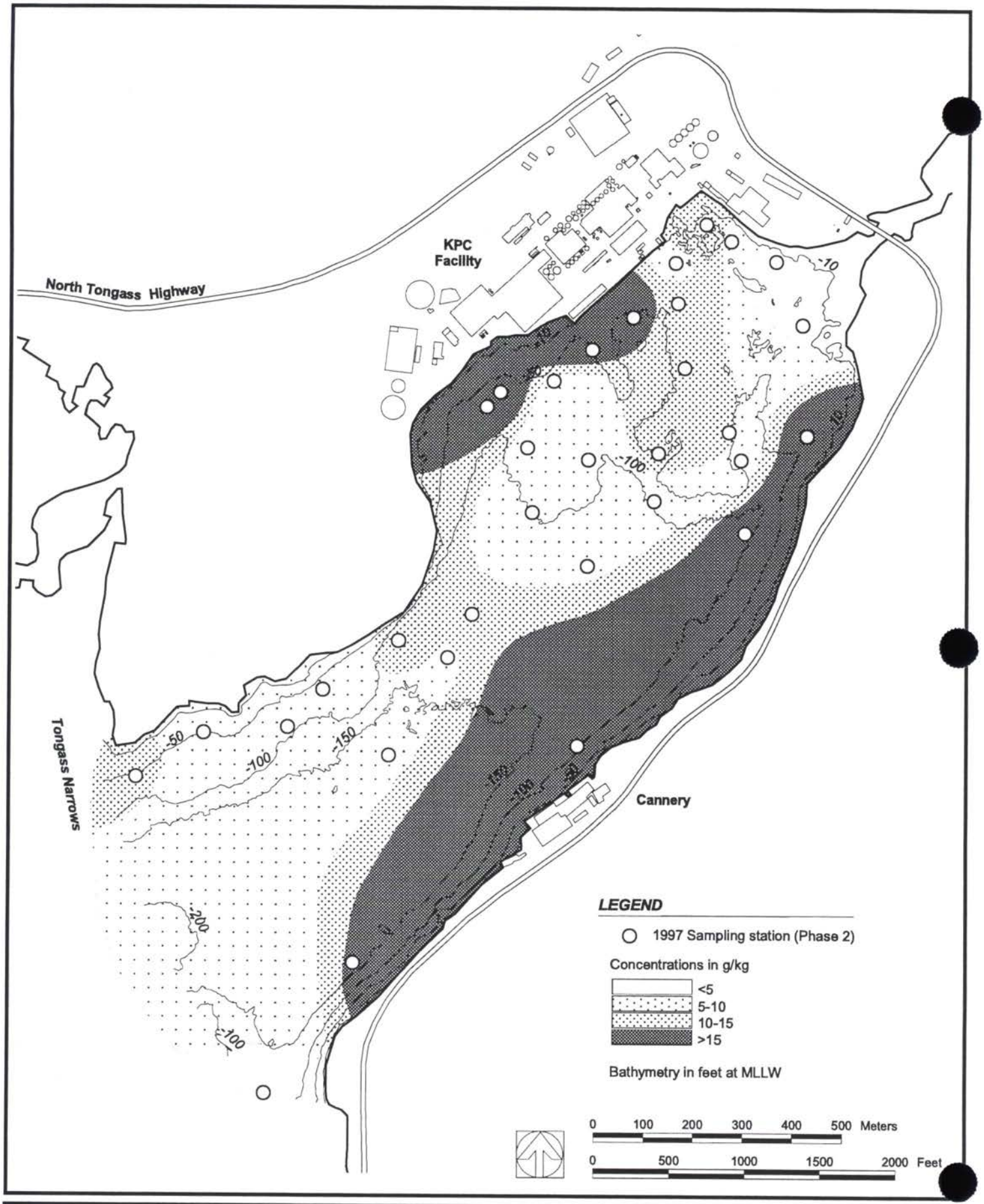

Figure 4-9. Distribution of BOD in Ward Cove sediments in July and August 1997.

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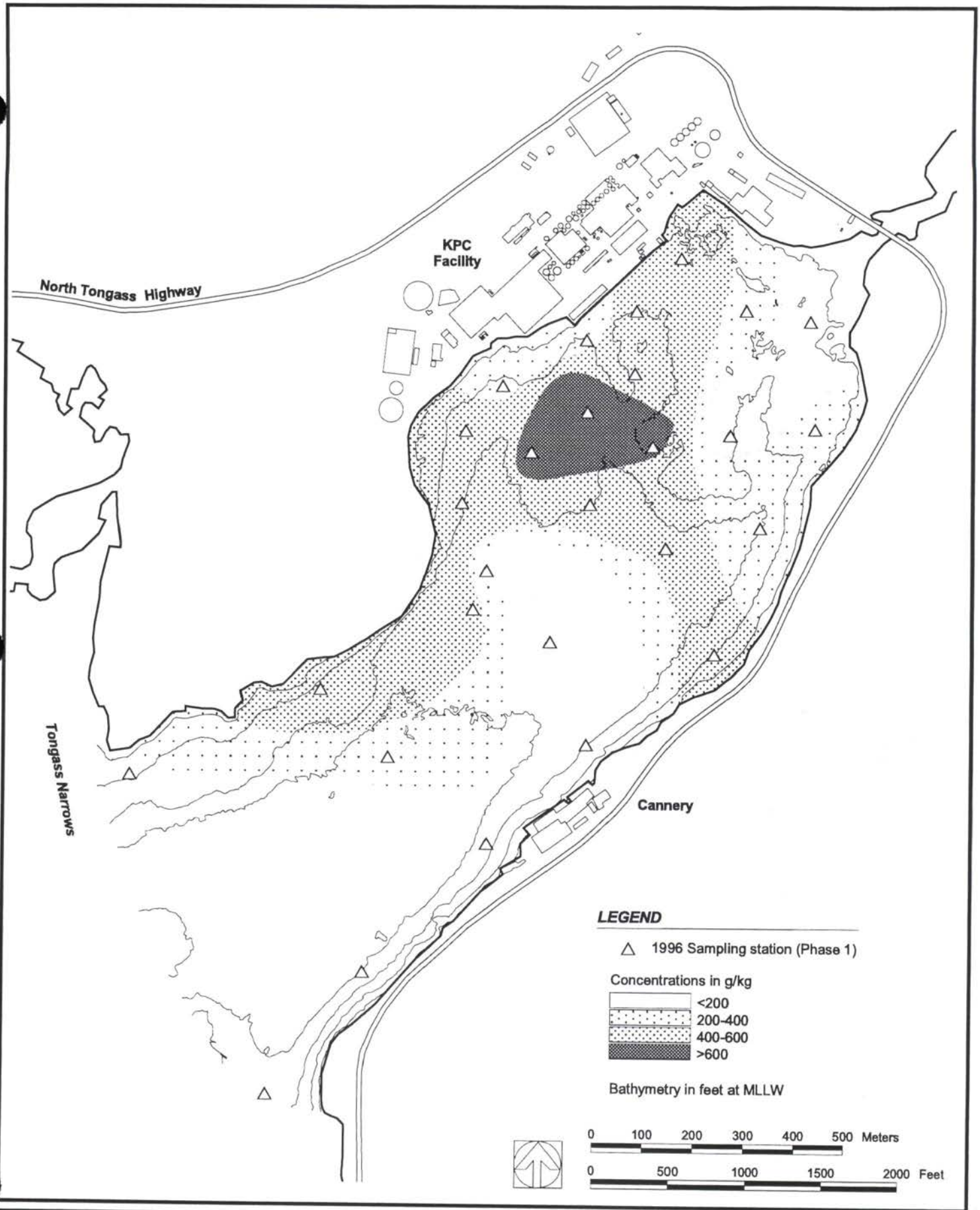

Figure 4-10. Distribution of COD in Ward Cove sediments in May and June 1996.

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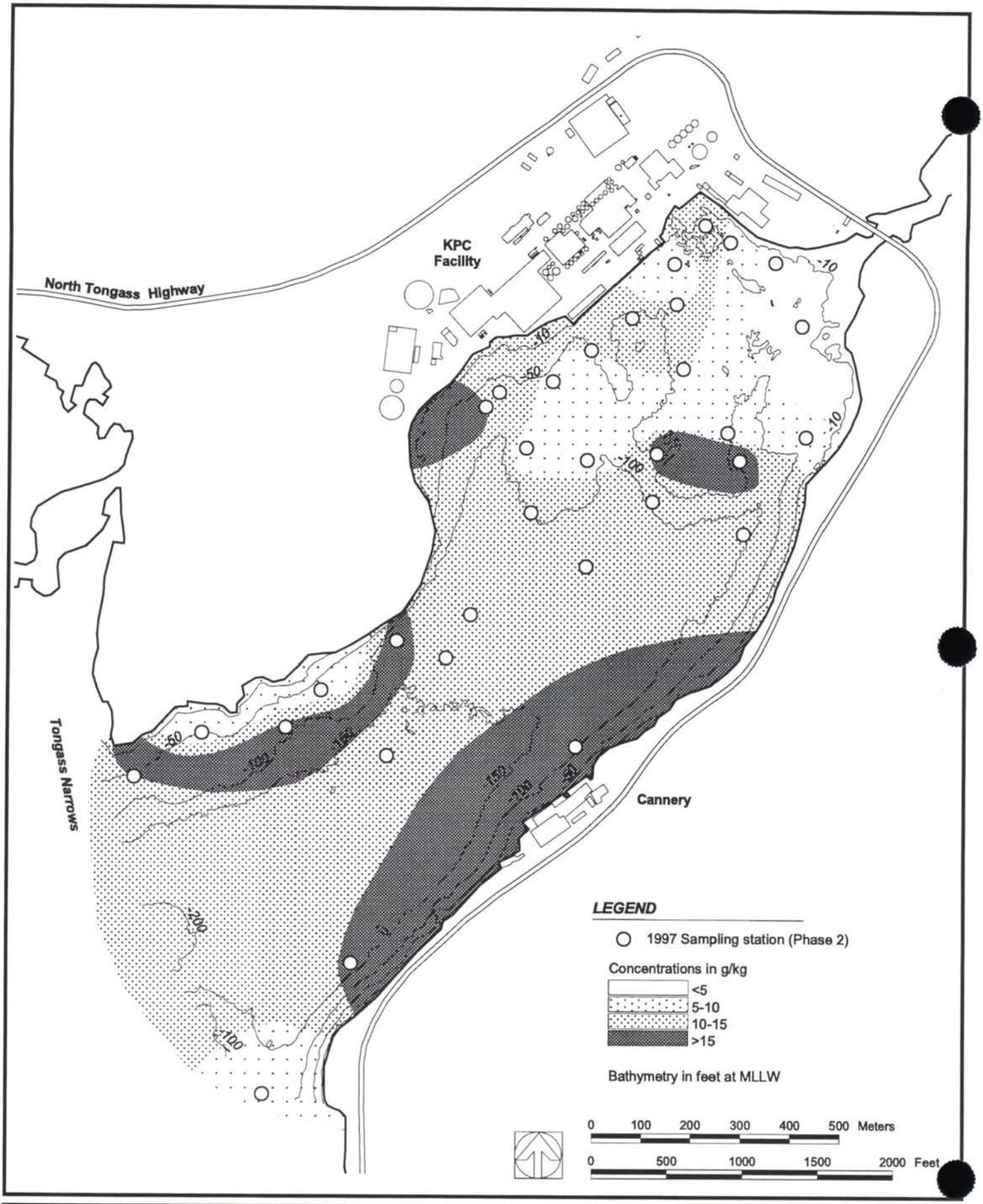

Figure 4-11. Distribution of COD in Ward Cove sediments in July and August 1997.

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and the termination of pulping activities in March 1997 (prior to sampling) should be considered in interpreting the COD data.

4.2.6 Cadmium and Arsenic

Cadmium and arsenic are metals (a metalloid in the case of arsenic). Cadmium and arsenic are normally present at detectable concentrations in sediments. In Moser Bay sediments, cadmium ranged from 0.3 to 1.5 mg/kg, and arsenic ranged from 5.2 to 12 mg/kg. Elevations over these concentrations were found in Ward Cove and were generally confined to the middle section of the Cove (Figures 4-12 and 4-13, respectively). Cadmium concentrations exceeding 4 mg/kg and arsenic concentrations exceeding 20 mg/kg are found in this area. High concentrations of cadmium and arsenic are not found directly adjacent to either the KPC facility or the cannery. Arsenic had been detected at elevated concentrations both in wastewater treatment plant grit and sludge samples and in soil samples from various areas of the mill. Crushed gravel and ESP flyash have been identified as sources of the arsenic at the facility. Cadmium had not been detected at elevated concentrations in any source material or soil samples.

4.2.7 Total Mercury

Mercury is also a metal that is present naturally in sediment; however, unlike arsenic and cadmium, it is often present below analytical detection limits. Mercury was initially considered a potential concern for sediments because it was present at elevated concentrations in caustic that was used for a short period at the KPC facility. However, results indicate that mercury concentrations were close to or below method detection limits for both Ward Cove and Moser Bay sediments during 1996 and 1997. The highest concentration of total mercury in the Cove (0.7 mg/kg) was at Station 3, immediately offshore of the KPC facility. However, mercury was undetected (at a detection limit of 0.2 mg/kg) in all samples collected in 1997, suggesting that the 1996 value of 0.7 mg/kg was an outlier value.

4.2.8 Zinc

Zinc is another metal that is present naturally in sediments at detectable concentrations. Zinc concentrations in Moser Bay sediments ranged from 70 to 90 mg/kg. No waste-related source of zinc has been identified at the KPC facility; however, galvanized roofing or piping may be a source of zinc. Unlike cadmium and arsenic, the highest concentrations of zinc in Ward Cove sediments were found near the KPC facility and the cannery (Figure 4-14), which is consistent with the incidental presence of zinc in construction materials. The highest zinc concentration in Ward Cove, 530 mg/kg, was measured off the cannery.

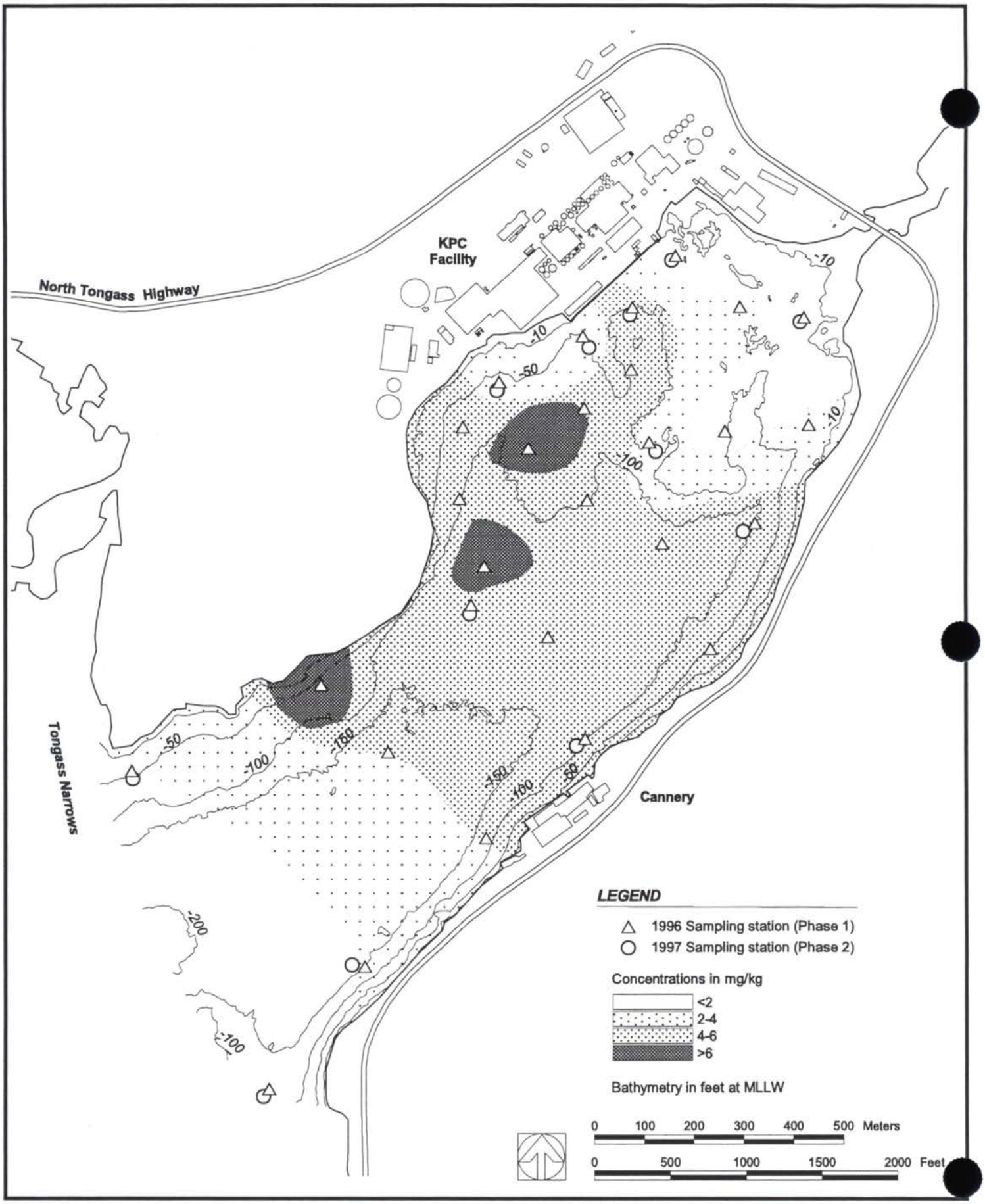

Figure 4-12. Distribution of cadmium in Ward Cove sediments in 1996 and 1997.

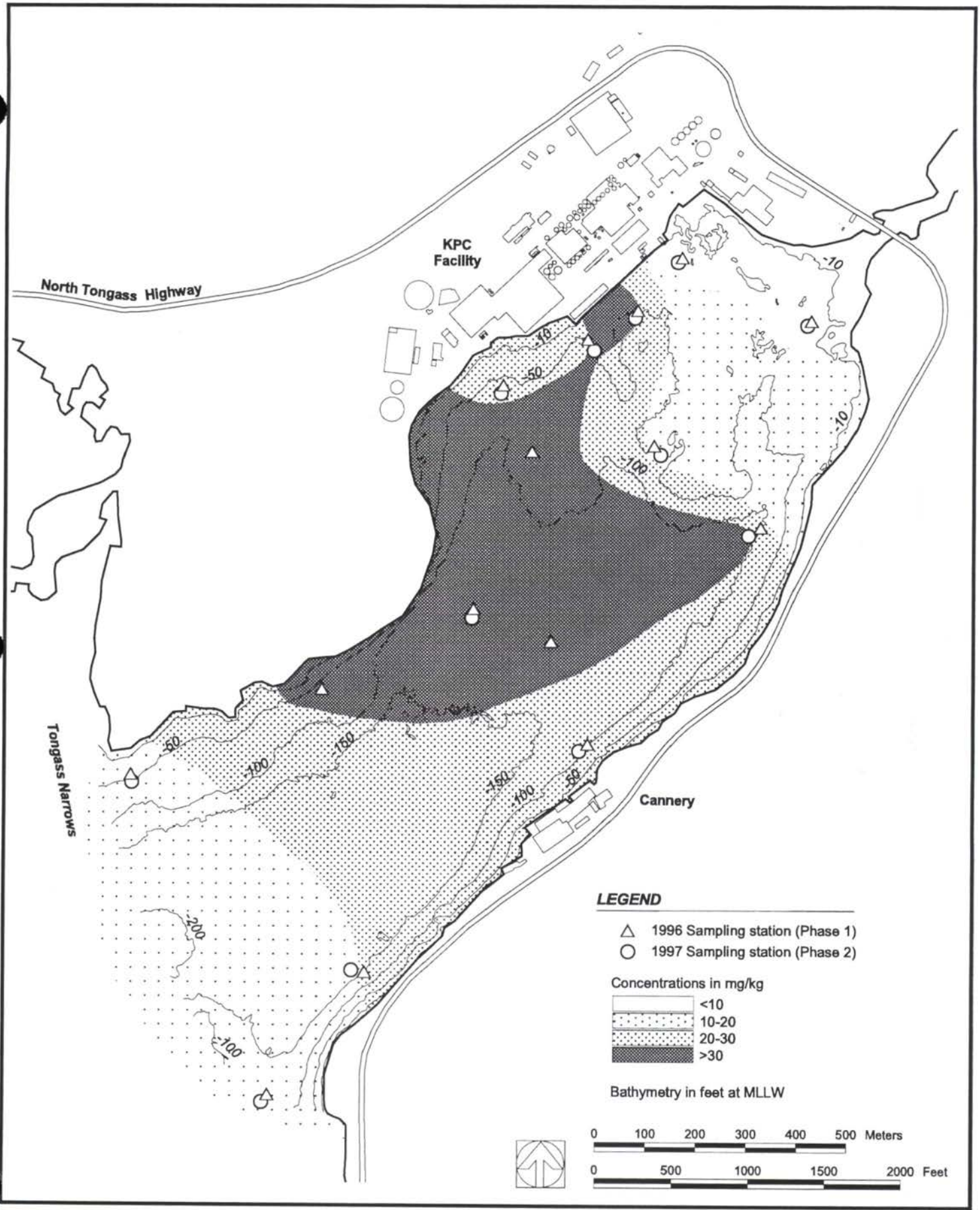

Figure 4-13. Distribution of arsenic in Ward Cove sediments in 1996 and 1997.

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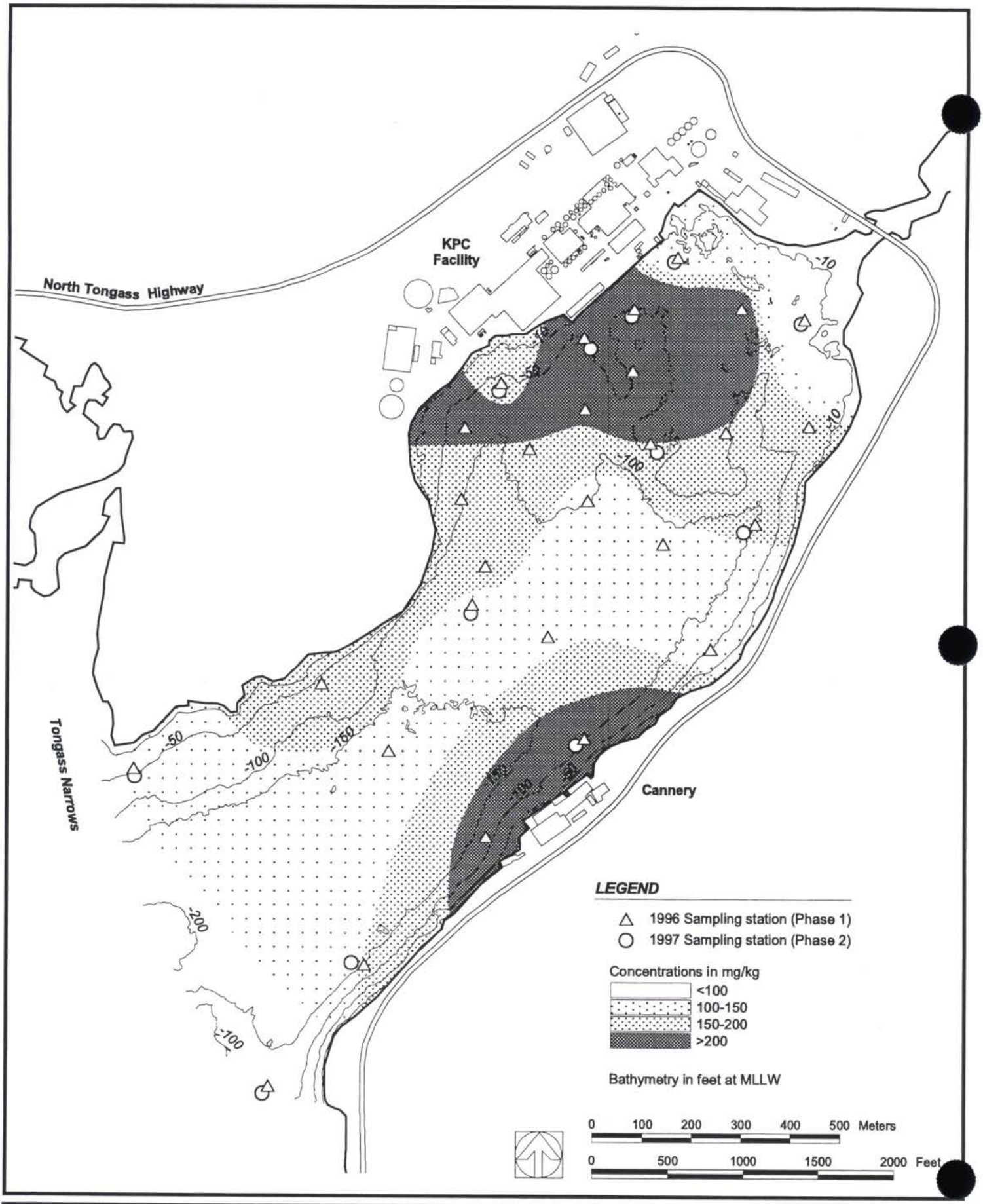

Figure 4-14. Distribution of zinc in Ward Cove sediments in 1996 and 1997.

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4.2.9 Phenol and 4-Methylphenol

Phenol and 4-methylphenol (*p*-cresol) are closely related aromatic compounds that are commonly associated with organic debris. Cresols in general and 4-methylphenol in particular are natural products that are widely distributed. *p*-Cresol (4-methylphenol) is used in the formulation of antioxidants and in the fragrance and dye industries, and mixtures of *m*- and *p*-cresol are used as disinfectants and preservatives. Cresols also occur naturally as metabolites of microbial activity and are known to be an intermediate biotransformation product of natural aromatics such as lignin constituents (lignin is the "glue" that holds a tree together). The elevated concentrations of 4-methylphenol adjacent to KPC facility are probably attributable to the microbial degradation of the lignin constituents of wood wastes and the production of 4-methylphenol as a biodegradation intermediate. The presence of elevated concentrations of 4-methylphenol in sediments adjacent to the cannery may also be due to the microbial degradation of natural aromatics associated with fish wastes (such as certain amino acids). Alternatively, it may be associated with activities at the cannery (e.g., a disinfectant, a food flavor additive, a fish smoking process). Detected concentrations of phenol in Ward Cove sediments ranged from 15 to 510 μg/kg in 1996 and from 12 to 993 μg/kg in 1997 (Table A1-3). 4-Methylphenol concentrations were typically an order of magnitude greater than phenol concentrations in Cove sediments.

There was a noticeable change in the distribution of 4-methylphenol in Ward Cove sediments from 1996 to 1997 (Figures 4-15 and 4-16). In 1996, the highest concentrations (greater than 500 μg/kg) were found adjacent to the KPC facility and the cannery. In 1997, the distribution of 4-methylphenol in sediments exceeding 500 μg/kg extended along much of the north shore of Ward Cove and adjacent to the cannery along Ward Cove's south shore. The increase in 4-methylphenol concentrations could be related to increased microbial activity during the summer when the water temperature is elevated. 4-Methylphenol was undetected in Moser Bay sediments at a detection limit that ranged from 10 to 20 μg/kg.

4.2.10 Carcinogenic PAHs

PAH compounds represent a broad class of large, multi-ringed aromatic compounds that occur naturally. They are also important constituents of petroleum products. PAH compounds are present throughout the Cove; however, levels are not considered to be a concern for benthic organisms (PTI 1995a). The distribution of carcinogenic PAH compounds was evaluated to support the human health risk assessment and the ecological food web assessment. The highest concentrations of carcinogenic PAHs in Ward Cove sediments are found off the cannery and off the state airplane ramp (near the mouth of the Cove on the south shore) (Figure 4-17). Elevated concentrations of carcinogenic PAHs were also found directly off the KPC facility. Concentrations of carcinogenic PAHs reported for Station 47 may be biased high because results reported as undetected in this sample were elevated by a factor of 10 as compared to other samples.

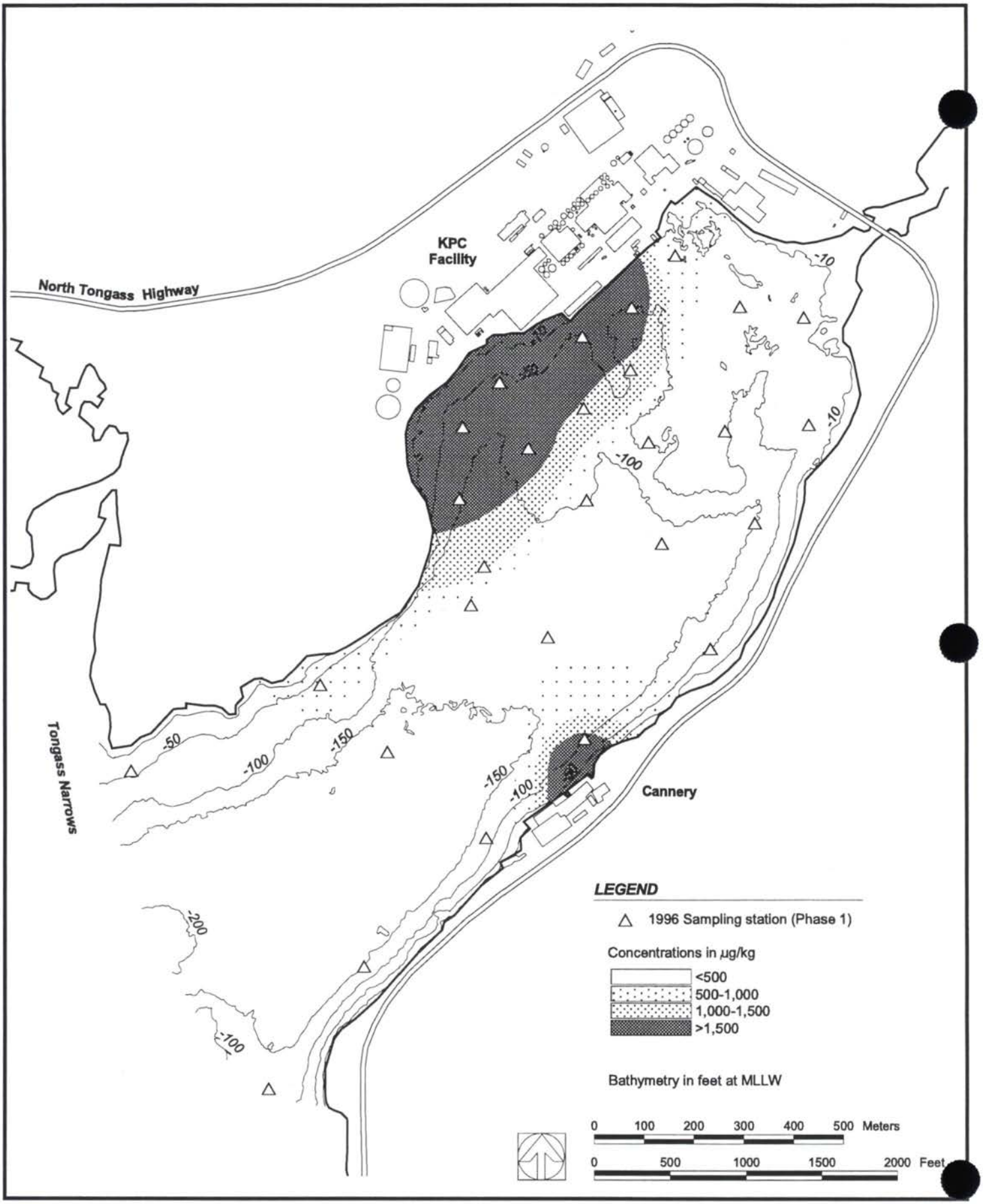

Figure 4-15. Distribution of 4-methylphenol in Ward Cove sediments in May and June 1996.

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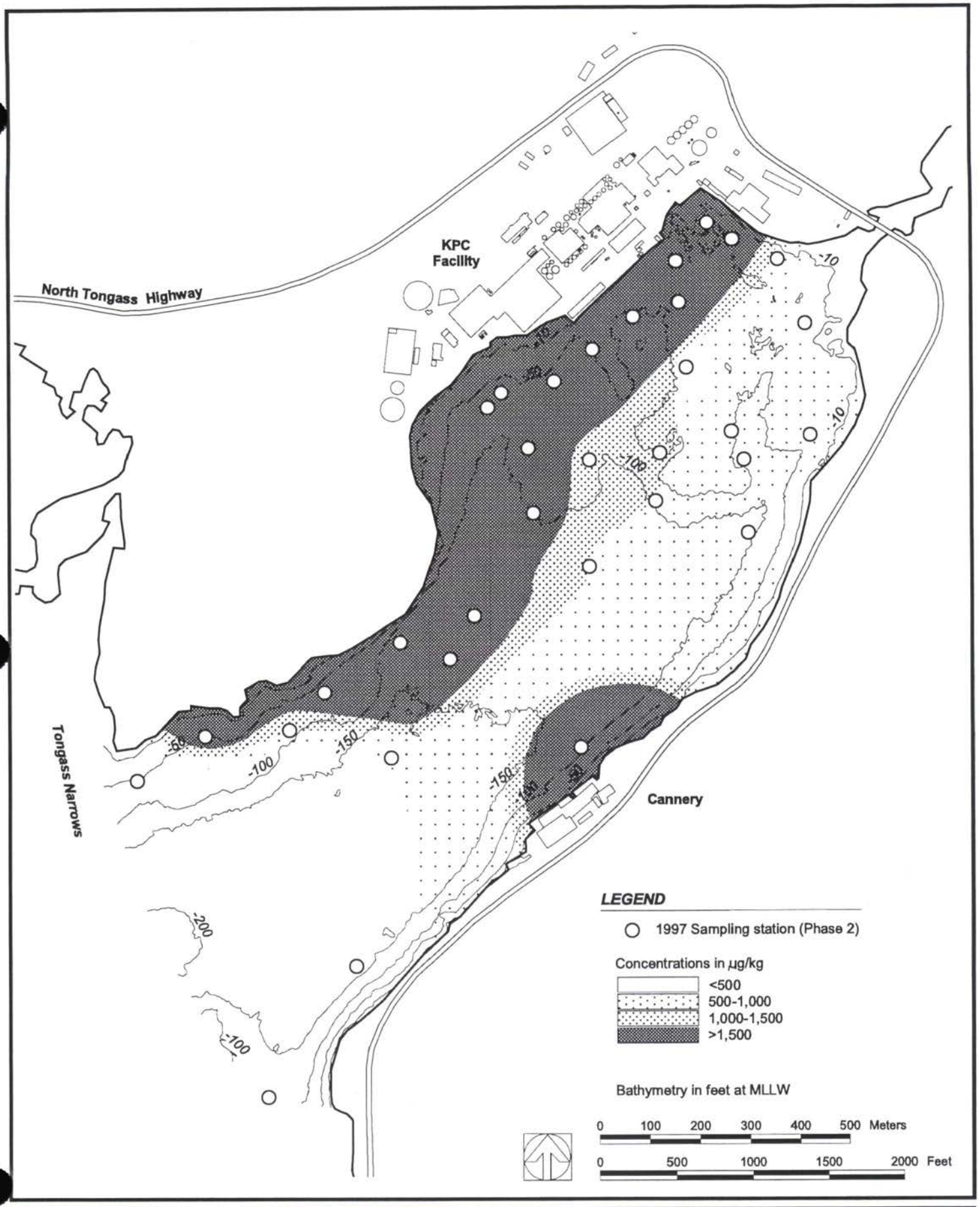

Figure 4-16. Distribution of 4-methylphenol in Ward Cove sediments in July and August 1997.

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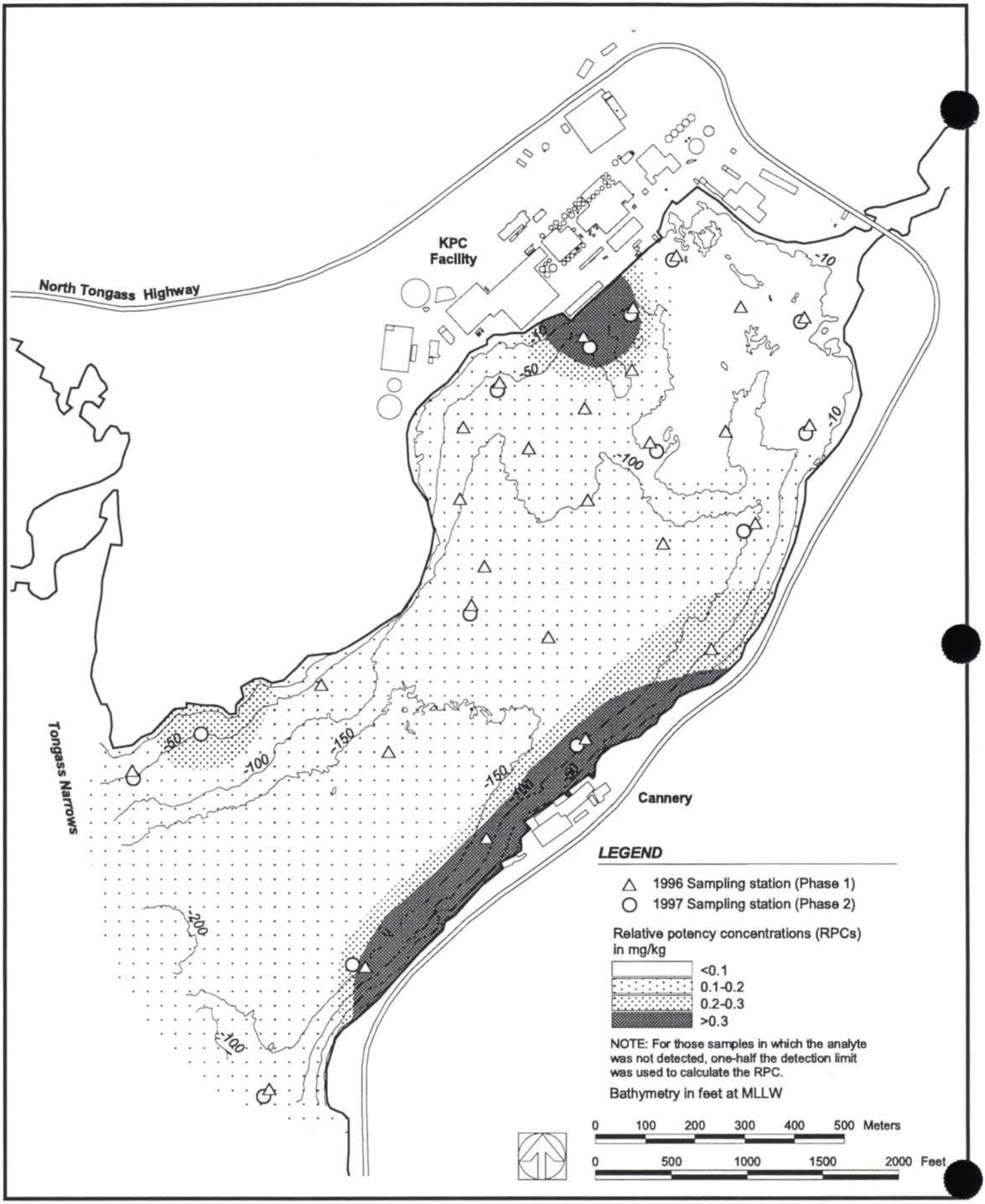

Figure 4-17. Distribution of carcinogenic PAHs in Ward Cove sediments in 1996 and 1997.

4.2.11 2,3,7,8-TCDD and TCDF Toxic Equivalent Concentrations

Chlorinated dioxins and furans are two classes of complex organic compounds that are frequently associated with pulp mills. The dioxin/furan compound with the highest concentration in Ward Cove sediment, 1,2,3,4,6,7,8,9-octachlorinated dibenzo-*p*-dioxin (OCDD), is present at concentrations ranging from 26 to 6,300 ng/kg. The most toxic dioxin/furan compound is 2,3,7,8-TCDD. 2,3,7,8-TCDD was undetected throughout most of the Cove; the highest detected concentration was 2.6 ng/kg.

The distributions of TCDD and TCDF TECs were similar in 1996 and 1997; however, because concentrations varied somewhat between the two years, they are plotted separately (Figures 4-18 and 4-19). TECs were highest adjacent to the KPC facility and the cannery. TECs in the central portion of the Cove exceeded 10 ng/kg, whereas concentrations in Moser Bay ranged from 1.1 to 1.7 ng/kg. It should be noted that 12 results (11 for Ward Cove and 1 for Moser Bay) included as 1996 data were from archived samples collected in 1996 but analyzed in 1997 (Table A1-4 in Appendix A1).

4.2.12 Intertidal Sediments

Intertidal sediments were collected from two intertidal transects at the mouth of Ward Creek and analyzed for CoPCs in 1997 (Phase 2). Each transect comprised five stations. Chemical analyses were conducted on composite sediment samples from each transect. In general, the CoPC concentrations in all of the intertidal sediments were lower than in subtidal sediments.

4.2.13 Summary of Chemical Distributions in Surface Sediments and Intertidal Sediments

Analyses of CoPCs in surface sediment indicate the following:

- Concentrations of most of the CoPCs exceed the concentrations found in Moser Bay throughout large portions of the Cove.
- The highest concentrations of many of the CoPCs were found near the KPC facility or the cannery.
- There are differences from year to year in the distributions of some, but not all, CoPCs. The greatest differences occur for those CoPCs that may be susceptible to seasonal changes in biological activity.
- Ward Cove is a hydrologically quiescent environment, and there appears to be little transport of organic solids (TOC) or other CoPCs out of the Cove.
- The composition of organic matter in Ward Cove sediments may be quite complex, with spatial and temporal variations in characteristics.

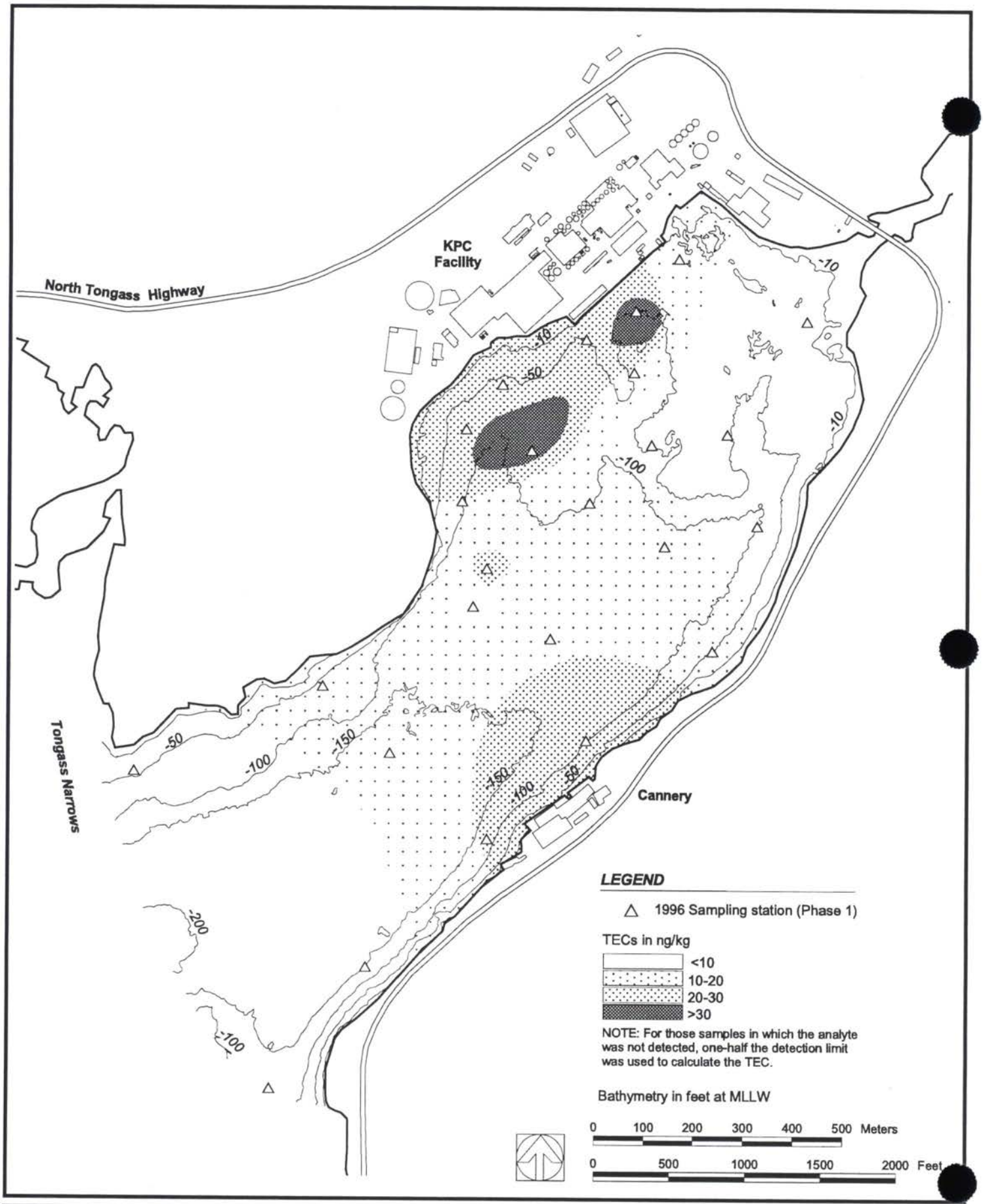

Figure 4-18. Distribution of dioxin and furan toxic equivalent concentrations in Ward Cove sediments in May and June 1996.

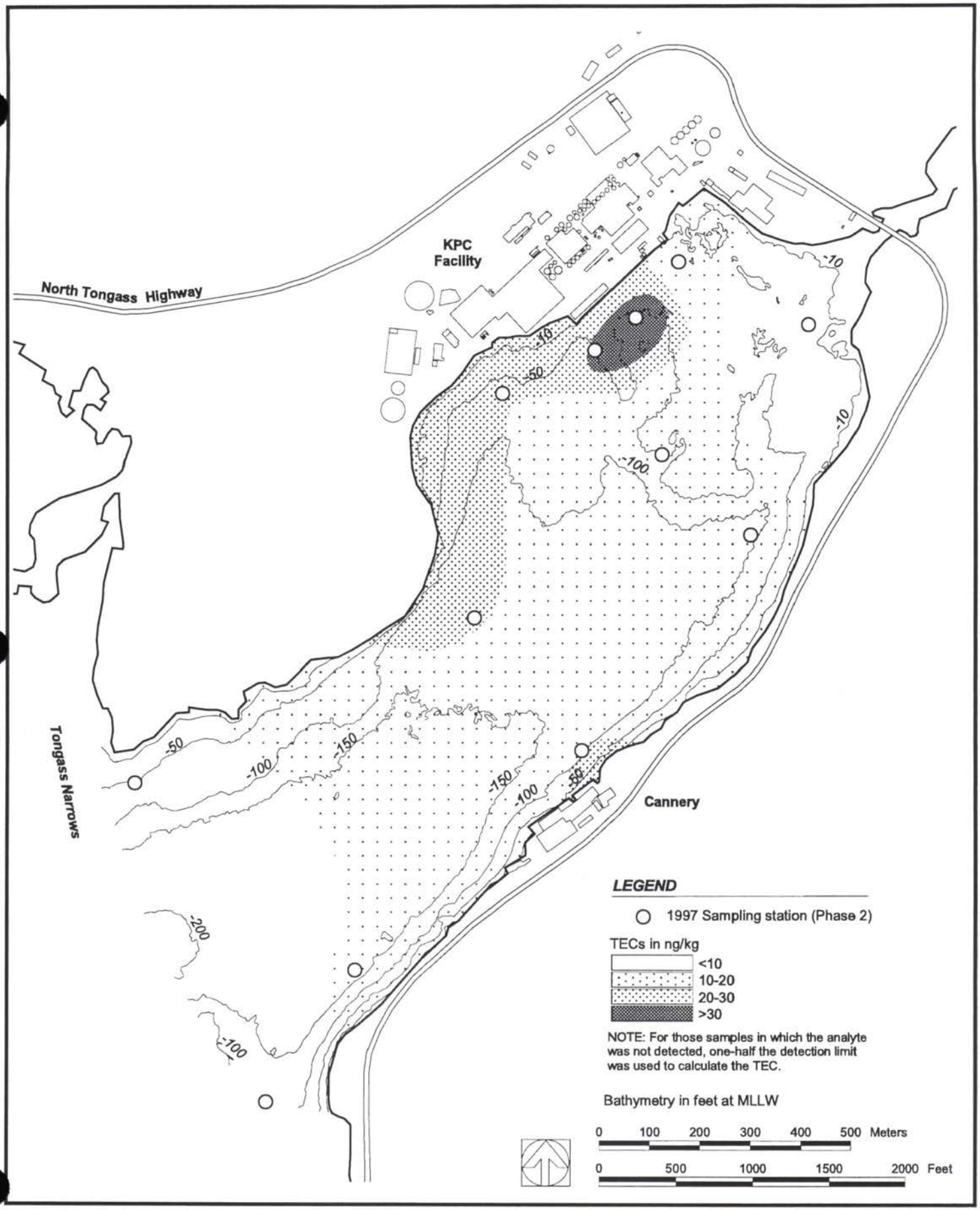

Figure 4-19. Distribution of dioxin and furan toxic equivalent concentrations in Ward Cove sediments in July and August 1997.

Analyses of CoPCs in Ward Cove intertidal sediments indicate that concentrations of CoPCs in those sediments are negligible.

4.3 SUBSURFACE SEDIMENTS

Sediment cores were collected at 16 locations in Ward Cove in 1997 to characterize the vertical extent of sediments affected by releases from the KPC facility. Cores were not collected in the vicinity of the cannery. Chemical analyses were conducted on composite sediment samples from the cores to determine the bulk properties of sediments affected by releases from KPC activities. The vertical extent of affected sediments was determined largely through visual observations and is discussed in Section 4.3.1. The bulk chemical properties of subsurface sediments are discussed in Section 4.3.2.

4.3.1 Vertical Extent of Organic-Rich Sediments

Sediments affected by the KPC facility are distinctly different from the underlying native sediments. The more recent sediment deposits contain wood debris, have high water and organic content, and are black. Underlying native sediments are gray to olive green clay, are comparatively low in organic matter, and sometimes contain shell fragments.

Core locations and the thickness of organic-rich sediments are illustrated in Figure 4-20. At four locations near the KPC facility (Stations 1, 2, 6, and 9), penetration to native material was not achieved, indicating that the thickness of affected sediments in these locations is greater than 8–10 ft. At the two deep stations located in the mid- to outer Cove (Stations 46 and 49), no surficial organic-rich sediments were observed, although some wood debris was present at Station 46. In general, the thickness of organic-rich sediments ranged from 2 to 8 ft at all other stations.

The visual properties and vertical extent of organic-rich sediments are also illustrated in cross-section, using the four transects specified in Figure 4-20. Detailed core logs are provided in Appendix C.[4] Transect 1 (T-1) runs along the north shore and illustrates the shoaling of the organic-rich layer with distance from the KPC facility (Figure 4-21). With the exception of the three cores where penetration to native material was not achieved, the thickness of affected sediments ranged from 1 to 7 ft along T-1. Transect 2 (T-2) runs parallel to T-1 but is further offshore and shares Station 13 with T-1. T-2 indicates that organic-rich sediments affected by the KPC facility are largely confined to the head of the Cove and that the two stations in deeper waters near the mouth of the Cove are largely unaffected (Figure 4-22).

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[4] During collection of sediment cores, water breaks were observed in some cores as an artifact of the piston coring technique. To construct accurate cross-sections, water breaks were eliminated from the stratigraphy.

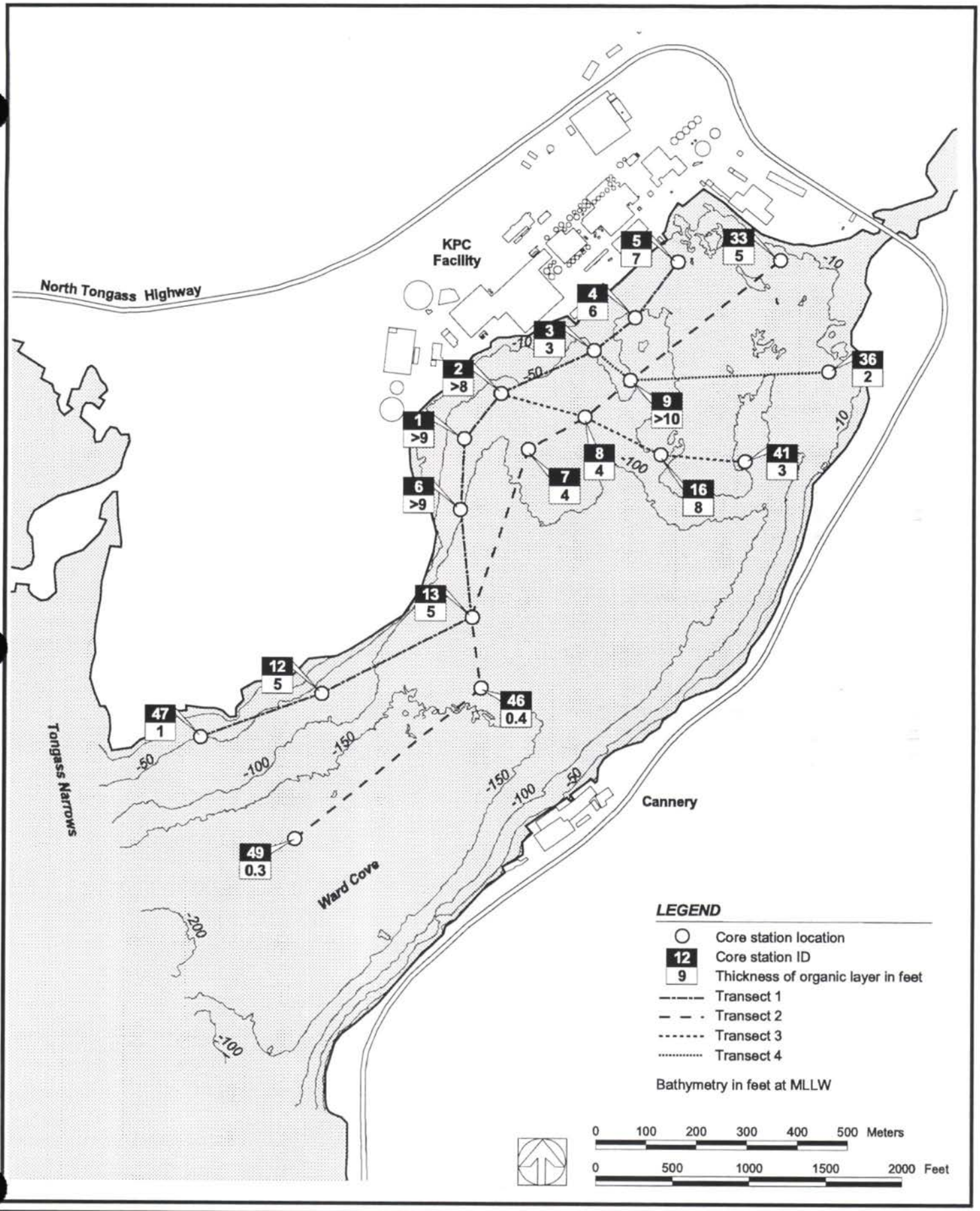

Figure 4-20. Ward Cove core stations, transects, and thickness of organic-rich sediments.

CB0W-16-01 06/15/98 GIS

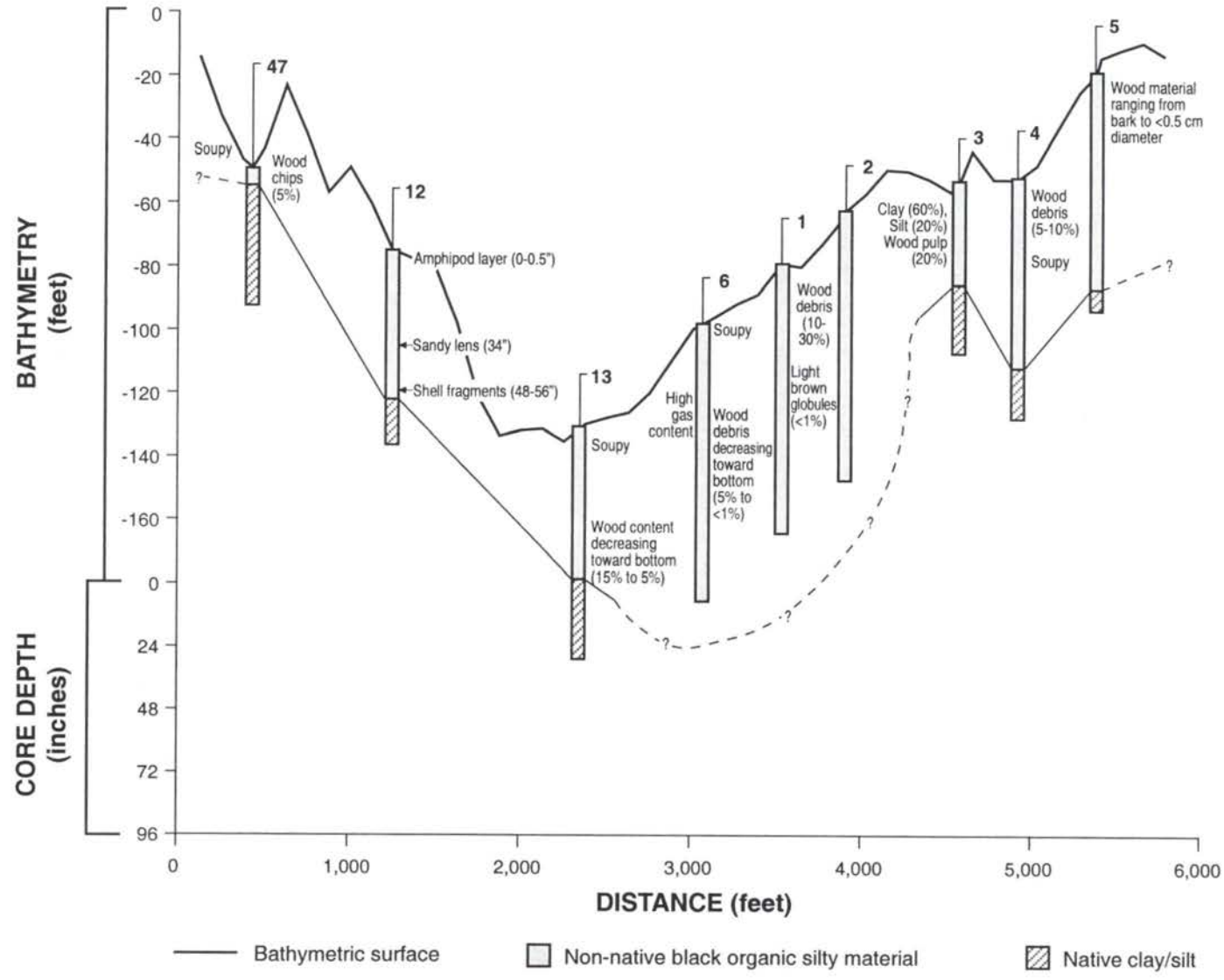

Figure 4-21. Profile of Transect 1.

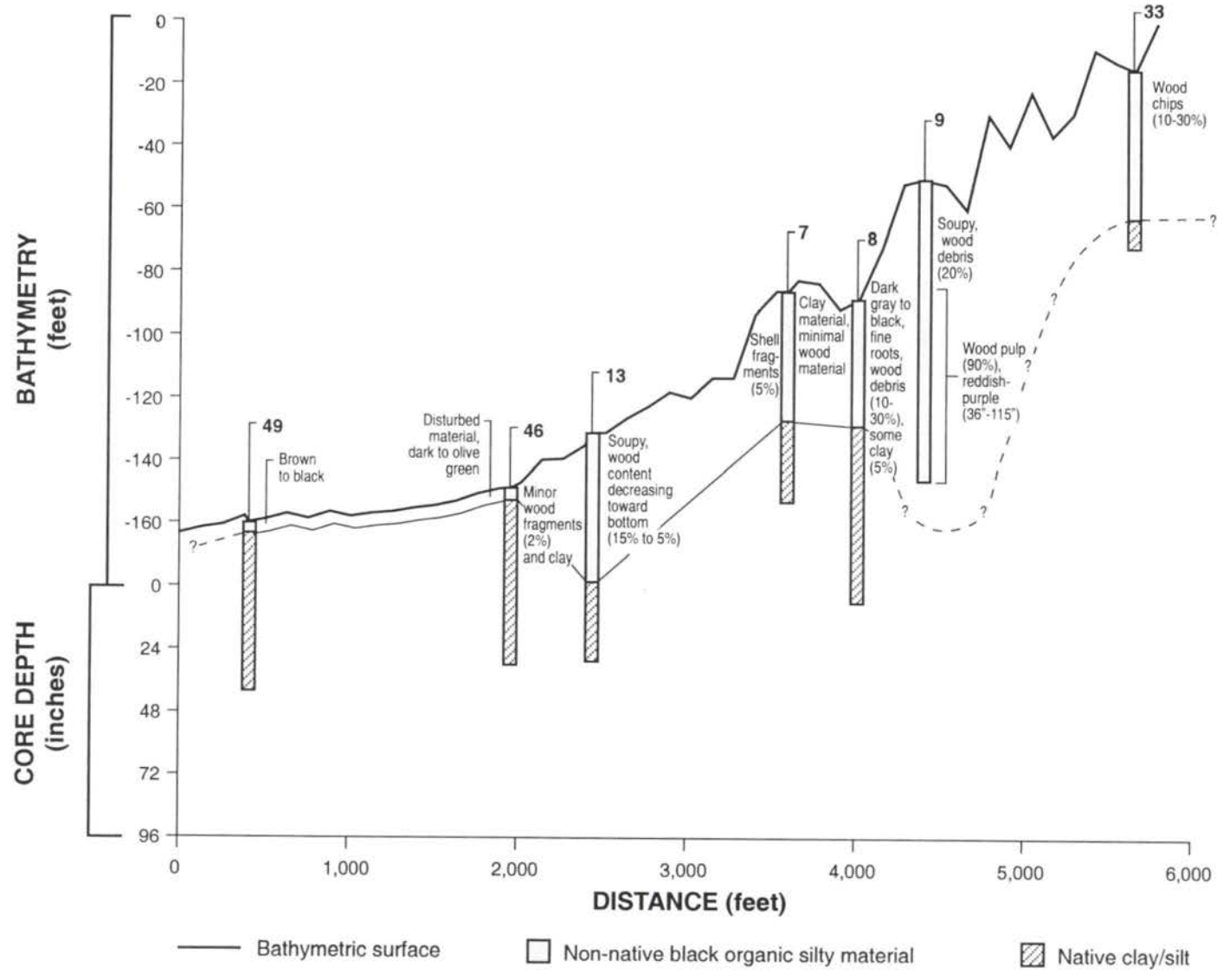

Figure 4-22. Profile of Transect 2.

8600BOW.001 1602 05/11/99 WA

Two transects perpendicular to the north shore and the KPC facility, T-3 and T-4, were also prepared in cross-section (Figures 4-23 and 4-24). Both of these transects indicate that the thickness of organic-rich sediments decreases away from the facility, with the thickness of impacted sediments ranging from greater than 10 ft near the facility to 2–3 ft near the far shore at Stations 36 and 41, respectively. The presence and density of logs are not included in these cross-sections. The collection of cores near areas of high log density (Figure 3-2) was difficult, requiring station repositioning and multiple core deployments.

In October 1995, ENSR collected 11 sediment cores in Ward Cove for a study of sediment depositional trends, which was required by NPDES Permit No. AK-000092-2 (ENSR 1996b). One of the primary objectives of the sediment coring program was to characterize the thickness of the upper sediment layer, which was expected to be rich in organic solids. The results of the ENSR study are summarized in Appendix E. The ENSR study found two types of organic material: wood debris and black, silty organic material. The black, silty organic material was observed to range in thickness from 2 to 25 in., and was generally less than 12 in. The combined thickness of the black silty layer and the wood debris layer was generally less than 2 ft, but reached a thickness of 4 ft near the KPC facility.

The results of the ENSR study are not entirely consistent with the results of the Exponent study. Although the studies covered slightly different areas, they did overlap in places. In general, the thickness of the organic-rich layer determined by Exponent is greater than the thickness of the organic-rich layer determined by ENSR. This difference in findings may be due to the different core sampling techniques used and the nature of the organic-rich material. ENSR used a gravity corer to collect samples and stated that it was unlikely that all of the black, silty organic material was recovered during coring, because the core catcher in the tip of the coring device likely restricted the flow of water and suspended material into the core tube as it descended into the sediment. Exponent used a piston corer to collect samples and may have achieved greater core retention.

4.3.2 Bulk Chemistry of Subsurface Sediments

The concentrations of chemicals measured in subsurface sediments in Ward Cove are summarized in Table 4-3. Detailed core data are provided in Tables A1-6 through A1-9, Appendix A1. Nearly all of the bulk chemistry data reflect the properties of the organic-rich surface sediments affected by KPC activities. Sediment horizons sampled generally represented 1-m (39-in.) depth intervals, unless a distinct change in sediment texture was encountered. Horizons that represented native sediments are listed at the end of Tables A1-6 through A1-9 in Appendix A1, and a comparison of native and non-native sediment is presented in Table 4-4. With the exception of dioxins/furans, horizons for individual cores were analyzed separately. Dioxins and furans were analyzed as five composited sediment samples (Table A1-9). Figure 2-7 presents the compositing strategy for the five dioxin/furan samples. Stations 46 and 49 are not included in the discussion of chemical distributions in organic-rich sediments because a detectable surficial, organic-rich layer was not observed at these stations.

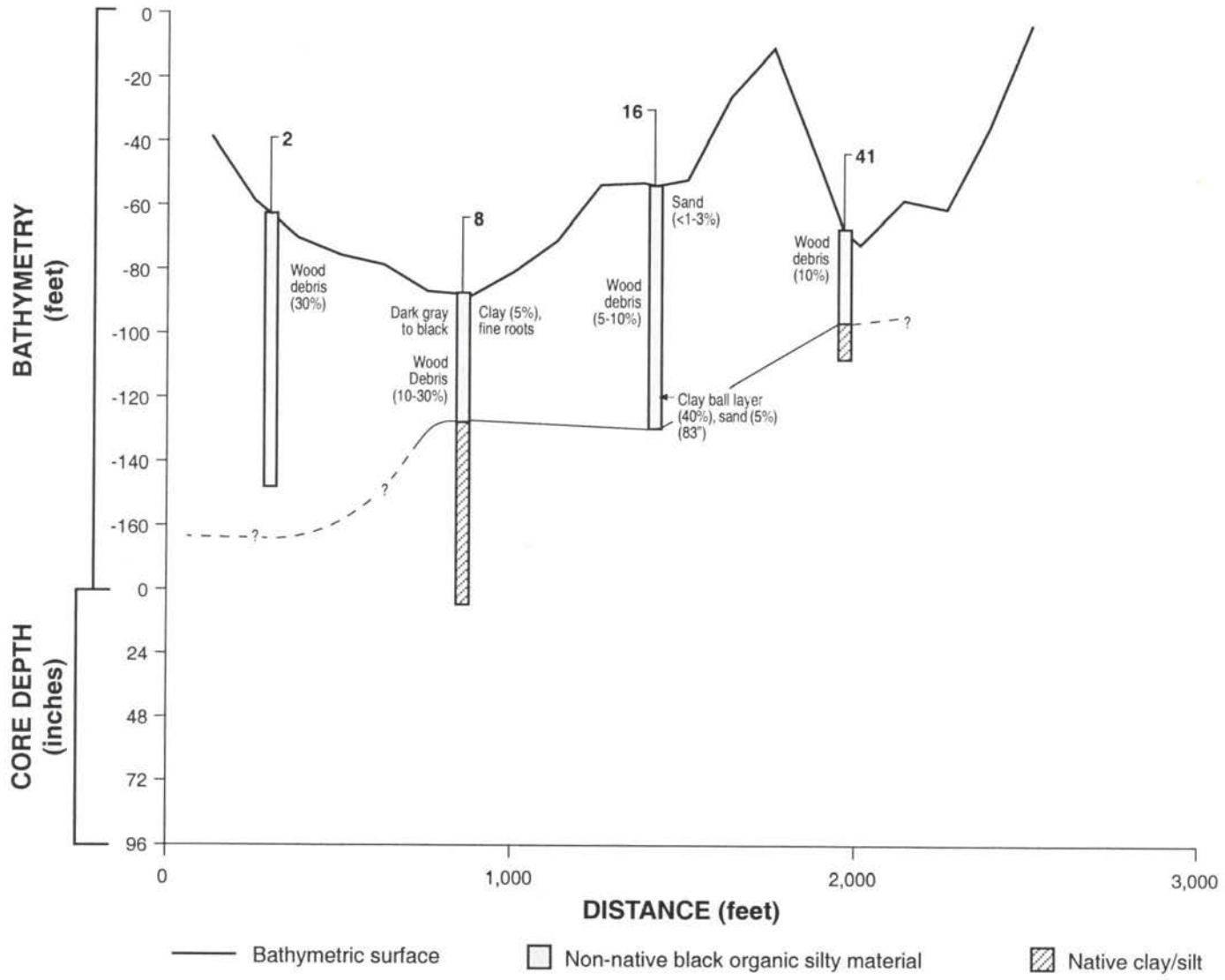

Figure 4-23. Profile of Transect 3.

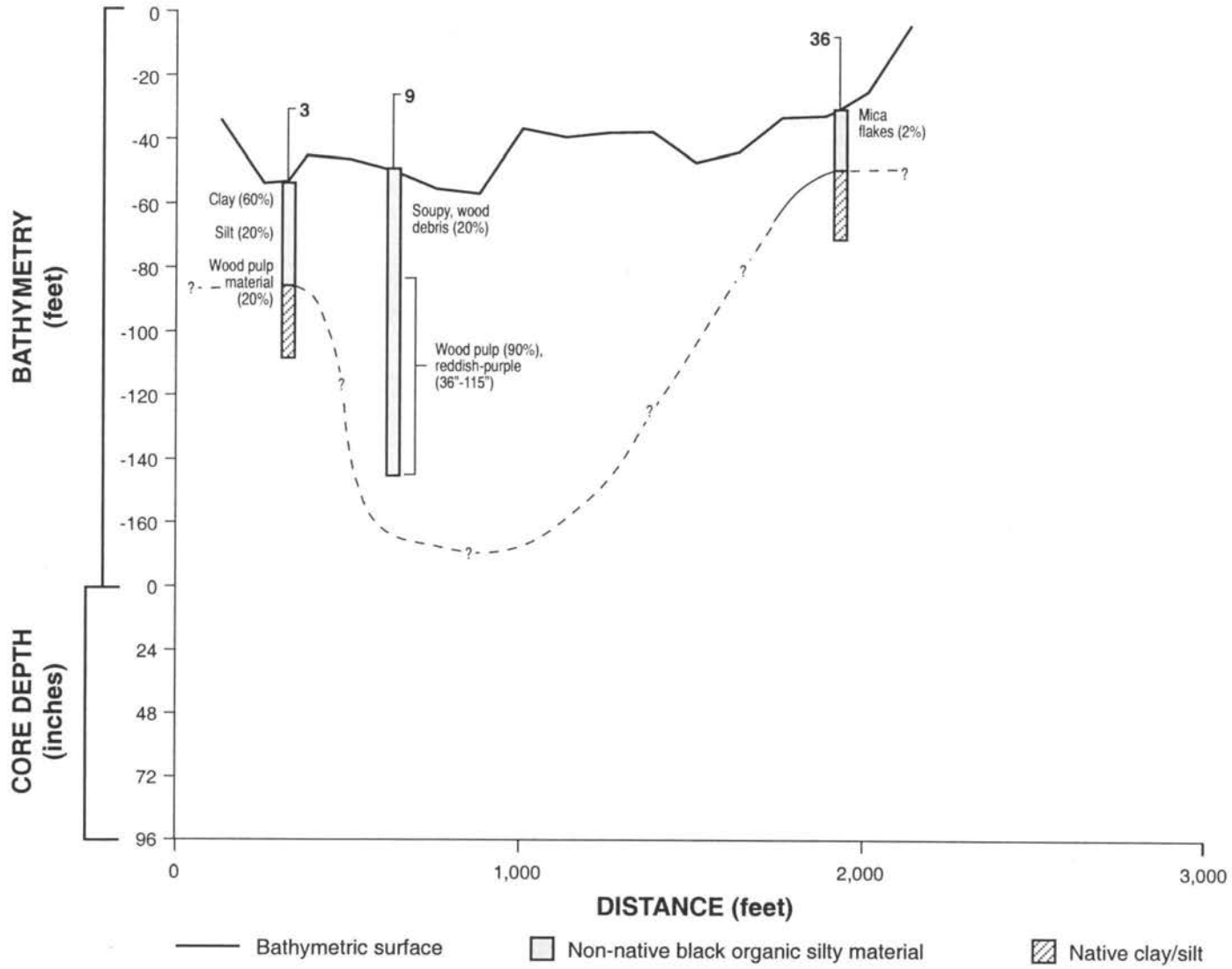

Figure 4-24. Profile of Transect 4.

TABLE 4-3. SUMMARY OF SUBSURFACE SEDIMENT DATA COLLECTED IN WARD COVE IN 1997 (EXCLUDING NATIVE SEDIMENTS)

Analyte	Concentration Range	Median	Number of Detected Values	Number of Samples	Frequency of Detection (percent)	Station with Maximum Concentration	Interval of Maximum (in.)	
							Upper Depth	Lower Depth
Conventional Analytes								
Total ammonia (mg/kg)	1.6 - 4,200	330	33	33	100	6	79	105
Biochemical oxygen demand 5-day test (g/kg)	3.0 - 120	7.5	33	33	100	6	0	39
Chemical oxygen demand (g/kg)	1.3 - 140	7.8	33	33	100	6	0	39
Total sulfide (mg/kg)	290 - 55,000	2,700	32	32	100	16	79	91
Total organic carbon (percent)	10 - 40	31	33	33	100	1	39	79
Gravel (percent)[a]	0.5 - 61	7.4	33	33	100	5	39	70
Sand (percent)								
1.0–2.0 mm	1.3 - 13	5.4	33	33	100	2	39	79
0.50–1.0 mm	1.3 - 33	6.4	33	33	100	9	39	79
0.25–0.50 mm	2.7 - 37	9.5	33	33	100	9	39	79
0.125–0.25 mm	1.7 - 19	7.9	33	33	100	36	0	22
0.062–0.125 mm	1.2 - 24	7.6	33	33	100	36	0	22
Silt (percent)	4.8 - 61	26	33	33	100	7	0	39
Clay (percent)	8.9 - 37	20	33	33	100	6	0	39
Total solids (percent of wet weight)	11 - 30	19	33	33	100	36	0	22
Metals (mg/kg)								
Cadmium	0.36 - 4.3	2.0	33	33	100	8	0	39
Total mercury	0.2 *U* - 0.7	0.2	7	33	21	4	0	39
Zinc	35 - 224	120	33	33	100	9	0	39
Phenols (µg/kg)								
Phenol	54 - 4,700	340	33	33	100	6	0	39
4-Methylphenol	180 - 78,000	3,300	33	33	100	6	0	39

TABLE 4-3. (cont.)

Analyte	Concentration Range	Median	Number of Detected Values	Number of Samples	Frequency of Detection (percent)	Station with Maximum Concentration
Dioxins and Furans (ng/kg)						
2,3,7,8-Tetrachlorodibenzo-*p*-dioxin	0.6 *U* - 1.3	0.7	0	5	0	--
1,2,3,7,8-Pentachlorodibenzo-*p*-dioxin	0.96 *U* - 1.6	1.4	0	5	0	--
1,2,3,4,7,8-Hexachlorodibenzo-*p*-dioxin	1.0 - 1.5[b]	1.3	4	5	80	D
1,2,3,6,7,8-Hexachlorodibenzo-*p*-dioxin	2.0 - 4.7	3.7	5	5	100	D
1,2,3,7,8,9-Hexachlorodibenzo-*p*-dioxin	1.6 *U* - 3.3	2.3	2	5	40	A
1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin	49 - 86	72	5	5	100	A
Octachlorodibenzo-*p*-dioxin	390 - 670	530	5	5	100	A
Total tetrachlorodibenzo-*p*-dioxins	17 - 61	46	5	5	100	B
Total pentachlorodibenzo-*p*-dioxins	4.4 - 21	14	5	5	100	D
Total hexachlorodibenzo-*p*-dioxins	17 - 44	35	5	5	100	D
Total heptachlorodibenzo-*p*-dioxins	120 - 190	180	5	5	100	A
2,3,7,8-Tetrachlorodibenzofuran	3.1 *U* - 4.7	4.3	0	5	0	--
1,2,3,7,8-Pentachlorodibenzofuran	0.66 - 0.89	0.9	2	5	40	B
2,3,4,7,8-Pentachlorodibenzofuran	0.87 *U* - 1.6	1.4	3	5	60	B
1,2,3,4,7,8-Hexachlorodibenzofuran	2.0 *U* - 6.7	4.7	0	5	0	--
1,2,3,6,7,8-Hexachlorodibenzofuran	0.88 - 1.9	1.9	4	5	80	B/C
1,2,3,7,8,9-Hexachlorodibenzofuran	0.63 *U* - 1.8	1.6	0	5	0	--
2,3,4,6,7,8-Hexachlorodibenzofuran	0.91 *U* - 2.1	1.5	3	5	60	A
1,2,3,4,6,7,8-Heptachlorodibenzofuran	14 - 29	18	5	5	100	A
1,2,3,4,7,8,9-Heptachlorodibenzofuran	0.82 *U* - 2.2	1.5	2	5	40	A
Octachlorodibenzofuran	30 - 46	39	5	5	100	C
Total tetrachlorodibenzofurans	7.7 - 23	20	5	5	100	A/D
Total pentachlorodibenzofurans	4.1 - 17	15	5	5	100	A
Total hexachlorodibenzofurans	14 - 39	27	5	5	100	A
Total heptachlorodibenzofurans	45 - 100	61	5	5	100	A
Dioxin and furan toxic equivalent concentration[c]	2.7 - 5.1	4.6	5	5	100	A
Dioxin and furan toxic equivalent concentration[d]	1.4 - 3.3	2.9	5	5	100	D

Note: Results are presented on a dry weight basis unless noted otherwise.

Concentrations for conventional analytes and organic compounds are rounded to two significant figures. Concentrations for metals are rounded to three significant figures if over 10 and two significant figures if less than 10.

Field replicates were treated as unique data points and the results were not averaged.

Medians were calculated using the detection limits for those congeners that were undetected.

-- - not applicable; the analyte was not detected at any station

PAH - polycyclic aromatic hydrocarbon

TABLE 4-3. (cont.)

RPC - relative potency concentration

U - undetected at concentration listed

[a] When grain-size distribution is determined by the analytical laboratory, the term "gravel" is a designation for a specific size fraction in the sediment. This verbiage does not mean that the sediment is gravel. In some shallower parts of the Cove, the "gravel" size fraction could consist of wood debris and probably includes organic material.

[b] At least one detection limit exceeded the concentration of the indicated maximum detected value.

[c] Detection limits are included in the sum at half their value.

[d] Detection limits are excluded from the sum.

TABLE 4-4. COMPARISON OF NATIVE AND NON-NATIVE SUBSURFACE SEDIMENT DATA COLLECTED IN WARD COVE IN 1997

Analyte	Native Sediment (4 samples) Concentration Range	Native Sediment (4 samples) Frequency of Detection (percent)	Non-native Sediment (33 samples) Concentration Range	Non-native Sediment (33 samples) Frequency of Detection (percent)
Conventional Analytes				
Total ammonia (mg/kg)	8.6–180	100	1.6–4,200	100
Biochemical oxygen demand 5-day test (g/kg)	0.2 *U*–2.1	75	3.0–120	100
Chemical oxygen demand (g/kg)	0.2–5.4	100	1.3–140	100
Total sulfide (mg/kg)	3.3–770	100 [a]	290–55,000	100
Total organic carbon (percent)	0.36–12	100	10–40	100
Gravel (percent)[b]	0.1–37	100	0.5–61	100
Sand (percent)				
1.0–2.0 mm	0.3–6.6	100	1.3–13	100
0.50–1.0 mm	0.5–5.5	100	1.3–33	100
0.25–0.50 mm	2.7–8.3	100	2.7–37	100
0.125–0.25 mm	3.8–13	100	1.7–19	100
0.062–0.125 mm	9.5–19	100	1.2–24	100
Silt (percent)	16–69	100	4.8–61	100
Clay (percent)	6–30	100	8.9–37	100
Total solids (percent of wet weight)	23–68	100	11–30	100
Metals (mg/kg)				
Cadmium	0.11–3.4	100	0.36–4.3	100
Total mercury	0.2 *U*	0	0.2 *U*–0.7	21
Zinc	56.8–96.3	100	35–220	100
Phenols (µg/kg)				
Phenol	10 *U*–150	75	54–4,700	100
4-Methylphenol	10 *U*–350	50	180–78,000	100

Note: Results are presented on a dry weight basis unless noted otherwise.

Concentrations for conventional analytes and organic compounds are rounded to two significant figures. Concentrations for metals are rounded to three significant figures if over 10 and two significant figures if less than 10.

U - undetected at concentration listed

[a] Only three native samples were analyzed for sulfide.

[b] When grain-size distribution is determined by the analytical laboratory, the term "gravel" is a designation for a specific size fraction in the sediment. This verbiage does not mean that the sediment is gravel. In some shallower parts of the Cove, the "gravel" size fraction could consist of wood debris and probably includes organic material.

Native and non-native sediments are distinguished principally by the difference in TOC content and the sediment properties (e.g., lower solids content, increased BOD and COD) and chemicals (increased ammonia, sulfide, and phenols) associated with TOC enrichment. The ranges of grain sizes observed in native and non-native sediment are similar except in the range of medium to coarse sand, suggesting that the organic material in non-native sediment is found principally in this size range.

4.3.2.1 Total Organic Carbon

The TOC content of sediments affected by the KPC facility typically ranged from 20 to 40 percent. Underlying native sediments contained 0.36 to 12 percent TOC. Consistent with the surface distribution of TOC (Figure 4-3), the highest values of TOC (i.e., greater than 30 percent) in subsurface sediments were found immediately offshore from the KPC facility near Outfall 001 (Stations 1, 2 and 6) and Outfall 002 (Station 5). TOC levels greater than 30 percent were also found in subsurface sediments at Stations 4, 9, and 16, suggesting that the TOC content from core samples was greater than surface concentrations at these locations. The highest TOC concentration in Ward Cove (39.5 percent) occurred at Station 1.

4.3.2.2 Total Ammonia

The highest concentrations of total ammonia in core sediment samples occurred at Station 6, with values ranging from 1,600 to 4,200 mg/kg. Station 1 also had high ammonia concentrations, with values ranging from 770 to 1,400 mg/kg. Two other stations had concentrations greater than 500 mg/kg: Stations 3 (880 mg/kg) and 12 (500–690 mg/kg). The distribution of elevated concentrations of ammonia in core sediment samples was generally consistent with the surficial distributions (Figures 4-4 and 4-5); however, ammonia concentrations in subsurface sediment samples are generally greater than in surface sediment samples, which exceeded 690 mg/kg at only one station (a field replicate at surface Station 2). Stations 5, 9, 16, 36, and 41 all had concentrations of total ammonia in subsurface sediment samples below 100 mg/kg.

4.3.2.3 Total Sulfide

Located approximately 1,000 ft from the KPC facility in the middle of Ward Cove, Station 16 had the highest concentrations of total sulfide (26,000–55,000 mg/kg) in subsurface sediment samples. Elevated concentrations of total sulfide in the center of the Cove are consistent with the surface sediment data for 1996 and 1997 and correspond to an area of high log density. With the exception of Stations 7 and 36, all other stations in the Cove had total sulfide concentrations ranging from 1,300 to 7,700 mg/kg. Stations 7 and 36 had relatively low sulfide concentrations of 340 and 740 mg/kg, respectively. The range of total sulfide concentrations in subsurface sediments is generally consistent with

the range in concentrations observed for surface sediments in 1997. As was the case for surface sediments, no clear distribution pattern emerged.

4.3.2.4 Biochemical Oxygen Demand

BOD concentrations ranged between 5 and 10 g/kg throughout most of Ward Cove. The highest BOD concentrations (23–120 g/kg) occurred at Station 6, offshore of Outfall 001. Stations 16 and 33 also had elevated BOD concentrations, ranging from 20–39 g/kg. The range of BOD concentrations in surface sediments (Figures 4-8 and 4-9) was similar to the range of BOD concentrations in sediment core samples. BOD concentrations in native sediments ranged from 0.2 *U* to 2.1 g/kg.

4.3.2.5 Cadmium

Cadmium concentrations displayed a limited concentration range in core sediment samples (0.36–4.3 mg/kg). The highest concentrations of cadmium in core samples were found at Stations 8 (4.3 mg/kg; located approximately 500 ft offshore of the facility) and 12 (4.1 mg/kg; located along the north shore west of the KPC facility). Cadmium concentrations in native sediments ranged from 0.11 to 3.4 mg/kg. Cadmium concentrations in subsurface sediments were systematically lower than concentrations observed in surface sediments (Figure 4-12).

4.3.2.6 Total Mercury

Total mercury was undetected in subsurface sediments at most locations in Ward Cove, at a detection limit of 0.2 mg/kg. The highest detected concentration (0.7 mg/kg) occurred at Station 4, approximately 200 ft offshore from the facility. Total mercury was also detected at concentrations near the detection limit (0.2 to 0.5 mg/kg) at Stations 1, 8, 12, and 13.

4.3.2.7 Zinc

Like cadmium and mercury, zinc displayed a limited concentration range in core sediment samples from the Cove, typically ranging from 50 to 200 mg/kg. The highest concentration of zinc in the Cove (224 mg/kg) was found at Station 9, located approximately 500 ft offshore from the KPC facility. Station 5 (offshore Outfall 002) had zinc concentrations less than 100 mg/kg. Zinc concentrations in native sediments ranged from 57 to 96 mg/kg. Zinc concentrations in subsurface sediments were lower than concentrations in surface sediments (Figure 4-14).

4.3.2.8 Phenol and 4-Methyphenol

As was the case for surface sediments, the concentrations of 4-methylphenol and phenol generally covary in core sediment samples, although 4-methlyphenol concentrations were typically an order of magnitude greater than phenol concentrations. The highest concentrations of 4-methylphenol in the Cove (ranging from 26,000 to 78,000 μg/kg) were found at Stations 1 and 6, located offshore of Outfall 001. Station 2 had 4-methylphenol concentrations ranging from 9,100 to 21,000 μg/kg. In general, the spatial distribution of elevated concentrations of 4-methylphenol in core samples was consistent with that observed for surface samples (Figures 4-15 and 4-16), with elevated concentrations localized in the vicinity of the KPC facility. However, the concentration of 4-methylphenol in subsurface sediment samples was systematically greater than the concentration measured in surface sediments. The concentration of 4-methylphenol in native sediments ranged from 10 *U* to 350 μg/kg.

4.3.2.9 2,3,7,8-TCDD and TCDF Toxic Equivalent Concentrations

As explained in the introduction to this section, analyses for dioxin and furan congeners were conducted on five composite subsurface sediment samples. As was the case for surface sediments, 1,2,3,4,6,7,8,9-OCDD was the dioxin/furan compound present at the highest concentrations, ranging from 390 to 670 ng/kg. The most toxic dioxin/furan compound, 2,3,7,8-TCDD, was undetected in all five of the sediment core composites at detection limits ranging from 0.6 to 1.3 ng/kg. The concentrations of 1,2,3,4,6,7,8,9-OCDD in subsurface sediments were lower than the concentrations measured in surface sediments (Table A1-4), while TCDD was generally undetected in both types of sediment samples.

TCDD TECs were less than 6 ng/kg for all five of the sediment core composites. The lowest TCDD TEC (2.7 ng/kg) occurred in the composite for Stations 16, 36, and 41. Composites representing stations directly offshore of the facility had TCDD TECs ranging from 4.3 to 5.1 ng/kg. The composite for Stations 12 and 13 (located along the north shore, west of the KPC facility) had a TCDD TEC of 4.6 ng/kg. TECs in subsurface sediment samples were less than TECs in surface sediments (Figures 4-18 and 4-19).

4.3.3 Summary of Subsurface Core Properties and Chemistry

The physical appearance and chemical data from the 18 core samples collected in Ward Cove indicate the following:

- Releases from the KPC facility, including log handling, have resulted in a black, organic-rich layer of sediment that contains wood debris and is generally 4–9 ft thick. This layer of sediment is generally found near the head of the Cove offshore of the KPC facility and along the north shore.

- The concentrations of metals and dioxin/furan congeners in subsurface sediments are lower than those in surface sediments.
- The concentrations of organic carbon, total sulfide, and BOD in subsurface sediments are similar to those in surface sediments.
- The concentrations of ammonia, phenol, and 4-methylphenol in subsurface sediments are greater than those in surface sediments.

4.4 TISSUE

Two sources of tissue data were used in screening analyses for CoPCs in the human health and ecological assessments:

- Tissue concentrations measured in previous investigations
- Estimated tissue concentrations derived through application of biota-sediment accumulation factors (BSAFs) to maximum concentrations of chemicals detected in sediments in the current investigation.

Available data on concentrations of PCDDs/Fs and mercury in tissues of fish and shellfish are described in Section 2.2.3 and summarized in Appendix D. The following section describes how tissue concentrations were estimated from sediment concentrations, and a subsequent section describes the application of measured and estimated values in identifying CoPCs.

4.4.1 Estimated Tissue Concentrations

Estimated concentrations of chemicals in fish, crabs, bivalves, shrimp, and gastropods were determined using maximum sediment concentrations measured in Ward Cove in the Phase 1 and 2 investigations. These estimated tissue concentrations were used (together with measured tissue concentrations for PCDD/F [TECs] and mercury) to determine if the chemicals present in site sediments pose a potential risk to humans and ecological receptors (Sections 6 and 7). BSAFs were used to predict tissue concentrations based on the maximum sediment concentrations for each detected chemical that had adequate toxicity data for use in the assessments.

BSAF values have been determined for various metals, polar organic compounds, and nonpolar organic compounds from field studies that related tissue and sediment concentrations for the various analytes (PTI 1995a,b). BSAF values were obtained from tissue concentration data presented in Boese and Lee (1992) and PTI (1995a,b). In this report, the term BSAF, when applied to metals, is synonymous with the term bioaccumulation factor and expresses the relationship between tissue concentration (not lipid normalized) and sediment concentration (not TOC normalized). For nonpolar organic compounds, BSAF, as used in this report, expresses the relationship of the lipid-normalized tissue

concentration to the TOC-normalized sediment concentration. A summary of estimated tissue concentrations is provided in Table 4-5.

The algorithm used to estimate tissue concentrations is provided in Table 4-6. Conservative assumptions were incorporated into these calculations to provide a protective estimate of potential risk. The highest sediment concentration found at Ward Cove during the Phase 1 or Phase 2 studies was used for each analyte with one exception: the highest (TOC-normalized) concentrations of carcinogenic PAHs[5], anthracene, phenol, and zinc were present at stations near the cannery (Stations 24 and 25) and the state airplane ramp (Station 23). Concentrations of site-related chemicals decrease at stations in the middle of Ward Cove and then increase again near the cannery and airplane ramp (Figure 4-25 shows the distribution of concentrations of carcinogenic PAHs). Therefore, because this investigation and cleanup focused on contamination related to KPC, the highest values measured in sediments near the KPC facility were used to estimate tissue concentrations for these compounds. In the ecological assessment, tissue concentrations estimated on the basis of mean sediment concentrations were also evaluated. The BSAF values that were used to estimate tissue concentrations for PAHs and 2,3,7,8-TCDD TECs represent the 95 percent upper confidence limit on the mean of all BSAFs for those compounds (PTI 1995a). For metals, the mean BSAF is used to estimate bioaccumulation of CoPCs in prey tissue because there were insufficient data reported in the compilation by Boese and Lee (1992) to calculate a statistically relevant upper 95 percent confidence limit on the mean. Values for lipids and solids fractions in each tissue type represent the average of values reported in the literature, if available, or single values where only limited data were identified.

For nonpolar organic compounds, the BSAF values are established using TOC-normalized sediment concentrations and lipid-normalized tissue concentrations (PTI 1995a). Therefore, measured concentrations of nonpolar organic CoPCs in Ward Cove sediment (e.g., PAHs and the TEC for PCDDs/Fs) were TOC normalized prior to application of the BSAF. BSAF values for metals and polar organic compounds are not based on TOC- or lipid-normalized data, and corresponding adjustments were not required to determine estimated tissue concentrations for these analytes. The convention of normalizing sediment concentrations of nonpolar organic compounds to TOC reflects the finding that such compounds preferentially bind to the organic fraction of sediments and to lipids within biological tissue. Adjustments using station-specific TOC measurements are typically applied. Because TOC concentrations in sediments at the site are elevated over those typically seen at many sites, uncertainties exist regarding the degree to which TOC in the sediments will demonstrate binding properties expected with other organic carbon sources. At the request of EPA, station-specific TOC concentrations were used in adjusting nonpolar organic compound concentrations where these TOC values were less than 10 percent. Where station-specific values were 10 percent or greater, a TOC value of 10 percent was used.

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[5] As described in Section 2.3.1.5, carcinogenic PAHs are presented as the RPC with undetected carcinogenic PAHs included in calculations using one-half the detection limit.

TABLE 4-5. ESTIMATED TISSUE CONCENTRATIONS OF CHEMICALS DETECTED IN WARD COVE SEDIMENTS IN 1996 AND 1997

Chemical	Maximum Sediment Concentration[a]		Fish Tissue[b]		Crab Tissue[c]		Bivalve Tissue[d]		Shrimp Tissue[e]		Gastropod Tissue[f]	
	mg/kg dw	TOC Fraction	BSAF	mg/kg ww	BSAF	mg/kg ww	BSAF	mg/kg ww	BSAF	mg/kg ww	BSAF	mg/kg ww
Metals and Organometallic Compounds (maximum sediment concentration)												
Arsenic	39	NA	0.12 [g]	0.12 [h]	0.02 [g]	0.022 [h]	0.7 [g]	0.50 [h]	--	--	0.7 [g]	0.50 [h]
Cadmium	7.3	NA	2 [g]	3.7	3 [g]	5.7	7.5 [i]	9.9	44 [i]	71	39 [i]	51
Total mercury (sediments; methylmercury in tissues)	0.7	NA	0.38 [g]	0.067	0.13 [g]	0.024	4.5 [i]	0.57	1 [i]	0.15	2 [i]	0.25
Zinc[j]	396	NA	5 [g]	500	3.2 [g]	330	7.3 [i]	520	0.2 [h]	14	5 [i]	356
Metals and Organometallic Compounds (mean sediment concentration: ERA only)												
Arsenic	22	NA	0.12 [g]	0.066	0.02 [g]	0.013	0.7 [g]	0.28	--	--	0.7 [g]	0.28
Cadmium	3.5	NA	2 [g]	1.8	3 [g]	2.7	7.5 [i]	4.7	44 [i]	34	39 [i]	24.6
Total mercury (sediments; methylmercury in tissues)	0.1	NA	0.38 [g]	0.0095	0.13 [g]	0.0034	4.5 [i]	0.082	1 [i]	0.022	2 [i]	0.036
Zinc	190	NA	5 [g]	240	3.2 [g]	160	7.3 [i]	250	0.2 [h]	6.7	5 [i]	170
Organic Compounds												
Phenol[j]	0.91	0.10	0.63 [k]	0.47	--	--	--	--	--	--	--	--
4-Methylphenol	17	0.10	0.63 [k]	8.8	--	--	--	--	--	--	--	--
PCDD/F (TEC)												
Max. Sediment Conc. (ERA)	4.6×10^{-5}	0.10	1.04 [l]	4.9×10^{-5} [m]	1.04 [l]	6.7×10^{-6}	0.9 [i]	1.2×10^{-5}	0.7 [i]	5.5×10^{-6}	0.9 [i]	6.2×10^{-6}
Mean Sediment Conc. (ERA)	1.7×10^{-5}	0.10	1.04 [l]	1.8×10^{-5} [m]	1.04 [l]	2.5×10^{-6}	0.9 [i]	4.3×10^{-6}	0.7 [i]	2.0×10^{-6}	0.9 [i]	2.3×10^{-6}
Max. Sediment Conc. (HHRA)	4.6×10^{-5}	0.10	1.04 [l]	3.9×10^{-5} [n]	1.04 [l]	6.7×10^{-6}	0.9 [i]	1.2×10^{-5}	0.7 [i]	5.5×10^{-6}	0.9 [i]	6.2×10^{-6}
PAHs[o]												
Carcinogenic PAH												
HHRA (RPC)[j]	0.41	0.10	NA	0	0.63 [l]	0.036	0.6 [l]	0.072	0.6 [l]	0.044	0.6 [l]	0.039
ERA (maximum)[j]	0.41	0.10	NA	0	0.63 [l]	0.036	0.6 [l]	0.072	0.6 [l]	0.044	0.6 [l]	0.039
ERA (mean)[j]	0.16	0.10	NA	0	0.63 [l]	0.014	0.6 [l]	0.028	0.6 [l]	0.017	0.6 [l]	0.015
Fluoranthene	2.2	0.10	NA	0	0.63 [l]	0.19	0.6 [l]	0.39	--	--	--	--
Pyrene	1.8	0.10	NA	0	0.63 [l]	0.16	0.6 [l]	0.32	--	--	--	--
Acenaphthene[j]	0.50	0.10	NA	0	0.63 [l]	0.044	0.6 [l]	0.088	--	--	--	--
Anthracene	0.26	0.10	NA	0	0.63 [l]	0.023	0.6 [l]	0.046	--	--	--	--
Fluorene	0.47	0.10	NA	0	0.63 [l]	0.041	0.6 [l]	0.083	--	--	--	--

Note: [ ] - values updated with 1997 data

-- - not available

BSAF - biota-sediment accumulation factor

dw - dry weight

ERA - ecological risk assessment

HHRA - human health risk assessment

NA - not applicable

PAH - polycyclic aromatic hydrocarbon

PCDD/F - polychlorinated dibenzo-*p*-dioxin and polychlorinated dibenzofuran

RPC - relative potency concentration for carcinogenic PAH

TEC - toxic equivalent concentration

TOC - total organic carbon

ww - wet weight

TABLE 4-5. (cont.)

[a] TOC assumed to be 10 percent where site-specific TOC was 10 percent or greater (see text). For undetected concentrations, one-half the detection limit was used in the RPC and TEC calculations.

[b] Fish tissue is assumed to be 25 percent solids based on U.S. EPA (1993b).

[c] Crab tissue is assumed to be 26 percent solids based on U.S. EPA (1993b). Lipid content of 1.4 percent is based on Sidwell (1981).

[d] Bivalve tissue is assumed to be 18 percent solids based on U.S. EPA (1993b). Lipid content of 2.8 percent is based on Ferraro et al. (1990).

[e] Shrimp tissue is assumed to be 22 percent solids based on average of pink, white, and brown shrimp reported in Sidwell (1981). Lipid content of 1.73 percent is based on Burkett (1995).

[f] Gastropod tissue is assumed to be 18 percent solids based on averaged data for snails, as reported in Sidwell (1981). Lipid content of 1.5 percent is based on averaged data for snails, as reported in Sidwell (1981).

[g] PTI (1995b).

[h] Estimated total arsenic concentrations are adjusted by 10 percent to reflect proportion of inorganic arsenic (ICF Kaiser 1996).

[i] Boese and Lee (1992).

[j] Concentrations are maximum sediment concentrations, except for phenol, carcinogenic PAHs (RPCs), anthracene, and zinc, which exclude higher sediment concentrations identified at locations remote from the site (i.e., Station 23 at the state airplane ramp and Stations 24 and 25 at the cannery; see Figure 4-25). For undetected PAH and PCDD/F concentrations, one-half the detection limit was used in the RPC and TEC calculations.

[k] BSAFs are not available for phenol or 4-methylphenol; BSAF for benzo[a]pyrene is used (PTI 1995b).

[l] PTI (1995a).

[m] For ecological receptors, assumptions are 70 percent consumption of herring with lipid content of 13.88 percent (Burkett 1995) and 30 percent consumption of rockfish with lipid content of 1.57 percent (Burkett 1995).

[n] For human health, assumptions are 30 percent consumption of rockfish with lipid content of 1.57 percent (Burkett 1995) and 70 percent consumption of salmon with lipid content of 11 percent (Sidwell 1981). Consumption percentage assumptions from Howe et al. (1995, 1996).

[o] BSAF for PAHs in shellfish from PTI (1995b) is used to estimate concentrations in crab, bivalve, shrimp, and gastropods. Fish are assumed not to bioaccumulate PAHs as a result of rapid metabolism (ATSDR 1989).

TABLE 4-6. ALGORITHM FOR ESTIMATED TISSUE CONCENTRATIONS

Tissue Concentration

$$\text{Metals: } C_f = BSAF \times C_s \times P_s$$

$$\text{Organic Compounds: } C_f = BSAF \times C_s \times \frac{f_l}{f_{oc}}$$

where:

C_f = analyte concentration in tissue (mg/kg wet weight)

BSAF = biota-sediment accumulation factor (unitless)

C_s = analyte concentration in sediment (mg/kg dry weight)

f_l = fraction of lipid in fish (unitless)

f_{oc} = fraction of organic carbon in sediment (unitless)

P_s = percent solids (unitless); applied to adjust dry weight BSAFs for metals to reflect wet weight tissue concentrations.

Assumptions

Tissue Type	Percent Solids[a]	Fraction Lipids[b] (f_l)
Fish (ERA)	0.25	0.102[c]
Fish (HHRA)	0.25	0.082[d]
Crab (ERA/HHRA)	0.26	0.014
Bivalve (ERA/HHRA)	0.18	0.028
Shrimp (ERA/HHRA)	0.22	0.017
Gastropods (ERA/HHRA)	0.18	0.015

Note: ERA - ecological risk assessment
HHRA - human health risk assessment

[a] Fraction solids data were obtained from U.S. EPA (1993b) and Sidwell (1981) and were applied in calculating wet weight tissue concentration estimates for metals only. Ecological risk calculations were based on dry weight concentrations (see text).

[b] Lipid corrections were made for nonpolar organic compounds.

[c] For ecological receptors, the lipids concentration in fish is based on 70 percent consumption of herring with 13.88 percent lipid (Burkett 1995) and 30 percent consumption of rockfish with 1.57 percent lipid (Burkett 1995).

[d] For human health, the lipids concentration in fish is based on 30 percent consumption of rockfish with 1.57 percent lipid (Burkett 1995) and 70 percent consumption of salmon with 11 percent lipid (Sidwell 1981). Consumption percentages based on Howe et al. (1995, 1996).

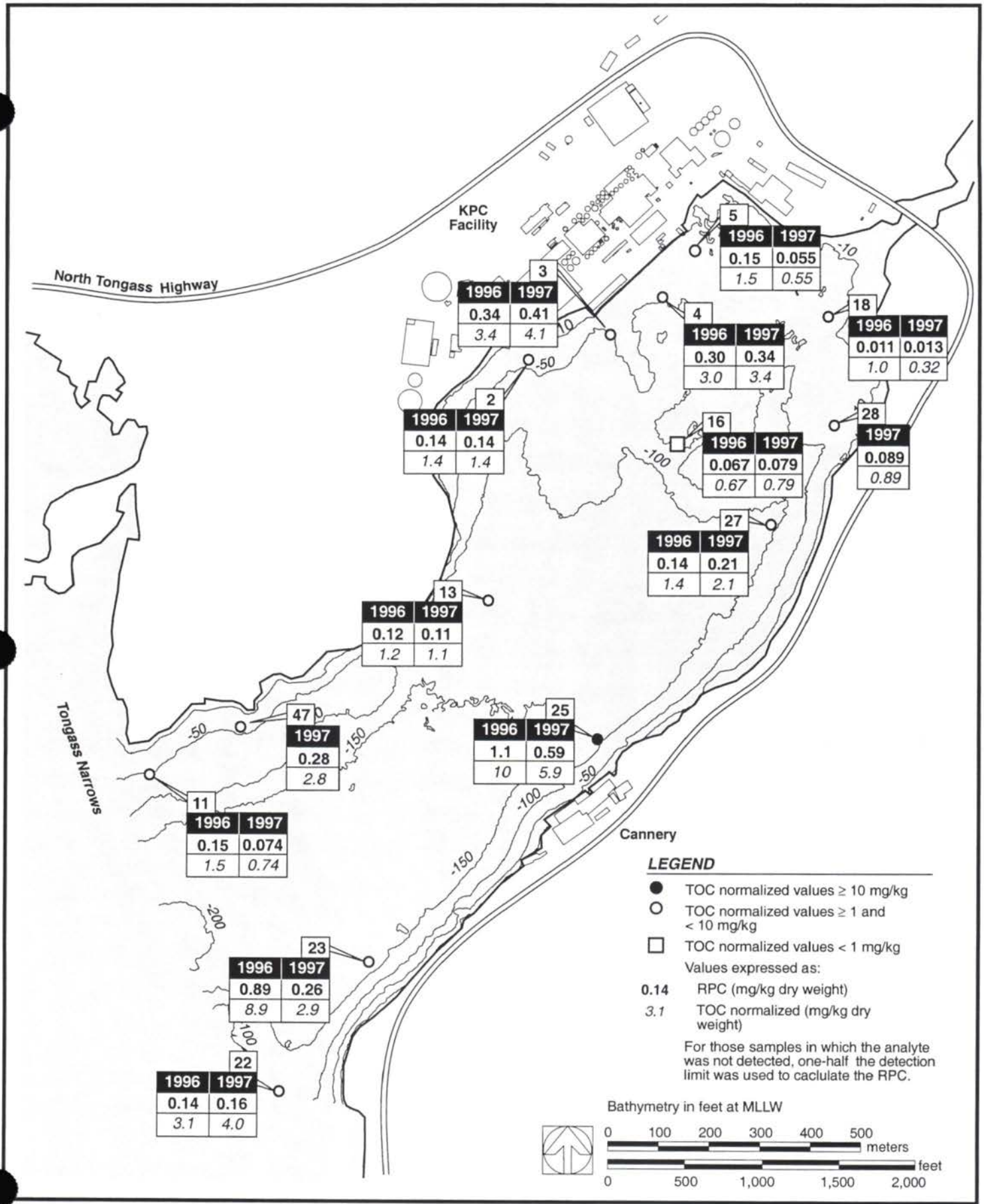

Figure 4-25. Concentrations of carcinogenic PAHs in Ward Cove in 1996 and 1997.

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The use of the 10 percent TOC adjustment is within the range of values suggested by EPA's Science Advisory Board, which suggested an apparent upper boundary of 10 to 12 percent organic carbon for TOC normalization (U.S. EPA 1992b). EPA's Science Advisory Board also identified the need for additional research to better define an upper-end limit in application of TOC normalization for nonpolar compounds (U.S. EPA 1992b). Moreover, the use of the 10 percent TOC assumption is within the range of the uncertainty in the values underlying key BSAF factors used in calculations at the site (i.e., K_{ow} and K_{oc} values for TCDD and PAHs). The use of a 10 percent TOC adjustment factor in the Ward Cove study was reviewed by scientists at two EPA laboratories, who determined that this approach was appropriate and protective (Keeley 1997a,b, pers. comm.)

Other lines of evidence suggest that this approach provides a conservative method to evaluate the binding properties of TOC in sediments. Scientific research suggests that most, if not all, of the TOC in site sediments would have the capacity to bind nonpolar organic chemicals. First, the surface area for binding is directly proportional to particle size. All visible fragments of wood were removed from the sample in the field, in keeping with sediment sampling guidance provided under the Puget Sound Estuary Protocols (U.S. EPA 1991c). Removal included (but was not limited to) any wood debris large enough to be removed without contaminating the sample. Thus, wood fragments that might not provide a representative surface area for binding were removed from the sample.

Furthermore, evaluation of the particle size distribution for Ward Cove sediment samples suggests that binding capacity in these samples would be high. As shown in Table A1-1 in Appendix A1, the percent fines (i.e., particles smaller than 0.062 mm) in the 1996 data set make up more than 50 percent of the total sample for all but six samples. In particular, the sample from Station W04, which was used as the basis for estimating tissue concentrations for PCDDs/Fs and PAHs in PTI (1996) has greater than 60 percent fines. Even if the 25.6 percent TOC measured at Station W04 was adjusted conservatively, assuming that the remaining 40 percent of TOC not represented as fines provides no surface area for binding, the resulting TOC would be 15.5 percent. Given these considerations and the consistently high percent fines identified in the Ward Cove sediments, use of a 10 percent TOC adjustment provides a conservative basis for TOC normalization of nonpolar organic concentrations in calculating tissue concentrations.

The TOC-normalized sediment concentrations were multiplied by the BSAF, and the resulting lipid-normalized tissue concentrations were corrected for lipid content to yield concentrations in the tissue. Concentrations of nonpolar CoPCs derived in this way are on a wet weight basis because the underlying BSAFs are based on wet weight tissue data. Wet weight concentrations are used in human health risk calculations, while dry weight concentrations are used in ecological risk calculations. For metals and polar organic compounds, however, underlying BSAFs are on a dry weight basis and correction for solids in tissue was completed to provide data on a whole (wet) weight basis for human health risk assessment purposes. All tissue concentrations were converted to dry weight

values for food-web exposure modeling, because food ingestion rates for receptors are estimated on a dry weight basis.

Uncertainties with the BSAF approach as applied in Ward Cove include the following:

- The presence of wood fibers overlying sediments in portions of Ward Cove may alter the assessment of how nonpolar organic compounds partition between sediments and organisms. Inclusion of wood fibers (e.g., lignin) in sediment samples may overestimate the amount of organic matter available for adsorption of nonpolar organic compounds. This uncertainty has been addressed by lowering the maximum acceptable TOC concentration at all stations to 10 percent.
- Use of a BSAF approach assumes steady-state equilibrium between the organism and sediment. For sessile organisms, there is a higher likelihood that this equilibrium exists. For motile organisms such as shrimp, crabs, gastropods, and fish, there is greater uncertainty that a steady-state equilibrium occurs and that a BSAF approach accurately predicts tissue concentrations.
- For sessile organisms that inhabit the upper sediment layer but feed on organisms or suspended organic matter at the sediment–water interface (i.e., some bivalves), there is uncertainty with the ability of the BSAF model to accurately predict tissue concentrations.
- In habitats where a constant interchange of water occurs (i.e., at the surface–water interface in tidally influenced marine coastal areas), a steady-state equilibrium may never be achieved. Therefore, uncertainty exists for the suitability of using a BSAF approach to predict tissue concentrations in biota.

The form of arsenic found in seafood is critical to evaluating potential adverse effects in consumers. Arsenic in seafood has long been recognized to occur primarily in organic forms that have reduced or negligible toxicity. Specifically, arsenic is present in almost all marine animal species chiefly as arsenobetaine with arsenocholine also occurring in shrimp tissues (Edmonds and Francesconi 1993; Phillips 1994). These stable compounds have been shown to be nontoxic in several studies (e.g., Eisler 1994). A review by Edmonds and Francesconi (1993) of arsenic forms in marine biota indicates that the inorganic arsenic concentrations ranged from 0.5 percent of total arsenic concentrations (where total arsenic concentrations were as high as 20 mg/kg) to 1 percent of total arsenic concentrations (where arsenic concentrations were low). EPA Region 10 assumed a 1 percent inorganic arsenic content in seafood in its *Health Risk Assessment of Chemical Contamination in Puget Sound Seafood* (PSEP 1988). Inorganic arsenic comprised 10 percent of the total arsenic concentrations in tissues collected from an estuarine system (ICF Kaiser 1996). Based on these data sources and at the request of EPA, estimated total arsenic concentrations were adjusted by a factor of 0.1 to account for the amount present in toxic inorganic forms.

4.4.2 Application of Measured and Estimated Values

Comparisons of measured tissue concentrations for PCDDs/Fs and total mercury (Section 2.2.3 and Appendix D) in Ward Cove with predicted values for those substances indicate that measured concentrations were consistently lower than estimated concentrations. The highest measured concentration for TECs in mussel tissue (0.78 ng/kg wet weight, whole body) (Table D1-5 in Appendix D) is approximately 2 percent of the highest estimated concentration of 39 ng/kg in fish whole body. The one exception is a measurement of 10 ng/kg TEC in crab hepatopancreas (Table D1-2 in Appendix D1). However, because the hepatopancreas represents a small proportion of the whole body weight, these values are still lower than estimated values on a whole body basis. Measured total mercury concentrations (Table D1-5 in Appendix D) are consistently lower than estimated concentrations (Table 4-5) once wet weight conversions are made.

As described in Section 2.2.3.2, similarly low concentrations of PCDDs/Fs were reported in mussels and rockfish collected in the APC investigation from areas with similar PCDD/F and TOC concentrations in sediments. The highest measured tissue concentration of 4.3 ng/kg wet weight in mussels is approximately 9 times less than the highest estimated tissue concentration of 39 ng/kg wet weight described above. Thus, the PCDD/F tissue data from the APC investigation provide further evidence of limited bioaccumulation of PCDDs/Fs from sediments.

Application of measured and estimated tissue concentrations in selection of CoPCs is discussed in Sections 6 and 7. In all cases, screening to identify CoPCs included comparison with estimated tissue concentrations. Maximum measured tissue concentrations of PCDD/F and mercury were also used in screening. Because estimated tissue concentrations were consistently higher than measured concentrations, the use of estimated concentrations appears to provide a conservative means to determine whether chemicals in sediments should be considered CoPCs where tissue data are unavailable.

4.5 SURFACE WATER

Prior to 1997 when the pulp mill was active, an important water quality concern in Ward Cove was potential oxygen depletion associated with the discharge of oxygen-demanding substances in the pulp effluent. This concern was largely addressed through effluent handling and treatment modification and monitored as part of the NPDES program. Effluent handling and treatment programs were successful, and oxygen depletion and water column characterization issues were not a focus of the technical studies work plan for Ward Cove sediments (PTI 1996). However, concerns regarding potential oxygen depletion in bottom water and its possible relationship to oxygen depletion in sediment were expressed in agency comments on the draft DTSR. This discussion addresses those concerns.

A brief summary of the key process and regional conditions affecting oxygen concentrations in seawater in the vicinity of Ward Cove is provided here as background for the discussion of water quality in Ward Cove. This summary is followed by an overview of NPDES monitoring requirements and associated water quality measurements in Ward Cove over the last several years.

4.5.1 Key Processes and Regional Conditions Affecting the Oxygen Content of Seawater in Ward Cove

The oxygen content of seawater is affected by a variety of processes, including gas exchange with the atmosphere (to produce oxygen saturation at the sea surface), oxygen production by photosynthetic organisms (which adds oxygen to seawater in the zone where light penetration is sufficient), and oxygen depletion during organic matter decomposition (which removes oxygen in the deeper waters as dead organisms fall through the water and are decomposed). The mixing of surface water (where oxygen is usually abundant) and deeper water (where oxygen is often depleted) is influenced by seasonal changes in the density structure of water. For example, warming of surface water during the summer creates a density gradient between shallow and deep water, which restricts mixing and limits oxygen exchange. Near rivers and streams, discharge of fresh water, which is much less dense than seawater, can also create a density gradient due to both salinity and temperature changes with depth. Water temperature also affects oxygen content; the saturated concentration of oxygen in seawater falls from about 10 mg/L at 2°C to about 6 mg/L at 25°C. The saturated concentration of oxygen is somewhat higher in fresh water than in seawater. Supersaturation may occur in the spring as a result of photosynthesis in the water column. Oxygen content can also reflect the past history of the water body. For example, deep ocean water can lose oxygen over time because it is isolated from the source of oxygen (the atmosphere and photosynthetic activity in surface water) and subject to a continual rain of decomposing organic mater, which consumes oxygen.

The "structure" of the water column in the vicinity of Ward Cove is characterized by the following features (Martinson and Kuklok 1977):

- **Surface Water Layer**—Where water temperature responds to seasonal changes, salinity is diluted by freshwater runoff, and oxygen content is highest because of gas exchange with the atmosphere and seasonal changes in photosynthesis
- **Pycnocline**—Where temperature decreases with depth and/or salinity increases with depth, creating a density gradient that restricts vertical mixing

- **Oxycline**—Where dissolved oxygen content rapidly decreases with depth (typically coinciding with the pycnocline)
- **Deep Zone**—Where temperature remains relatively constant, salinity increases only slightly, and oxygen content remains relatively constant.

Oxygen concentrations is seawater in southeast Alaska are strongly influenced by regional conditions. The oxygen content at 300-m depth on the open ocean off southeastern Alaska is quite low, approximately 1 mL/L (or 1.4 mg/L), which reflects an oxygen saturation of only 20 percent (Favorite et al. 1977). These low oxygen concentrations in deeper water are clearly present in the inlets in the Ketchikan area and are most pronounced in inlets with direct access to northeast Pacific Ocean water (Pickard 1967; Martinson and Kuklok 1977). In the Clarence Strait, oxygen concentrations decline from 6–7 mL/L (8.6–10 mg/L) in surface water to 2 mL/L (2.8 mg/L) at 100-m depth and remain low through the deep water layer.

4.5.2 Oxygen Content in Ward Cove Surface Water

Oxygen content, salinity, and temperature are monitored in Ward Cove as part of NPDES monitoring. Water column profiles of these variables are collected every 2 weeks at nine stations in Ward Cove and at four stations near the mouth of the Cove in Tongass Narrows (Figure 4-26). Temperature and dissolved oxygen measurements were collected at 5-m intervals as the sampling device was lowered and raised throughout the water column at each of the 13 stations shown in Figure 4-26. The deepest measurement was collected 1 m above the sediment surface (KPC 1999). If elevated turbidity was observed (i.e., if turbidity measurements were greater as the sampling device was raised compared to when it was lowered, then the sampling device may have hit the bottom and disturbed the sediment surface), a dissolved oxygen measurement was not collected at a given station until the turbidity readings were acceptable (KPC 1999).

Representative profiles of temperature, salinity, and oxygen for selected sampling events and stations (Figure 4-27) reflect the general trends described above (e.g., surface maximum in temperature and oxygen, usually followed by a decrease in concentration with depth) and also reflect seasonal trends in these variables. Pulp mill discharges to Ward Cove ceased in March 1997 when operations were terminated. Station 44 is located in the center of the Cove off of the KPC facility and Station TDP is located in Tongass Narrows. Representative profiles in temperature, salinity, and oxygen are shown for sampling events in January, April, August, and October of 1997.

The temperature profiles for Stations 44 and TDP indicate relatively constant temperatures in winter (represented by January) and spring (represented by April), followed by the gradual buildup of a thermocline in summer (August), which breaks down by October. Salinity profiles in Ward Cove have a surface minimum because of freshwater runoff, which is present throughout the year. The surface minimum in salinity is less pronounced at Station TDP. Oxygen concentrations generally decrease with depth. In

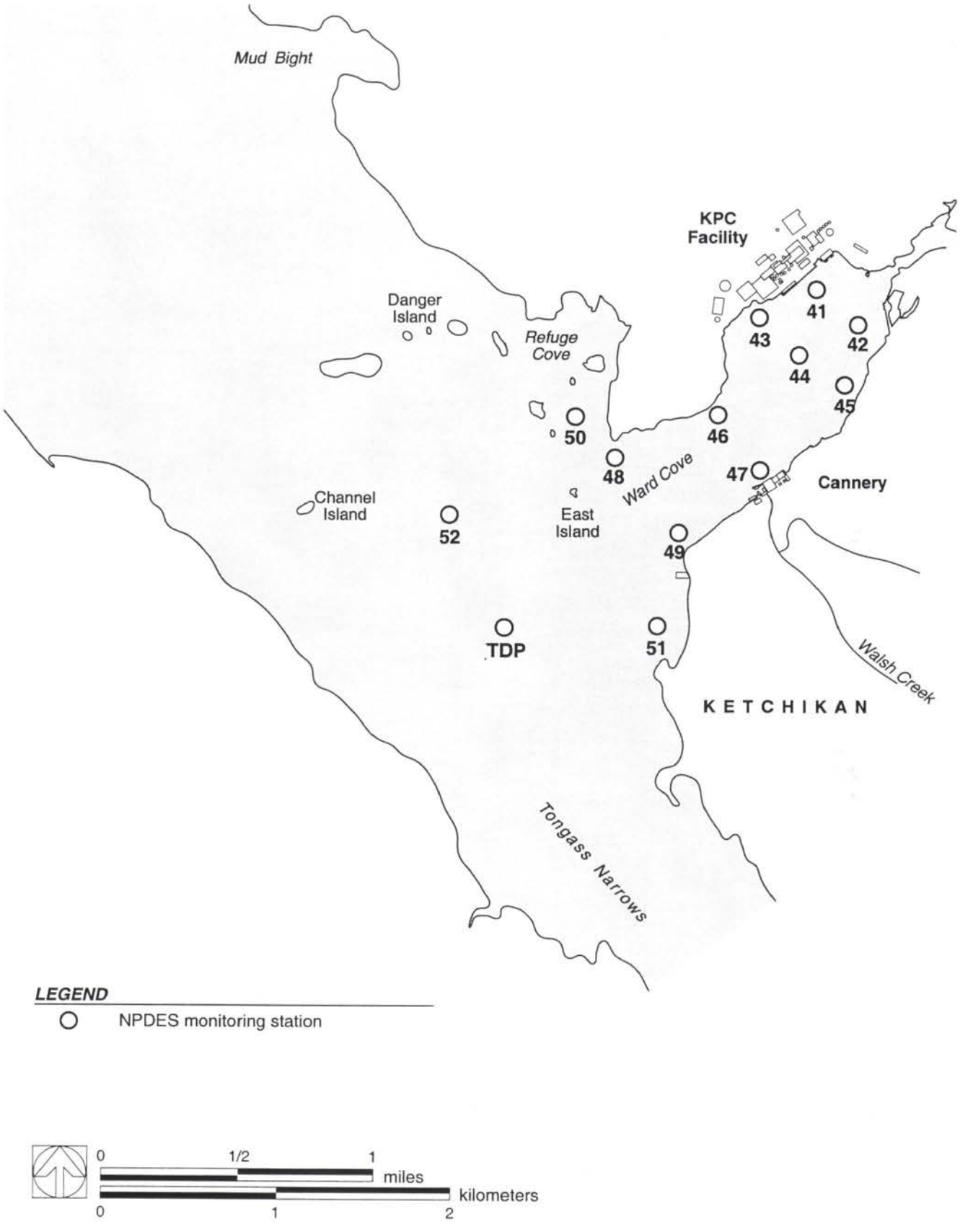


Figure 4-26. NPDES water column sampling locations.

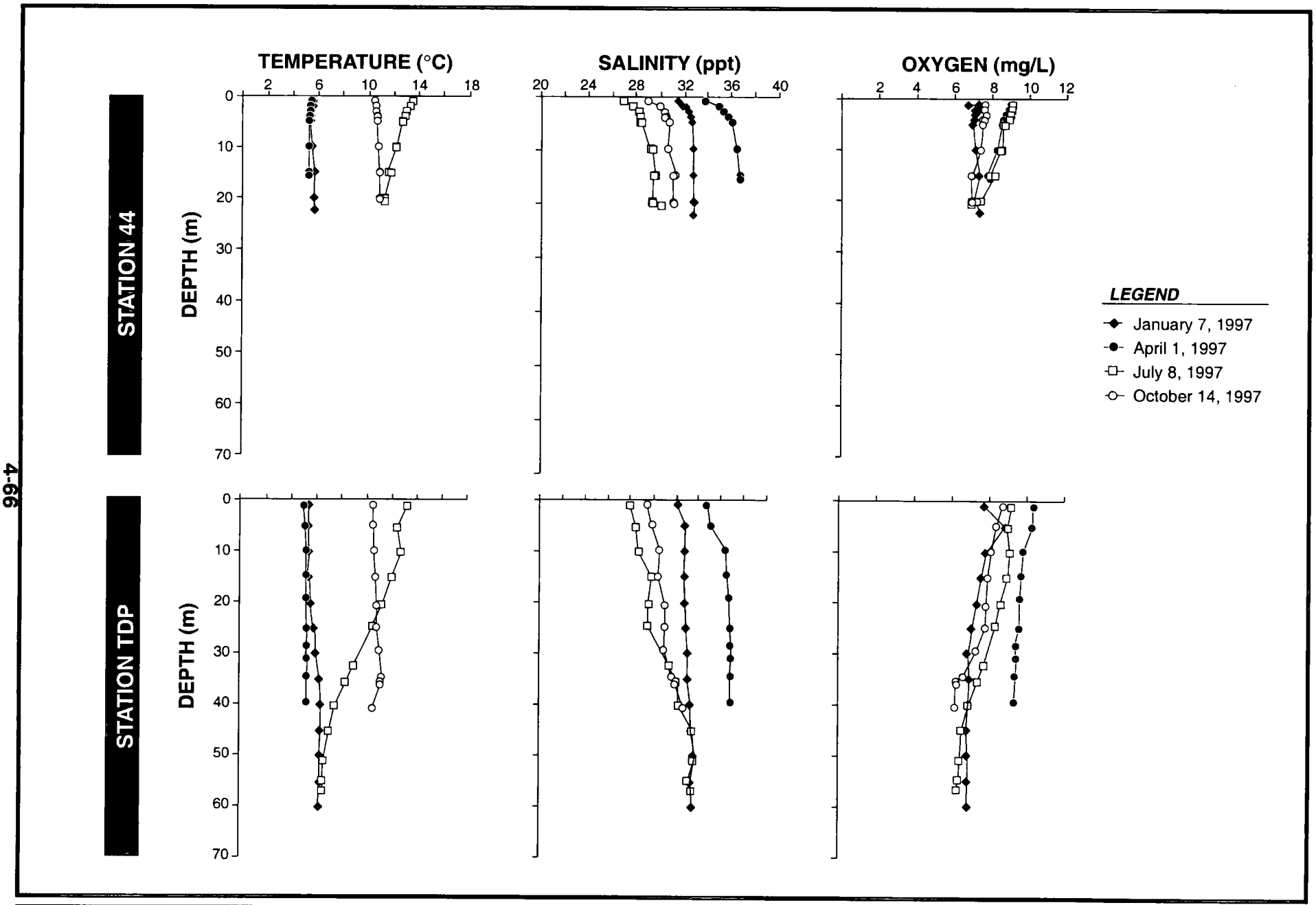


Figure 4-27. Temperature, salinity, and oxygen content in the water column of Ward Cove (Station 44) and Tongass Narrows (Station TDP).

Tongass Narrows, oxygen concentrations seldom dropped below 6 mg/L in 1997; however, in 1995 and 1996, concentrations were as low as 4 mg/L in the deeper waters. In Ward Cove, oxygen concentrations were typically above 6 mg/L at Station 44; however, concentrations dropped to 6 mg/L at the deepest sampling point during summer stratification (July 1997).

Seasonal patterns in dissolved oxygen can be evaluated in greater detail by looking at the last 3 years of NPDES monitoring data at Station 43, located adjacent to the KPC facility and near the major outfall (Figure 4-28), Station 44, located in the center of Ward Cove off the KPC facility (Figure 4-29), and Station 48, located at the mouth of Ward Cove along the north shoreline (Figure 4-30). In winter and spring, oxygen concentrations are relatively uniform with depth. Water column stratification during the summer months limits mixing, and oxygen profiles reflect production in surface water due to primary productivity and depletion in deeper waters as organic particles fall through the water column and are degraded, a process that consumes oxygen. Oxygen depletion in the water column is more likely attributable to the seasonal cycles of water column stratification and productivity supplemented by an ongoing discharge of oxygen depleting substances (i.e., organic matter) than to the presence of organic-rich sediments. In the latter case, reduction of oxygen in bottom water would be limited by the rate at which oxygen-consuming substances can diffuse out of the sediment and react with oxygen in the water column, a very slow process. Jones & Stokes and Kinnetic (1989) documented an increase in the incidence of low oxygen concentrations in bottom water in the late 1980s. In recent years, oxygen concentrations in Ward Cove typically have been above 8 mg/L.

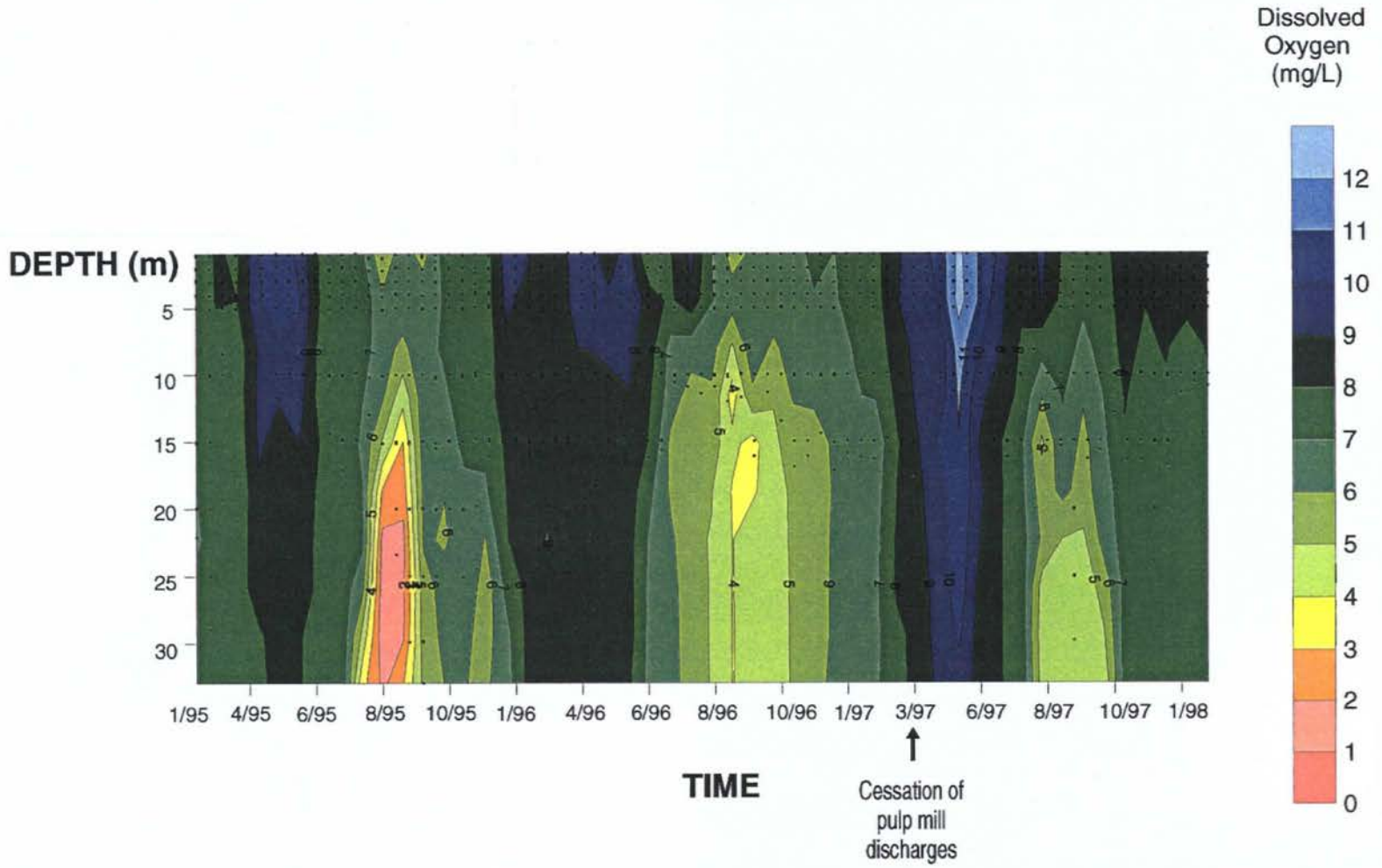


Figure 4-28. Dissolved oxygen concentration profile from 1/95 to 1/98 (Station 43).

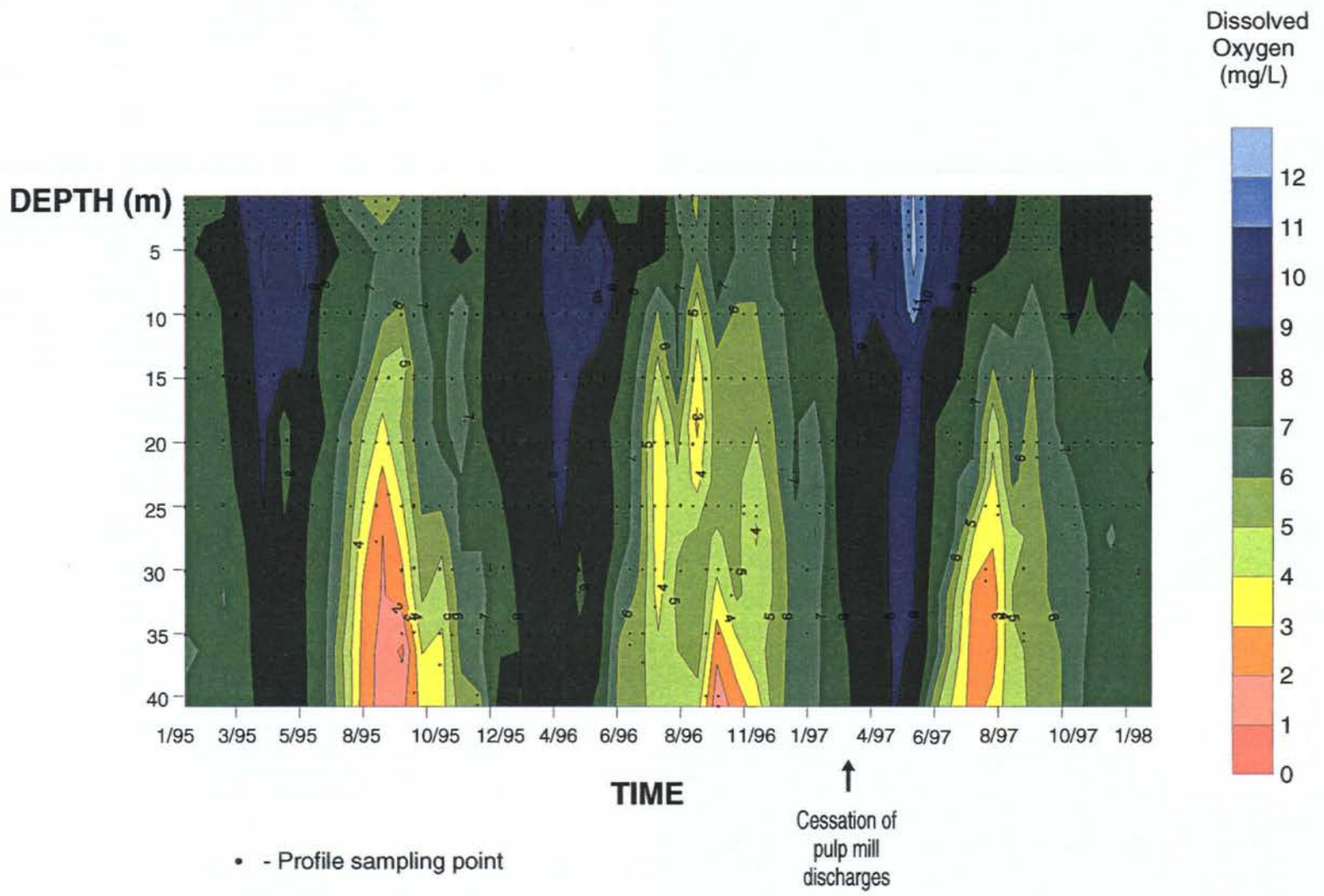


Figure 4-29. Dissolved oxygen concentration profile from 1/95 to 1/98 (Station 44).

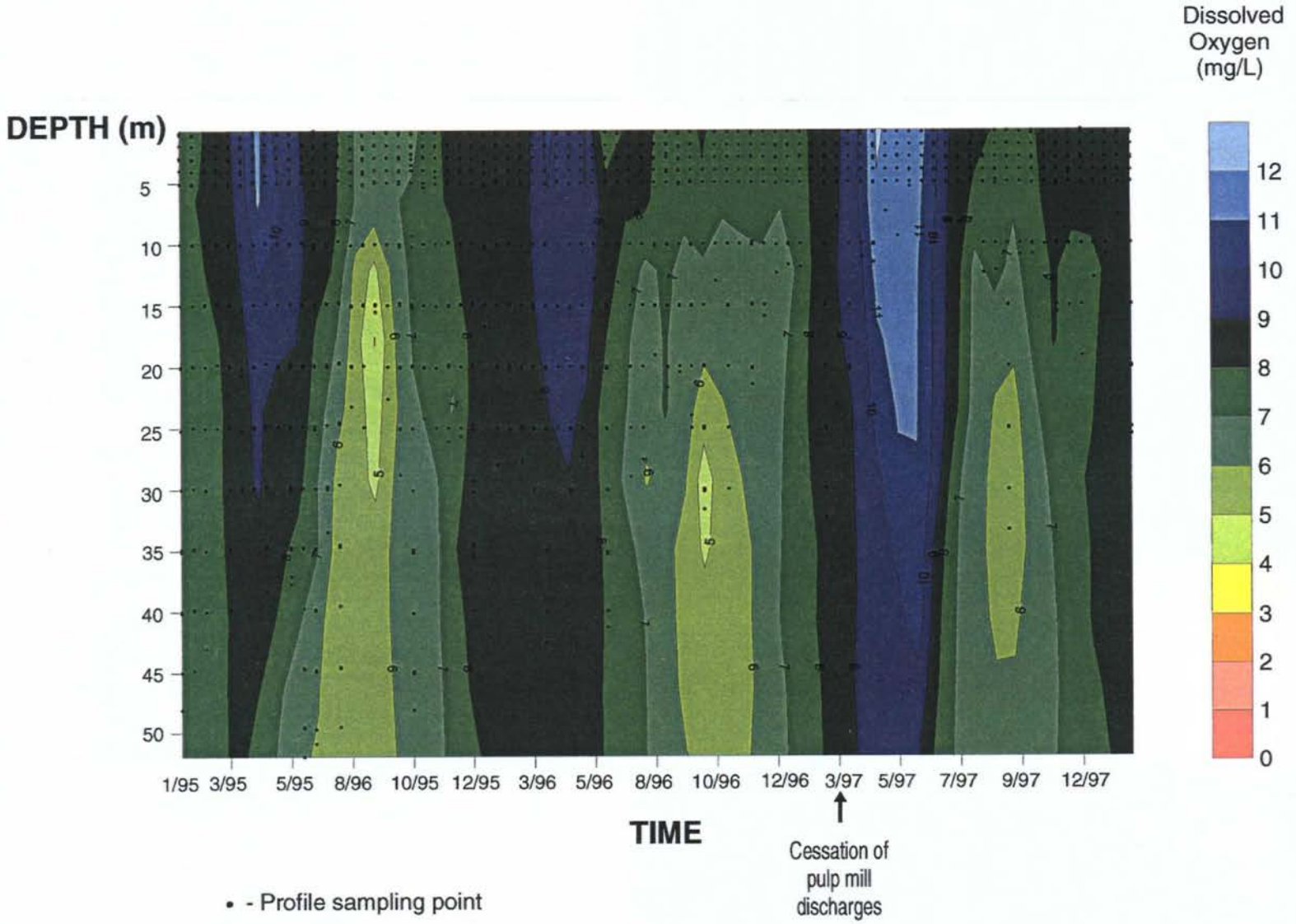
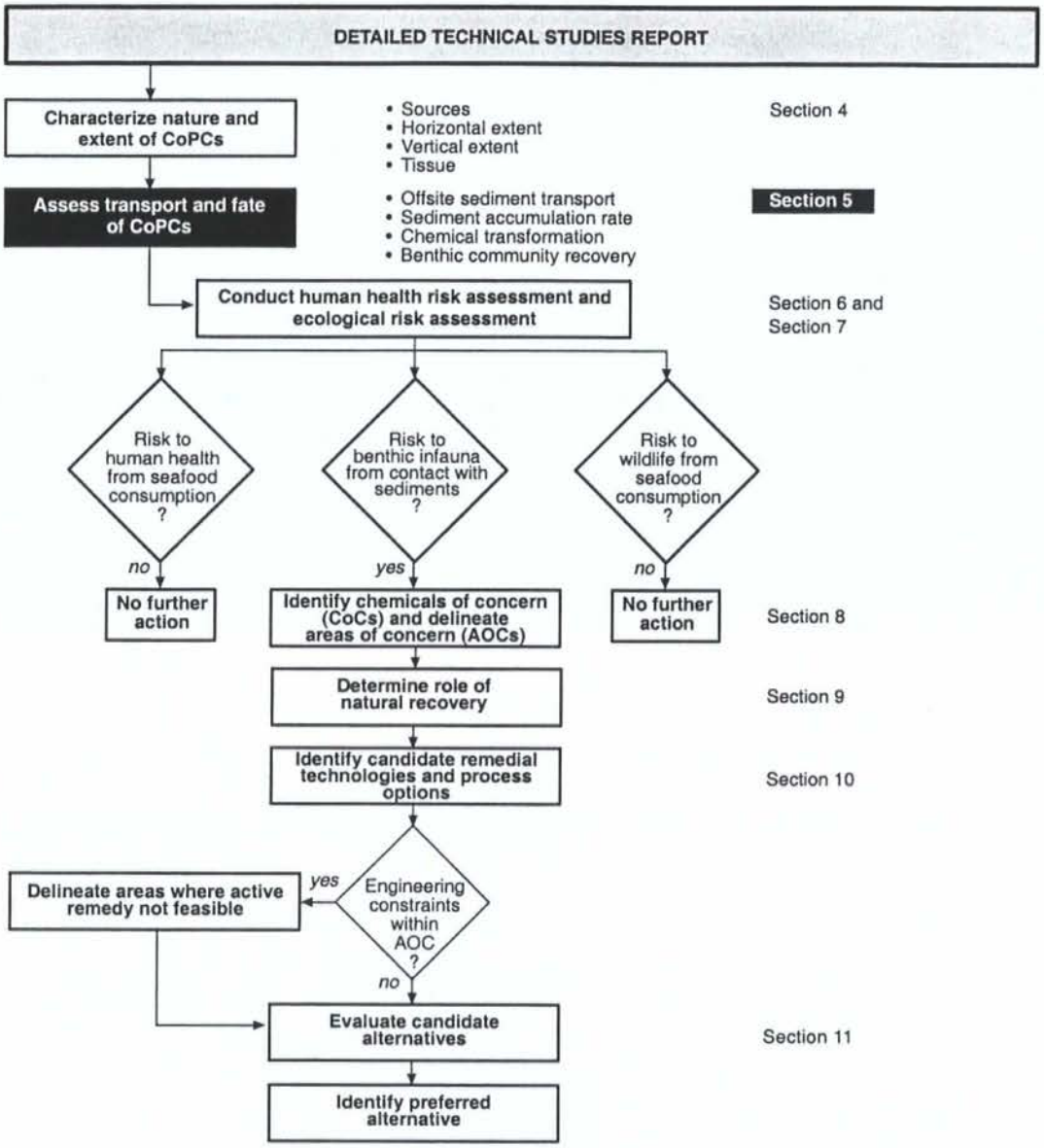


Figure 4-30. Dissolved oxygen concentration profile from 1/95 to 12/97 (Station 48).

5. CHEMICAL TRANSPORT AND FATE



The recent shutdown of the KPC mill has led to changes in conditions affecting chemical dynamics in Ward Cove. The abrupt reduction in organic matter loading is expected to lead to future changes in sediment characteristics. The expected rate and magnitude of those changes affects the selection of appropriate remedial alternatives. This section addresses several topics that bear on the prediction of future conditions and the appropriate present-day response. The transport and fate processes described in this section are incorporated into an assessment of natural recovery and its role in remediation in Section 9.

One of the concerns regarding current conditions in Ward Cove is the potential for transport of sediment-associated contaminants out of the Cove to nearby embayments. Sediment resuspension and offsite transport is addressed in Section 5.1, using data collected in 1997.

Sediment accumulation is an important component of recovery from a contaminated condition, because the buildup of clean sediment acts to dilute and isolate problem sediment. The sediment accumulation rate in Ward Cove has been calculated using radioisotope data collected in 1997, and the results are presented in Section 5.2.

Production and degradation of chemicals in sediment are also important components of sediment recovery, acting to retard or accelerate the process. Chemical transformations of organic matter are particularly important in Ward Cove, because organic material constituted both the raw material and the product of the KPC mill. Section 5.3 describes the transformation processes that are expected to affect future concentrations of organic compounds in Ward Cove sediment.

5.1 POTENTIAL FOR SEDIMENT RESUSPENSION AND OFFSITE TRANSPORT

The potential for remobilization and transport of sediment out of Ward Cove can be assessed by evaluating current velocities and sediment grain size data. The current meter at Station C (see Figure 2-5) is located nearest the mill in the area of affected sediments, and near the northern shore of the Cove, where the strongest outflow takes place. The current meter was in place from July 22 to August 23, 1997, a period that included the highest spring tide of the summer. The bottom current meter at this location was located 12 ft (3.7 m) off the bottom (Orders Associates 1997). The closest sediment station is Station 42. During the period of deployment, bottom current speeds at Station C averaged 1.2 cm/s; the average upper quartile speed was 2.6 cm/s, and the maximum speed (observed on one occasion) was 8.0 cm/s. The distributions of current speed and direction at this location are shown in Figures 5-1 and 5-2. Current speeds at the sediment surface will be lower than these speeds, in accordance with a logarithmic increase in current speed with distance from the bottom.

The current speed necessary to remobilize sediment (the critical shear velocity) can be estimated using some reasonable, but conservative, assumptions. Sediment cohesiveness has a strong influence on the potential for resuspension, and calculation of the critical shear velocity is more straightforward for noncohesive sediments than for cohesive sediments. Sediment cohesiveness is influenced by both particle grain size and organic material content. Particles of silt size or larger are generally noncohesive, whereas clay particles are cohesive. Clay has a controlling effect on cohesion when it is present at concentrations greater than approximately 10 percent (Raudkivi 1995). Organic material can increase or decrease cohesiveness, depending on the type, size, and amount of organic material and the type and size of mineral material. Because there are no standard techniques for estimating its effect on cohesiveness, the effect of organic material on cohesiveness is not considered here. Because of the potential that large woody debris

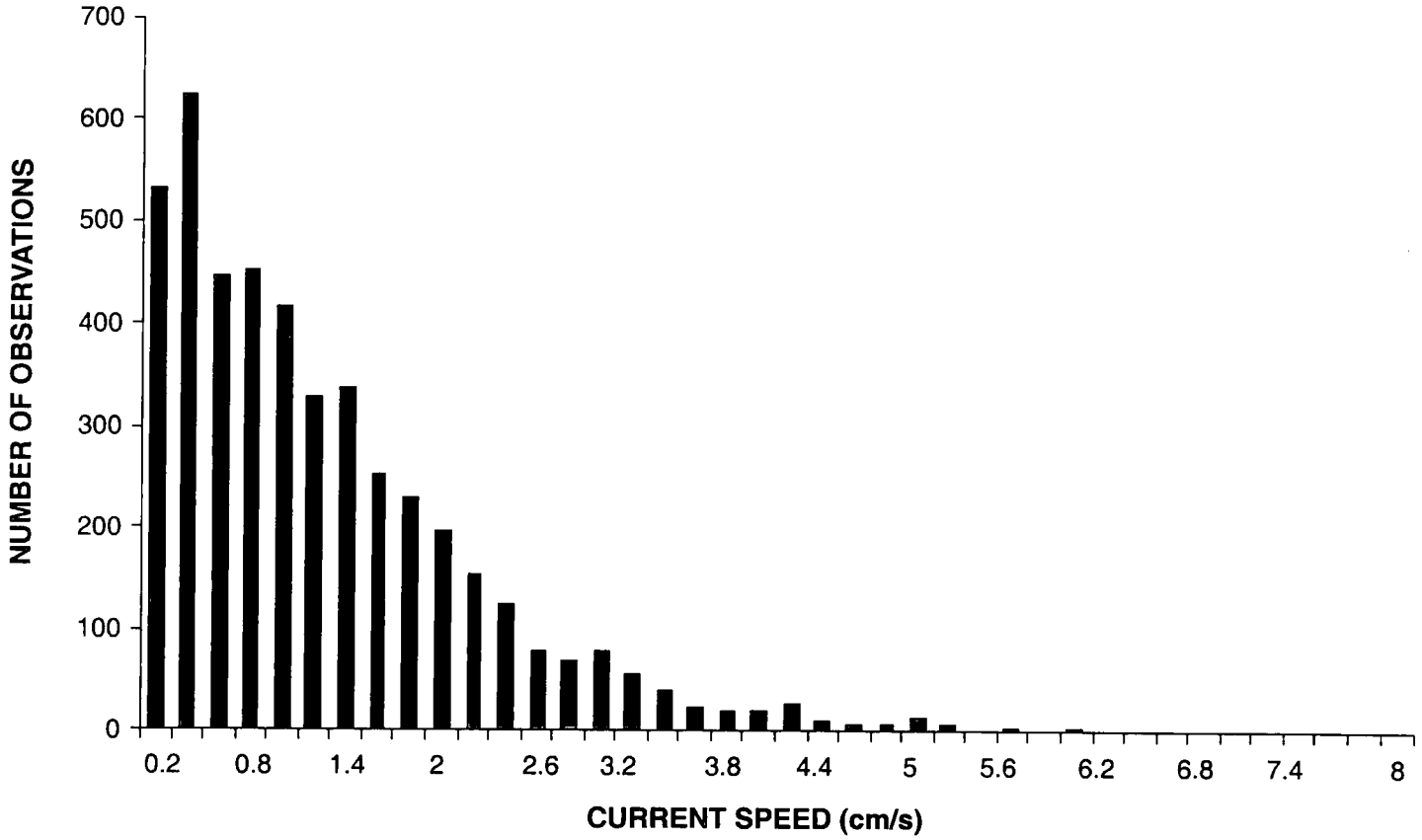


Figure 5-1. Current speed spectrum 3.7 m from the bottom at current meter station C.

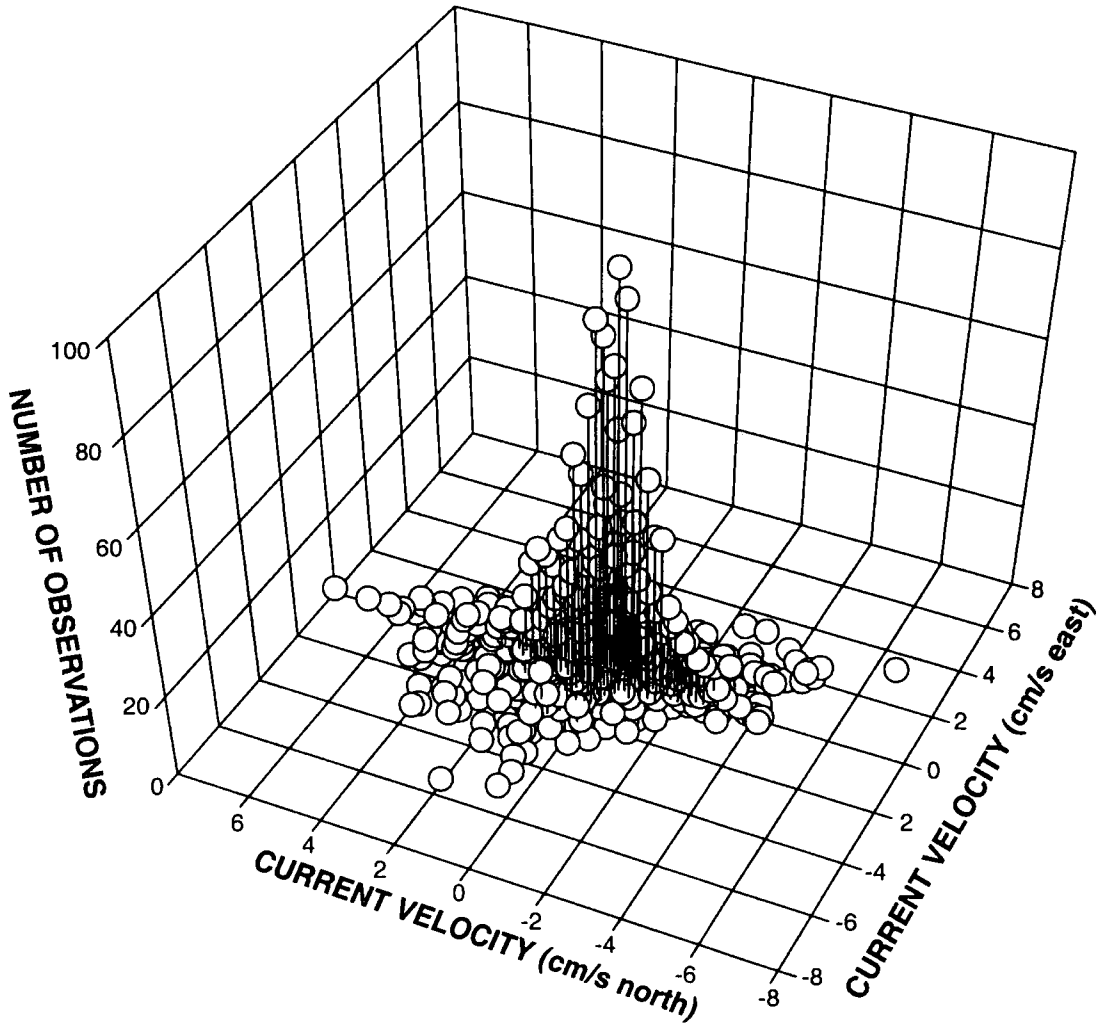


Figure 5-2. Current velocity 3.7 m from the bottom at current meter station C.

may decrease sediment cohesiveness, and because the TOC content of different sediment size fractions was not measured, a conservative approach has been taken to estimate the erodability of Ward Cove sediment.

Sediment at Station 42 (closest to Current Meter C) is 19 percent (by weight) clay-sized particles and 46 percent silt-sized particles. The remaining 35 percent of the sediment is sand (and possibly coarser material). Sediment with a clay content as high as this will be fairly cohesive. However, for the purpose of calculating a conservative estimate of resuspension potential, the sediment is assumed to consist entirely of the finest silt fraction—that is, the most easily resuspended noncohesive material—with a diameter of 2 μm.

With reasonable assumptions for particle and water density, the critical shear velocity at the sediment surface is calculated to be 1.5 cm/s (Raudkivi 1995, Equation 3.1, reproduced here as Equation 1).

$$u_{*_c} = B\sqrt{\left[\frac{\rho_s - \rho}{\rho} gd\right]^{\frac{1}{2}}} \qquad \text{Equation 1}$$

where:

u_{*c} = critical shear velocity, m s^{-1}
B = static threshold value for water ≈ 0.2 $m^{½}$ $s^{-½}$ (Raudkivi 1995)
ρ_s = density of solids ≈ 2.5 g cm^{-3}
ρ = density of water ≈ 1.02 g cm^{-3}
g = acceleration of gravity ≈ 9.8 m s^{-2}
d = particle diameter $\approx 2 \times 10^{-6}$ m

The corresponding critical velocity at a distance of 3.7 m above the bottom is 54 cm/s (Raudkivi 1995, Equation 3.2, reproduced here as Equation 2).

$$u_c = 5.75u_{*_c} \log\frac{D}{d} \qquad \text{Equation 2}$$

where:

u_c = shear velocity at a distance D above the sediment, m s^{-1}
u_{*c} = critical shear velocity, m s^{-1}
D = distance of current meter above sediment, m
d = particle diameter $\approx 2 \times 10^{-6}$ m

Thus, current speeds of 54 cm/s at the position of the lower Current Meter C are necessary to resuspend sediment if the sediment were composed entirely of fine silt. This value is 20 times greater than the observed mean upper quartile speed and more than 6 times greater than the observed maximum speed. Observed current speeds are therefore far too low to resuspend silt. Given that the actual sediment will be fairly cohesive as a

result of its clay content, the actual potential for resuspension is likely to be even lower than is indicated by the mismatch of velocities. Observation of a flocculent layer at the sediment surface along the northwest shoreline of the Cove (ENSR 1995b) is consistent with current velocities that are too low to mobilize fine sediment.

Because of the bilayer flow in Ward Cove, the net movement of bottom water is into the Cove, so any sediment that is resuspended is likely to move toward the head of the Cove. Furthermore, chemical concentrations in sediment immediately outside Ward Cove are more like those at a local reference area than like those within the Cove (PTI 1997b). Therefore, three lines of evidence—current velocities, current direction, and sediment conditions outside Ward Cove—all indicate that sediment resuspension and offsite transport is negligible.

5.2 SEDIMENT ACCUMULATION RATE

The rate of net sediment accumulation in Ward Cove is one of the most important factors affecting the fate of chemicals in the Cove. During the 1997 sampling effort, sediment cores were collected and analyzed from two locations in Ward Cove to determine the rate of sediment accumulation. Two independent techniques were used to estimate sediment accumulation. The primary technique used is evaluation of the profile of lead-210 radioactivity in the sediment. The secondary technique, used to confirm the results of the lead-210 analysis, is an evaluation of the depth of maximum cesium-137 radioactivity.

Lead-210 is a naturally occurring radionuclide that is ultimately derived from long-lived uranium radioisotopes in rocks of the Earth's crust. However, one of the more immediate antecedents of lead-210 in the uranium decay series is gaseous radon-222. The atmosphere is therefore the proximate source of lead-210, which is deposited uniformly on soil and water with an effectively constant rate of supply. Lead-210 is relatively insoluble and immobile in sediment and has a half-life of 22.3 years. Thus, if lead-210 is deposited at a constant rate in a physically undisturbed environment, a vertical profile of the deposit will show a logarithmic decline in lead-210 radioactivity with depth; the vertical distance over which lead-210 radioactivity declines by half corresponds to 22.3 years of accumulation. Observation of a logarithmic decline in lead-210 radioactivity with depth can therefore be used to calculate the accumulation rate.

There are two factors that must be accounted for in practice when calculating accumulation rates from lead-210 data:

- Depositional environments, particularly shallow-water sediments, are rarely completely undisturbed. Surface sediment will be mixed to some depth by both physical and biological processes. Lead-210 radioactivity in this mixing zone will be relatively uniform; only below the mixing zone can the profile of radioactivity be used to calculate the accumulation rate.

- Crustal material containing uranium isotopes is usually present in natural sediment deposits. Some lead-210 is therefore produced within the sediment after it is deposited. This portion is referred to as "supported" lead-210, whereas the accumulation rate must be calculated using only "unsupported" lead-210. If the profile is deep enough, the amount of supported lead-210 can be identified because below some depth lead-210 radioactivity becomes constant.

Calculation of a sediment accumulation rate from lead-210 data therefore requires that data from depths within the mixing zone be excluded, the quantity of supported lead-210 be excluded, and that the remaining data show a logarithmic decline in radioactivity with depth.

Cesium-137, a radionuclide with a half-life of 30.2 years, is produced during the fission of uranium isotopes and was introduced to the atmosphere during aboveground nuclear testing. As with lead-210, because of its atmospheric source, cesium-137 is distributed ubiquitously. However, the rate of supply of cesium-137 to sediments is not constant: peak production and deposition of cesium-137 occurred in the period 1963 to 1965 (Eisenbud 1973). Because of the abrupt cessation of aboveground nuclear testing after this time, and the consequent cutoff of the supply to sediments, the period of 1963 to 1965 can still be associated with the peak of the cesium-137 profile in sediments. Although mixing processes spread out the peak somewhat, cesium-137 profiles are nevertheless an effective means of confirming sediment accumulation rates determined by analysis of lead-210 profiles.

Cores from Stations 40 and 49 in Ward Cove were analyzed for both lead-210 and cesium-137. The depth profiles of these constituents at both stations are shown in Figures 5-3 and 5-4. Lead-210 profiles at both stations have the expected form: a mixed layer at the surface and a relatively constant lead-210 activity at the greatest depths. Between these two features, however, lead-210 changes approximately linearly with depth at Station 40, whereas it decreases exponentially with depth at Station 49. Because of the relatively deep mixing zone at Station 40 and the deviation from an exponential decline with depth at this station, data from Station 40 were not used to calculate a sediment accumulation rate. The absence of an exponential decline in lead-210 with depth implies that sediment at Station 40 has been disturbed or that the sedimentation rate was not constant. Bioturbation, ship traffic, and impacts from sunken logs could be responsible for disturbance; alterations in discharges from the mill or Ward Creek could be responsible for variations in sedimentation rate. The inability to calculate a sedimentation rate at Station 40 does not imply either a high or low sedimentation rate at this location.

The core logs indicate that there is no fine structure in the Station 40 core. The absence of a fine structure can be interpreted as consistent with mixing, which would, as described above, also have prevented the observation of useful profiles of lead-210 and cesium-137. There is a 6-cm layer of organic silty material at the top of the core from Station 49, which corresponds roughly to the upper layer (0–4 cm) that was interpreted as

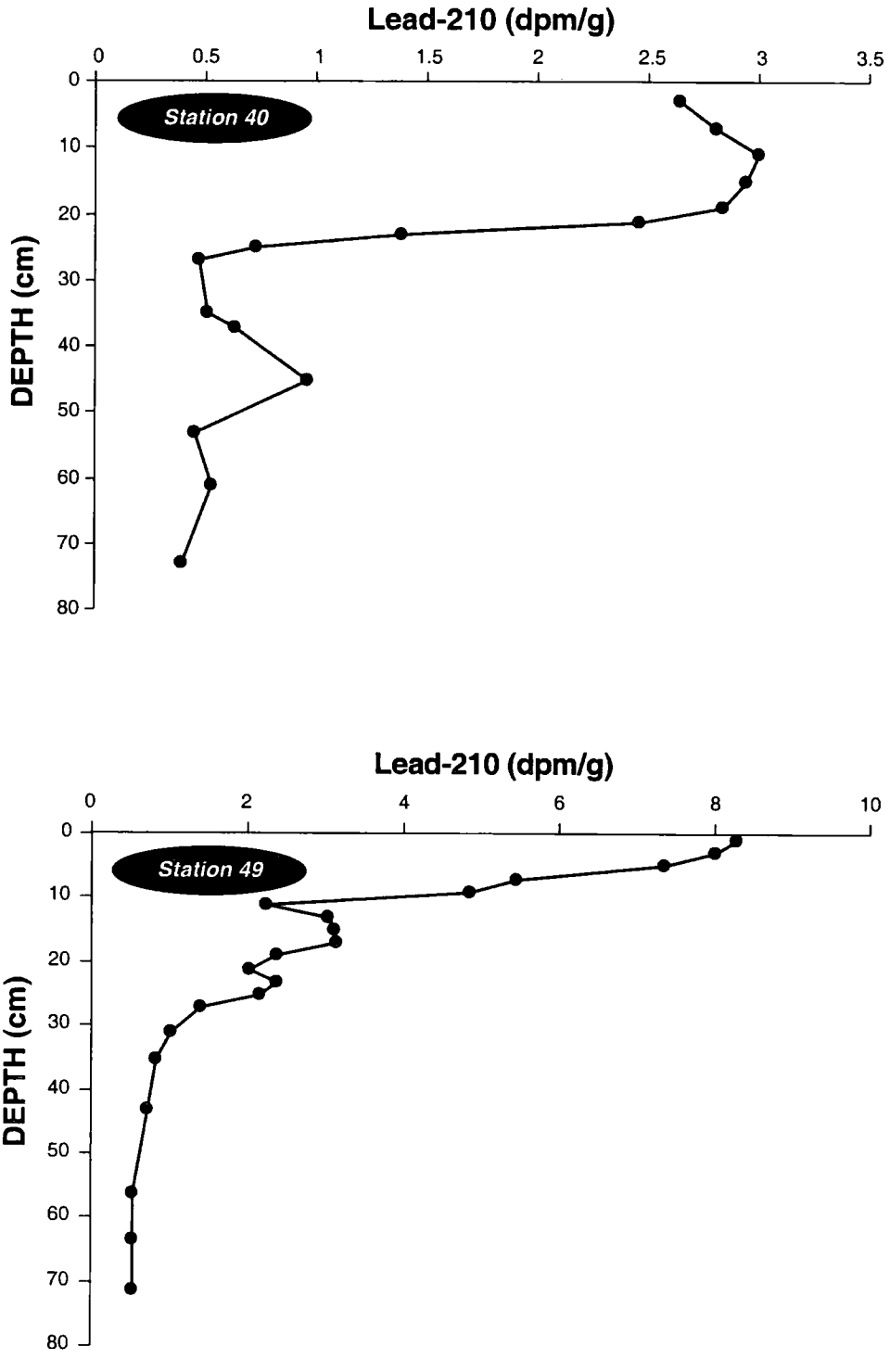


Figure 5-3. Sediment dating data for lead-210 for Stations 40 and 49.

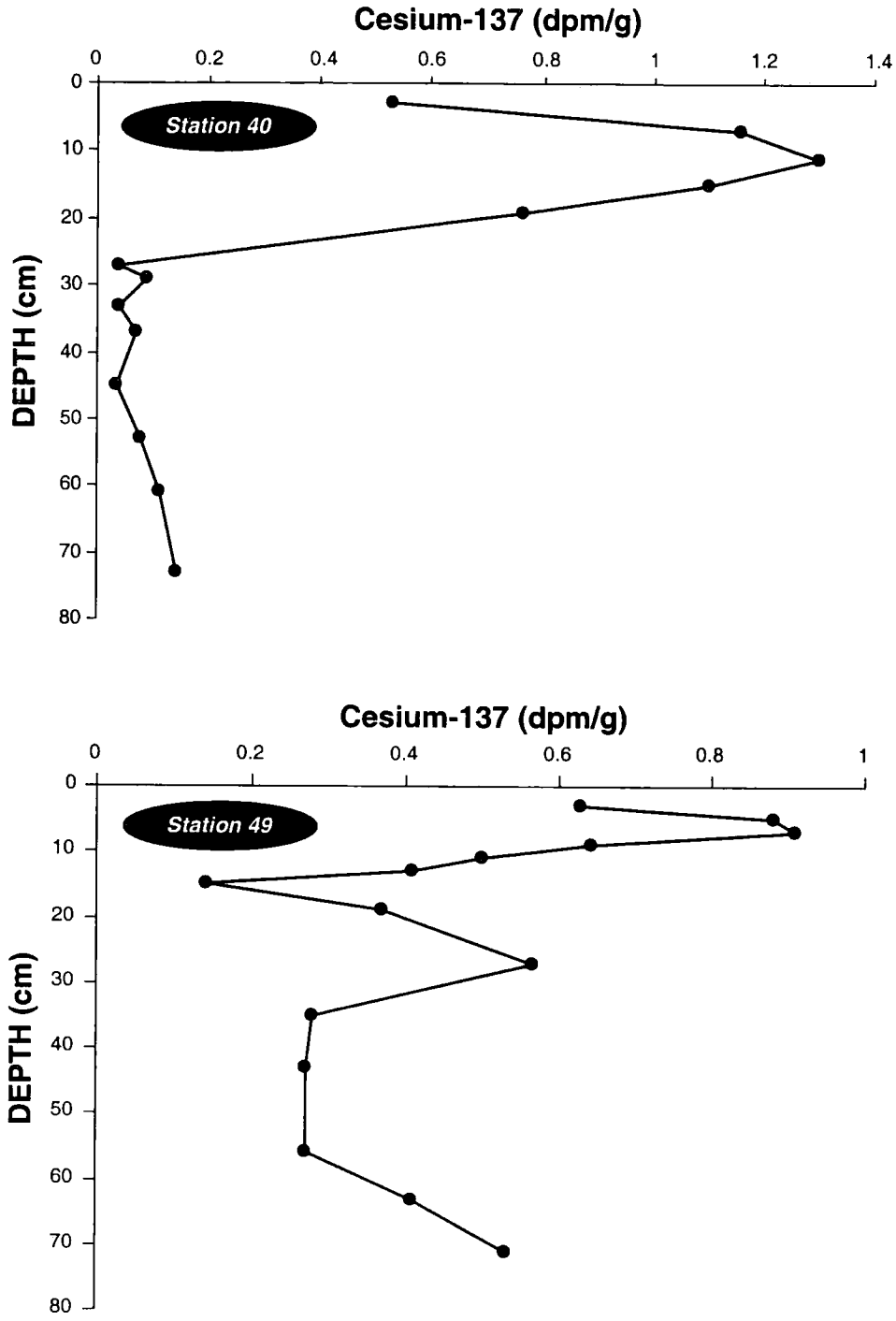


Figure 5-4. Sediment dating data for cesium-137 for Stations 40 and 49.

a mixed layer and excluded from the deposition rate calculation. Below the upper layer at Station 49 is silty clay that was described as "native" by the field crew. This observation is evidence that the deposition rate measured at Station 49 is representative of the deposition of native material, and thus of the rate that will occur after cessation of the KPC discharge.

Unsupported lead-210 data (dpm/g) from the exponentially declining region of the profile at Station 49 were log-transformed and regressed against sediment depth (cm). The regression was statistically significant ($P = 2.4 \times 10^{-8}$) and residuals were uniformly distributed. The slope of this regression is –0.0888 ln(dpm/g) cm^{-1}. This slope is equivalent to a half-depth (i.e., the depth over which lead-210 radioactivity declines by half) of –ln(0.5)/0.0888, or 7.8 cm. Thus, 7.8 cm of sediment accumulate over the 22.3-year half-life of lead-210, corresponding to a sediment accumulation rate of 0.35 cm/year.

The good fit of the logarithmic regression line to data down to a depth of more than 40 cm at Station 49 indicates that the sediment deposition rate has been essentially constant at this location for the last 100 years or more. The lead-210 profile therefore indicates that Station 49 is likely to have been minimally affected by the discharge of solids from the mill. As a result, the observed sediment accumulation rate at Station 49 is likely to be representative of sediment deposition at this and other locations in Ward Cove after cessation of the mill discharge.

The cesium-137 data from Station 49 peaks at 7 cm, indicating a sediment accumulation rate over the period 1963 to 1996 of 0.21 cm/year. The value derived from cesium-137 is close to that derived from lead-210 and confirms the general magnitude of the sediment accumulation rate. The cesium-137-derived sediment accumulation rate is considered to be less accurate than the lead-210-derived rate because the former is based on only a single data point whereas the latter is based on many data points. Furthermore, retention of cesium-137 in terrestrial watersheds and gradual washout can result in an upward shift of the cesium-137 peak in the sediment and a low bias to the sediment accumulation rate. Thus, the magnitude of the cesium-137-derived accumulation rate, and even the direction of its deviation from the lead-210-derived value, support the use of the lead-210-derived value.

Sediment accumulation throughout Ward Cove is assumed to take place at a rate similar to that measured at Station 49. Some differences are likely to exist throughout Ward Cove as a result of different rates of supply of settleable solids and variations in topography. The presence of substantial amounts of organic matter in the water column may also have affected the historical sedimentation rate, particularly in the area near the mill that currently has organic-enriched sediment (Figure 4-3). Different current speeds in shallow water and steeper slopes near the north shore of Ward Cove will also affect spatial variation in sediment deposition rate, but another important factor is distance from the mouth of Ward Creek, which is the principal post-closure source of settleable solids. The area of concern (AOC) is closer to the mouth of Ward Creek than is Station 49, and is therefore expected to experience a greater sedimentation rate. Because currents are tidally driven, and tidal water enters principally in deep water, current speeds may

actually be less in shallow water. However, wind stress also affects the movement of surface water to a limited depth. The exact balance between these opposing factors cannot be determined. However, within the bounds of this uncertainty, the sedimentation rate measured at Station 49 is expected to be representative of most of Ward Cove following shutdown of the KPC mill.

5.3 CHEMICAL TRANSFORMATIONS IN SEDIMENT

The CoPCs in Ward Cove are derived from the woody material that was both raw material and product of the KPC mill. Microbially mediated decomposition of the woody material leads to oxygen depletion and production of ammonia, sulfide, and 4-methylphenol in the sediment. The resulting conditions affect the sediment's suitability as habitat for other organisms. However, both abiotic processes (e.g., porewater diffusion) and biotic processes (e.g., sediment irrigation) can act to mitigate the harmful effect of these conditions. Exchanges of dissolved oxygen and other constituents between the sediment and the overlying water affect the rate of production, the vertical distribution, and the rate of loss of the CoPCs. Solubility of the chemicals and sorption characteristics of the sediment also affect chemical distributions. The sediment quality in Ward Cove is therefore established by the interplay of multiple process acting on several different sediment characteristics. At the base of all of these processes is the biodegradation of organic matter that gives rise to adverse sediment conditions.

5.3.1 Organic Matter Degradation

Sediment in parts of Ward Cove contain a large fraction of organic matter. TOC concentrations in the sediment range from 1.1 to 41 percent (dry weight), with a median concentration of 22 percent. The primary type of organic matter is wood and wood by-products, which consist of three main constituents: cellulose, hemicellulose, and lignin. Other types of organic matter (e.g., plankton and terrestrial debris) that are present in most nearshore sediment are also expected to be present. The cellulosic components make up 70 to 80 percent of wood by weight and are composed primarily of polymerized saccharoids, such as glucose. Cellulose is the salable product of wood pulping, and most of the cellulose is retained during the pulping process. Hence, the concentrations of cellulose in the Ward Cove sediments will be considerably less than their concentration in wood. Lignin makes up the remaining 20 to 30 percent of wood, and is composed of polymerized methoxy phenols (Hedges 1990).

Under the anaerobic conditions present in the Ward Cove sediments, only the cellulosic components of wood are expected to biodegrade (Hedges 1990). However, anaerobic degradation of wood is slow (Hedges 1990), and even the cellulosic components can persist for extended periods of time (Hatcher 1988). Degradation rates decrease exponentially with time as a result of increasing recalcitrance of the lignocellulose to microbial degradation (Benner et al. 1984), and wood in the interior of logs and timbers persists relatively unchanged for extended lengths of time when buried in anoxic sediments (Wilson et al. 1993).

Many types of bacteria consume organic matter in sediments as a food source. The reactions that they mediate depend upon the chemistry of the surrounding pore water (Berner 1980). Near the sediment–water interface, where oxygen is available, the organisms use oxygen to assist in the breakdown of organic compounds, generating carbon dioxide and water (oxygen is used as an electron acceptor). Because this process consumes oxygen, its rate is limited by the diffusion of oxygen into the sediments. As a result, oxygen concentrations in sediments usually drop off quickly below the sediment–water interface.

When oxygen concentrations drop below about 0.5 mg/L, microorganisms must find another source of oxygen to enable them to break down the organic matter. At this point, they begin using nitrate as an electron acceptor, which is subsequently reduced to molecular nitrogen (via denitrification). Nitrate reduction is enhanced in sediments containing higher concentrations of organic matter (Smith and DeLaune 1986). As with oxygen, the rate of this reaction is dependent upon the diffusion of nitrate into the sediments. However, the rate of ammonia production is not dependent on ammonia concentrations in the sediment pore waters (Callender and Hammond 1982).

In most sediments, manganese and iron reduction are important below the zone where nitrogen is consumed (Berner 1980). The sediments in Ward Cove consist primarily of wood by-products; however, iron and manganese oxyhydroxides are expected to be present as coating on the inorganic particles.

After all of the iron has been reduced, microorganisms begin using sulfate as an electron acceptor, generating hydrogen sulfide. Unlike ammonia, excessive concentrations of hydrogen sulfide are toxic to the organisms that generate it, so the reaction rate is dependent on diffusion of hydrogen sulfide out of the sediment.

At the depth where all of the sulfate is consumed, microorganisms begin using the oxygen in organic matter itself. This process, called fermentation, converts organic matter into carbon dioxide and methane. This reaction will not occur in the presence of dissolved oxygen, nitrate, or sulfate. However, it also does not depend on the diffusion of an electron acceptor into the sediment, so the extent of the reaction is controlled only by the amount of available organic matter. In the case of wood and wood by-products, only a fraction of the organic matter present is available to be used by microorganisms. Lignin is particularly recalcitrant and does not appear to be biodegraded to any appreciable extent under anaerobic conditions (Zeikus et al. 1982; Kirk and Farrell 1987). Thus, lignin can persist more or less indefinitely in anaerobic sediments and over geologic time will be converted to coal.

5.3.2 Ammonia Production and Loss

Ammonia is ubiquitous in surface waters and is an integral part of the nitrogen cycle (Frazier et al. 1996). Sources of ammonia to the environment include sewage, industrial and farm wastes, and fertilizers. Ammonia is also contributed to the environment by the anaerobic breakdown of nitrogen-bearing organic matter (ammonification), one of the

major processes in the nitrogen cycle. The most important mechanisms for removal of ammonia from the environment include assimilation into organisms as a nutrient and biological oxidation to nitrite and nitrate, a process known as nitrification (Caffrey 1995; Sarda and Burton 1995). Because of its ubiquitous distribution, and because it is toxic to aquatic organisms over a wide range of concentrations, ammonia is a common source of toxicity in sediments. Free or un-ionized ammonia (NH_3) is more toxic than the ammonium ion (NH_4^+) (Sarda and Burton 1995).

Ammonia is most likely contributed to Ward Cove sediments as a nutrient in the nitrogen cycle. The circulation patterns in estuaries and fjords create a trap in which nutrients tend to accumulate (Stumm and Morgan 1981). Ward Cove has this type of circulation, in which nutrient-rich bottom water is flowing into the Cove, while nutrient-poor surface waters are flowing out of the Cove, trapping nutrients in the Cove. The increased nutrient concentrations, in conjunction with the very high organic carbon content of the sediment, should result in high rates of ammonification. Ammonia concentrations in Ward Cove sediments (11–2,800 mg/kg dry weight) are similar to sediment concentrations in other estuaries. For example, total inorganic nitrogen concentrations in Potomac River estuary sediments ranged from 72 to 1,710 mg/kg (dry weight), with the highest concentrations found in the upper 10 cm of the sediment (Simon and Kennedy 1987). These sediments also contained high concentrations of dissolved iron, indicating that the sediments were anaerobic and that the nitrogen was present as ammonia.

The concentration of ammonia at the sediment–water interface will depend on the production rate of NH_4^+ in the sediment and the flux of NH_4^+ out of the sediment. Production rates of NH_4^+ in coastal sediments are variable and depend on factors such as temperature, sediment type, and the amount and type of nutrient input. For example, Sumi and Koike (1990, as cited by Caffrey 1995) measured NH_4^+ production rates from 2.1 to 63 mmol/m^2-day (37.8 to 1,130 mg/m^2-day) in Japanese coastal sediments, and Lomstein et al. (1989, as cited by Caffrey 1995) found 2.9 mmol/m^2-day (52 mg/m^2-day) in sediments on the Bering Shelf.

Ammonia flux out of sediment is also variable and depends upon the physical properties of the sediment. Total flux of NH_4^+ out of sediments in the Potomac River was as high as 26 mmol/m^2-day (468 mg/m^2-day), with average flux rates of 8 mmol/m^2-day (Callender and Hammond 1982). However, diffusive flux was only 0.7 to 4.5 mmol/m^2-day. The authors attribute the difference between total flux and diffusive flux to macrofaunal irrigation (i.e., pumping of water through the sediment by benthic macrofauna). Caffrey (1995) also found that macrofaunal irrigation increases the flux of NH_4^+ out of sediments. Ammonia is rapidly oxidized to nitrite and nitrate in the presence of oxygen, and the water column in Ward Cove is highly oxygenated. Therefore, ammonia will be rapidly degraded upon diffusion out of the sediments.

Concentrations of NH_4^+ in near-surface sediments and bottom waters vary seasonally in most systems, including estuaries (e.g., Nedwell et al. 1983; Caffrey 1995; Frazier et al. 1996). During the winter, when productivity is low and little organic detritus is reaching the sediments, the surface sediments can become oxygenated, and NH_4^+ diffusing out of

deeper sediments is converted to NO_3^- before reaching the water column. During the late summer, however, organic detritus blankets the bottom sediments. As discussed in Section 5.3.1, the organic matter consumes oxygen, and the surface sediments become anoxic, allowing NH_4^+ to diffuse out. Thus, the oxidation of NH_4^+ in the surface sediments, which prevents its diffusion into the overlying water column, is dependent upon the extent of the surface oxidized layer (Nedwell et al. 1983). In Ward Cove, the extremely high organic carbon content of the sediments will likely limit the extent of the oxidized layer. However, recolonization of the sediments by benthic macrofauna may enhance NH_4^+ oxidation, because bioturbation extends the depth of the aerobic oxidized layer.

5.3.3 Sulfide Production and Loss

Most marine sediments contain hydrogen sulfide, the production of which is a natural consequence of elevated organic material concentrations (e.g., >2 percent TOC; Thompson et al. 1991). Sulfide is generated by bacterial sulfate reduction, a common process of organic matter decomposition in continental margin sediments, both in and below the zone of bioturbation (Berner 1980). The process occurs only in the complete absence of oxygen (Section 5.3.1).

As sediments accumulate, seawater is trapped in the pores. Seawater has a sulfate concentration of 2,700 mg/kg (Stumm and Morgan 1981), and in the absence of oxygen, this sulfate is reduced to sulfide. If iron is present, the sulfide will react with iron to form iron sulfides (e.g., pyrite), thus removing the sulfide from the pore water (Berner 1980).

Hydrogen sulfide is rapidly oxidized to a variety of oxidized sulfur species (including sulfate) in the presence of oxygen (Stumm and Morgan 1981). Hence, any sulfide diffusing out of the sediments will not persist when it reaches the oxygenated water column. As with NH_4^+, concentrations of sulfide vary seasonally in near-surface sediments and bottom waters (Section 5.3.2). The sulfide concentration at the sediment–water interface will depend on the thickness of the oxidized layer, which is likely to be extremely limited in Ward Cove because of the extremely high organic carbon content of the sediments.

5.3.4 4-Methylphenol Production and Loss

4-Methylphenol, a natural product widely used by industry, is produced either by recovery from petroleum or coal tar or by specialty processes that produce specific isomers. 4-Methylphenol is used in the formulation of antioxidants and in the fragrance and dye industries. Synthetic food flavors also contain 4-methylphenol. Methylphenols also occur naturally as metabolites of microbial activity, in various plant lipids, and in the urine of mammals. Methylphenols have been detected in foods and beverages such as tomatoes, cooked asparagus, coffee, black tea, and smoked food.

4-Methylphenol in the Ward Cove sediments is probably a by-product of lignin degradation (Sjöström 1981; Hatcher et al. 1988). Although unaltered lignin does not readily

degrade under anaerobic conditions (Zeikus et al. 1982; Kirk and Farrell 1987), pulping breaks down lignin molecules and makes them more readily degraded (Crawford et al. 1977). Sulfite pulping, the type of pulping used at the KPC facility, breaks down the polymeric structure of the lignin into individual aromatic sulfonic acids (Gellerstedt 1976). These acids are then available to microorganisms in the sediment, which probably use the sulfonate group as an electron acceptor. The methoxy group is then cleaved and the molecule is converted to a catechol (Hatcher 1988; Stout et al. 1988). Finally, the second hydroxyl group is lost, leaving 4-methylphenol (Hatcher et al. 1988).

Even in the absence of sulfite pulping, wood generates 4-methylphenol as it degrades. Hatcher et al. (1988) analyzed progressively degraded wood from recently waterlogged samples to peat to lignite to coal and demonstrated that the concentration of 4-methylphenol in the samples increased with increasing age. These results indicate that the wood and wood by-products in Ward Cove will continue to generate 4-methylphenol for some time to come. However, the rate of biodegradation of wood, and thus the rate of 4-methylphenol production, will decrease over time (Hodson et al. 1983).

Although it is likely to continue to be generated in the sediments in Ward Cove, 4-methylphenol is also very easily degraded. Degradation tests in oxic saltwater from three sites in Pensacola Bay, Florida, resulted in half-lives ranging from 9 to 43 hours. In marine water and anoxic sediment cores from three Pensacola Bay sites, biodegradation half-lives ranged from 3 to 16 hours (Howard 1989). Other studies have demonstrated half-lives ranging from 1 to 16 hours in soil, 1 to 16 hours in surface water, and 2 to 672 hours in groundwater (Howard et al. 1991). Under aerobic conditions, the aqueous biodegradation half-life ranges from 1 to 16 hours, while under anaerobic conditions, the aqueous biodegradation half-life ranges from 240 to 672 hours (Howard et al. 1991).

4-Methylphenol degradation rates will be slower in the colder waters of Ketchikan than in the Pensacola Bay sites. A decrease in temperature of 10°C results in a 2 to 3 times decrease in reaction rates (Brady and Holum 1981, p. 491). The average temperature of ocean water in Ketchikan is approximately 10°C, as compared to an average ocean-water temperature in Pensacola of approximately 20°C (NOAA 1998). Therefore, the half-life of 4-methylphenol in Ward Cove will be 2 to 3 times longer than the half-life in Pensacola Bay.

Extrapolation of the literature half-lives to the temperatures in Ketchikan results in aerobic half-lives on the order of 2 to 129 hours. This range in half-lives indicates that 4-methylphenol will still degrade rapidly (half-life of less than 5.5 days). Anaerobic half-lives should be on the order of 480 to 2,016 hours (20 to 84 days, or less than 3 months).

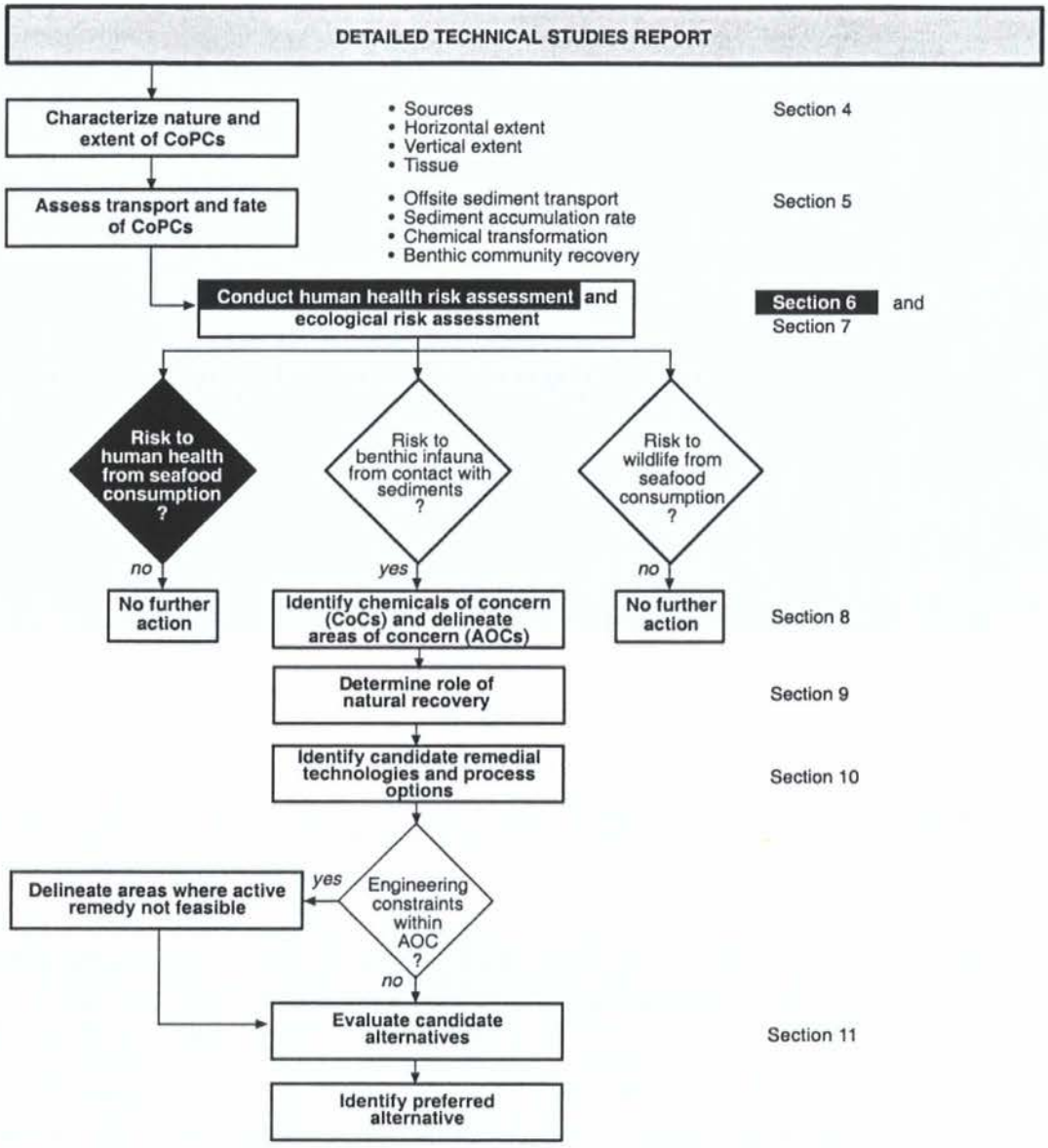
Based on the above information, the following conclusions can be made about 4-methylphenol biodegradation:

- 4-Methylphenol is readily biodegraded under aerobic and anaerobic conditions and in seawater.
- Aerobic aqueous biodegradation is considerably faster than anaerobic aqueous biodegradation. Anaerobic biodegradation nevertheless has a half-life of less than 3 months.
- The half-lives cited for the Pensacola Bay experiments suggest that biodegradation in sediments will be somewhat faster than in the water column. However, the oxygen conditions are not cited, and the experiments were probably aerobic. Thus, the addition of anaerobic sediments probably caused an increase in degradation rates as a result of increases in microbial populations, substrate, or nutrients. However, it does not imply that degradation rates in *anaerobic* sediments will exceed those in the *aerobic* water column.

In addition to being easily biodegraded, 4-methylphenol also readily sorbs to organic matter in the sediments. Partition coefficients to organic carbon (K_{oc}s) reported in the literature range from 17 mL/g (Fetter 1994) to 650 mL/g (Howard 1989). Sorption to sediment will retard 4-methylphenol transport out of the sediment by diffusion.

4-Methylphenol is likely to be generated in the sediments for an indefinite future period. It will slowly diffuse out of the sediments, but its diffusion will be retarded by sorption to the sediment organic matter. Oxygenation of sediments by benthic organisms as recolonization proceeds would enhance the degradation of 4-methylphenol. As it diffuses out of the sediment, its concentration will be reduced by biodegradation. Upon reaching the oxygenated bottom waters, it will be rapidly biodegraded.

6. BASELINE HUMAN HEALTH RISK ASSESSMENT



A baseline human health risk assessment was conducted to identify potential risks related to chemicals detected in sediments, fish, or shellfish collected near the site in Ward Cove. Risks associated with exposures in upland areas were evaluated in a separate remedial investigation. Cumulative risk estimates for individuals who might be exposed to chemicals in both upland media and Ward Cove media will be derived during the process of selecting remedial actions for upland and Ward Cove operable units as part of evaluating residual risks (Yost 1998, pers. comm.).

Risk analyses were consistent with guidance provided by EPA (U.S. EPA 1989e, 1991a,b, 1996a) and incorporated fish and shellfish consumption rates that are representative of average consumption in a local subsistence fishing community (Wolfe 1995, pers. comm.; Freeman 1995, pers. comm.). The following sections discuss the potential for people to be exposed to chemicals detected in sediments. Potential human receptors and exposure pathways are reviewed, and seafood consumption is identified as the only complete exposure pathway. Subsequent sections describe screening of site data to determine whether any chemicals pose potential risks to human health. Maximum estimated seafood concentrations for all chemicals and measured concentrations for PCDDs/Fs and mercury were compared with available background concentrations and with risk-based concentrations for seafood derived using site-specific seafood consumption rates. In general, chemicals were to be considered chemicals of concern (CoCs) if both background and risk-based concentrations were exceeded. Despite the use of conservative screening methods, no CoCs were identified for human health, and thus no further risk analyses were conducted. Uncertainties associated with risk estimates are summarized in Section 6.3 and discussed in more detail in Appendices G and H.

6.1 HUMAN EXPOSURE POTENTIAL

This section summarizes human populations that might be exposed to chemicals in sediments or in tissues (i.e., receptor populations) and pathways that could lead to human exposure (i.e., exposure pathways). Conservative, site-specific seafood consumption rates are then identified for the Ward Cove area.

6.1.1 Human Receptors and Pathways

Exposures are expected only where an exposure pathway is complete. Exposure pathways are considered complete when they have each of the following characteristics: CoCs identified in an exposure medium (e.g., CoCs in tissues at concentrations exceeding background); an actual or hypothetical means that a receptor may come in contact with that medium (e.g., anglers who fish in affected areas within Ward Cove); and a route of exposure (e.g., consumption of seafood containing CoCs). Where one of these elements is absent, the exposure pathway is considered not to be complete and no hazards are expected.

Human receptors may contact chemicals in Ward Cove sediments or seafood through the following hypothetical exposure pathways: 1) direct contact with affected sediments through ingestion or dermal contact, and 2) consumption of fish or shellfish that have bioaccumulated chemicals from sediments. Because of the depth of affected sediments and the cold climate, no direct contact with sediments is expected in Ward Cove. People could come into contact with sediments, however, at the mouth of Ward Creek, in an area used for recreational fishing and wading. Site-related chemicals were not elevated at the mouth of Ward Creek. Exposure to site-related chemicals resulting from direct contact with sediments is considered to be highly unlikely. Thus, exposure to chemicals in fish or shellfish that have bioaccumulated these chemicals from sediments was identified as the

only complete exposure pathway and was used as the basis to identify chemicals in sediments with the potential to pose risks to human health. Risk estimates for direct contact with sediments are provided in Appendix H, however, to provide a worst-case analysis.

6.1.2 Site-Specific Consumption Rates

Seafood consumption rates are difficult to identify precisely and may differ greatly between population groups. Conservative consumption rates for fish and shellfish were identified through discussions with ADFG. The Ketchikan area includes people who rely heavily on seafood in their diet (i.e., subsistence populations). Therefore, screening to identify CoCs used conservative consumption rates of 65 g/day of fish and 11 g/day of shellfish, compiled in a data package provided by ADFG and described as representative of average seafood consumption rates for a subsistence community in the area (Wolfe 1995, pers. comm.). These rates were derived by ADFG by dividing the mean edible pounds of all the fish and shellfish[6] harvested per year in Saxman, Alaska, a predominantly Native Alaskan community, by the Saxman population.

Use of harvest rate data to represent consumption rates is a conservative means to evaluate consumption because not all of the fish and shellfish harvested in the community would be consumed in that community. For example, ADFG harvest data were also used in a recent investigation to estimate consumption rates at the 50th, 90th, and 95th percentile for five regions in Alaska (IDM 1997). Where both harvest rate and consumption rate data were available for fish and shellfish, harvest rates consistently overestimated seafood consumption (IDM 1997). The IDM (1997) estimates were considered for use in the risk assessment, but were not selected because the estimates were regional and were based on the same ADFG harvest data used in this evaluation. As indicated by ADFG (Wolfe 1998b, pers. comm.), local indicators of consumption provide a more accurate basis for risk assessment than regional values. In addition, the IDM (1997) estimation of upper percentile consumption rates from harvest rates is highly uncertain given the finding that harvest rates consistently overestimate consumption rates.

The use of fish consumption rates representative of average rates in a subsistence community is also a protective means to evaluate Ward Cove risks given that Ward Cove is designated as a nonsubsistence area (per 18 AAC Parts 1, 2, and 99). A nonsubsistence area is an area or community where dependence upon subsistence is not a principal characteristic of the economy, culture, and way of life of the area or community [see 5 AAC 99.016(a)]. Ordinary fishing and gathering are allowed. Ward Cove is not designated for Customary and Traditional Use. Thus, the use of fish consumption rates representative of subsistence use in nearby Saxman, Alaska, is likely to overestimate exposures for many residents in Ketchikan, Alaska.

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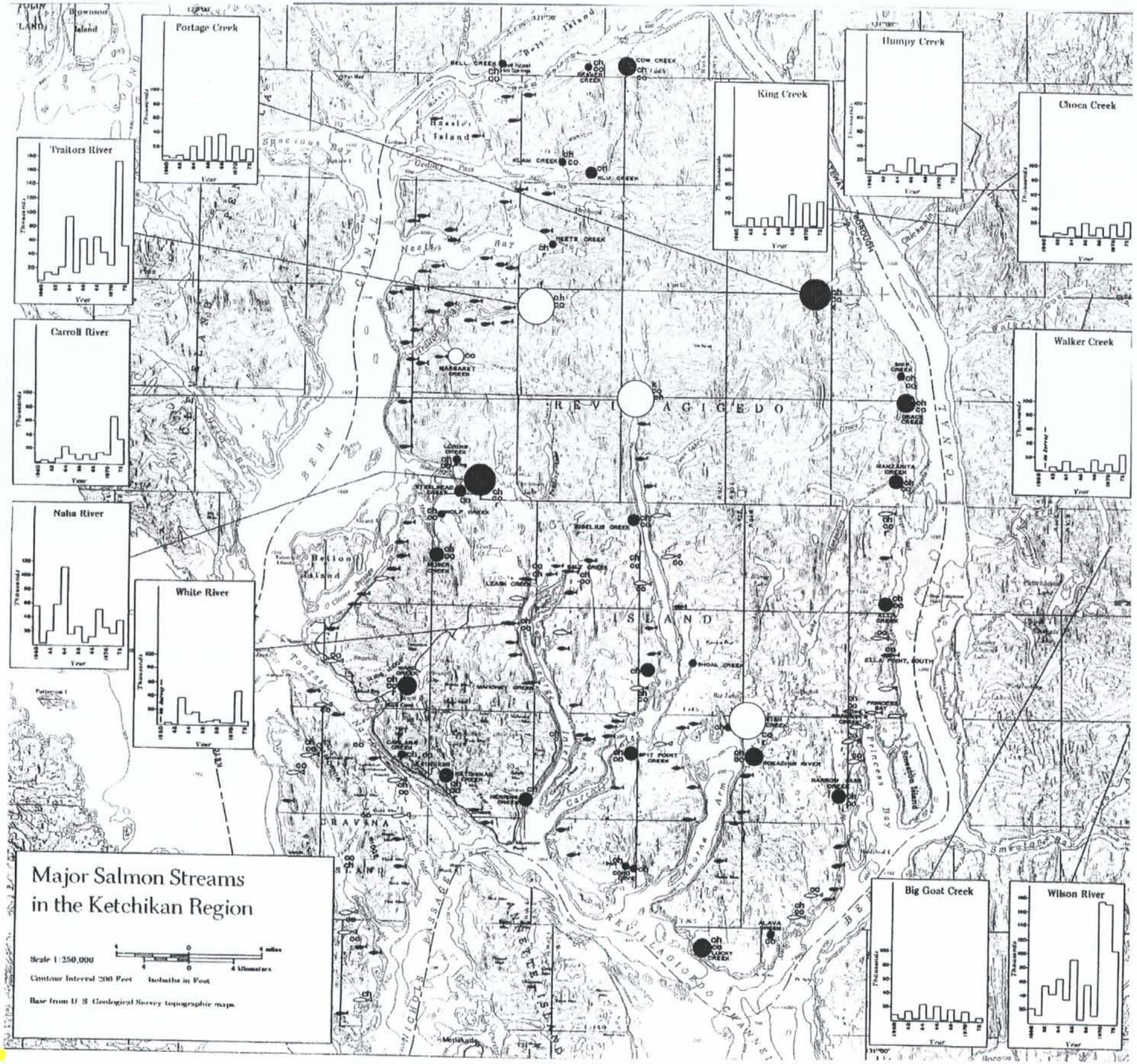
[6] Fish consumption rates were based on harvest data for all fish. Shellfish consumption rates were based on the ADFG harvest category "Marine Invertebrates," which included the following subcategories: abalone, crab, scallops, chitons, octopus, sea cucumber, sea urchin, shrimp, and "unknown" (Wolfe 1995, pers. comm.).

Creel survey data for Ward Cove and Ward Creek are also available. These data are from evaluations of catch and harvest of steelhead and of the coho hatchery in Ward Creek (Freeman 1998, pers. comm.; Hubartt 1998, pers. comm.). These surveys were considered as a basis for estimates, but were not selected because they are representative of lower fish consumption rates than those identified by ADFG for Saxman, Alaska. Specifically, the creel surveys are focused on recreational uses rather than subsistence uses and on salmon rather than all fish and shellfish people might consume.

After consideration of the available sources for seafood consumption rates, the ADFG data set for Saxman, Alaska, was selected as the most representative for evaluating potential high-level fish and shellfish consumption from Ward Cove. Use of average intake rates based on Saxman data provides a health protective means to evaluate intake in the Ketchikan area because Saxman data are representative of a sensitive subpopulation (i.e., predominantly native groups) and the population in Ketchikan is both native and non-native (Wolfe 1998a, pers. comm.). Although these subsistence level consumption rates are likely to greatly overestimate seafood consumption in the general population, they were used here to provide a means to screen site data for CoCs for all hypothetical site users.

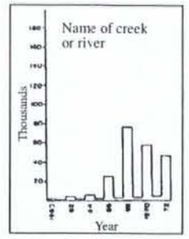
While seafood consumption rates may be relatively high for some communities within the Ketchikan area, Ward Cove is one of many fishing areas available to area residents. Fishing in the Ward Cove area is limited and primarily takes place at the outlet of Ward Creek, where anglers predominantly take salmon when they are present during 1–2 months of the year. Fishing from the shores of Ward Cove is limited, and log rafts and permanent structures in the Cove limit access to site areas by boat. Collection of shellfish is uncertain but is expected to be limited (Freeman 1995, pers. comm.). In screening site data for identification of CoCs, seafood consumption rates were combined with a fractional intake estimate of 5 percent (i.e., 0.05) to account for the availability of many more attractive alternative fishing locations in the area. This fractional intake estimate also accounts for the fact that salmon, the most popular fish species for human consumption in the area, are migratory, thus limiting (or eliminating) the opportunity for salmon to bioaccumulate chemicals from Ward Cove sediments.

The fractional intake is not intended to account for any reduction in use of Ward Cove resulting from current conditions and instead is based only on geographic considerations and on the migratory nature of the primary fish caught in Ward Creek and Ward Cove. While people may be collecting rockfish from Ward Cove, the data from a composite sample of five rockfish collected in Ward Cove indicated that concentrations were below background concentrations (Section 6.2.1). Figure 6-1 identifies major salmon streams in the Ketchikan region as presented in the Ketchikan Atlas (Martinson and Kuklok 1977). As is indicated there, the region includes abundant salmon fishing locations, of which Ward Creek off of Ward Cove is a very small fraction. Based on the wide availability of areas to fish, a fractional intake of less than 5 percent might be warranted, but 5 percent was applied here to provide a health protective means to evaluate exposure to populations living near Ward Cove.



LEGEND

Highest Single Pink Salmon Count Observed



Escapement Number of Fish Counted

- ○ Less than 2,000 fish
- ○ 2,001 to less than 10,000 fish
- ○ 10,001 to less than 20,000 fish
- ○ 20,001 to less than 40,000 fish

Major Species in Escapement

- ● Pink salmon
- ◯ Pink salmon and chum salmon

Other Species Present

- ch Chum or dog salmon
- co Coho or silver salmon
- r Red or sockeye salmon
- k King or Chinook salmon

Other Species Present

- Pink salmon major species, no escapement magnitude data available
- Coho salmon major species, no escapement magnitude data available
- Pink and chum salmon major species, no escapement magnitude data available
- Pink salmon and other species as noted, no escapement nor species relative abundance data available
- Other small streams in anadromous fish catalog, not surveyed and with little existing information. They are presumed to be mainly small pink systems.

Source: U.S. Fish and Wildlife Service, 1959, Stream Catalog of the Eastern Section of Ketchikan Management District in Southeast Alaska; and Alaska Department of Fish and Game, 1978.

Figure 6-1. Major salmon streams in the Ketchikan region.

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Seafood consumption rates used here are expected to overestimate exposures for most people who use Ward Cove. For example, the consumption rate used here of 3.8 g/day (derived by combining the seafood consumption rate of 76 g/day with the fractional intake of 0.05) is nearly identical to the comparable seafood consumption rate of 3.9 g/day used in human health risk evaluations for subsistence anglers in Tongass Narrows (ENSR 1996a). The Tongass Narrows risk assessment was conducted as part of the KPC NPDES discharge permitting process and included consideration of risks in a much larger exposure area than is represented by the affected area within Ward Cove. Therefore, application of these consumption rates to the Ward Cove area provides a conservative means to evaluate risks.

A discussion of uncertainties in seafood consumption rates is provided in Appendix H, and the effects of applying an alternative fractional intake estimate of 10 percent and a 70-year exposure duration are discussed in Appendix G.

6.2 SCREENING CoCs FOR HUMAN HEALTH

As described above, seafood consumption was identified as the only complete exposure pathway for human health. Chemicals in seafood would be identified as CoCs in instances where concentrations exceed both background and risk-based concentrations. Two sources of data were used to evaluate the potential for exposure to chemicals in seafood: 1) estimated tissue concentrations in fish and shellfish derived from chemical concentrations in current sediment samples through application of published BSAFs; and 2) bioaccumulation data for Ward Cove mussels, clams, crabs, and finfish collected in previous investigations (EVS 1996; ENSR 1995c,d; Spannagel 1991; Crook 1995, pers. comm.) (Section 4.4). Tissue concentrations were estimated for all chemicals that had EPA-derived toxicity values for use in human health risk assessment through application of BSAFs to measured concentrations of chemicals in sediment samples. Tissue concentrations identified by previous investigators were also evaluated, including results of PCDDs/Fs and total and methylmercury analyses in mussel and clam samples from Ward Cove and Tongass Narrows (ENSR 1995c,d; EVS 1996) and results of PCDD/F analyses in crab and finfish samples collected in or near Ward Cove (Tables D1-1, D1-2, D1-4, and D1-5 in Appendix D1). Section 4.4 provides a description of both of these data sets.

In screening site data for CoCs for human health, the maximum estimated tissue concentration for each chemical was compared with background concentrations (where available) for chemicals in seafood and with risk-based concentrations derived on the basis of seafood consumption. Fish tissue concentration estimates were used in comparisons for all chemicals except PAHs because fish represents the majority of intake. Higher estimated concentrations of some chemicals in other seafood would be offset by lower (or absent) consumption rates for other seafood items. PAH evaluations were based on the highest estimated shellfish concentration because PAHs do not bioaccumulate in fish (ATSDR 1989). Maximum measured concentrations of PCDDs/Fs and mercury in biota samples from Ward Cove were also compared with background and risk-based concentrations. As described in Section 4.4, because estimated concentrations were consistently

higher than measured concentrations, estimated concentrations appear to provide a conservative means to evaluate site risks where tissue data are unavailable. These comparisons are presented in Table 6-1 and described in the following sections.

6.2.1 Comparison with Background Concentrations

Concentrations of site chemicals in seafood collected at locations with no known source (i.e., background concentrations) were compared with measured and estimated tissue concentrations. Background concentrations of chemicals in seafood were available for arsenic, mercury, and PCDDs/Fs. U.S. EPA (1992a) identified a maximum background concentration for mercury of 1.8 mg/kg, but this concentration was not used here because it was higher than other values in the EPA data set that EPA identified as being from industrial areas. Sources of background data reviewed include the *National Study of Chemical Residues in Fish* (U.S. EPA 1992a), a review of arsenic concentrations and hazards (Eisler 1994), EPA's recent reevaluation of the toxicity and exposure characteristics of dioxin-like compounds (U.S. EPA 1994d), a review of background concentrations of PCDDs/Fs in marine fishes (Schecter et al. 1997), and background samples from the APC investigation near Sitka, Alaska (Delta Toxicology 1995). The estimated concentration of arsenic did not exceed background concentrations identified in the contiguous United States, suggesting that site-related effects are nonexistent or minimal (Table 6-1).

Background concentrations of PCDDs/Fs include a maximum TEC of 1.2 ng/kg for fish fillets from freshwater and estuarine waters identified in the contiguous United States (U.S. EPA 1994d), a TEC of 0.25 ng/kg for marine fish fillets, and PCDD/F data from two reference locations (i.e., Deep Inlet and Katlian Bay) identified in an investigation of the APC site in Sitka, Alaska (Delta Toxicology 1995). The maximum concentration was 0.2 ng/kg in a crab collected at Katlian Bay. Additional samples had a TEC of 0.1 ng/kg measured in each of the following: two analyses of one mussel each from Deep Inlet, and two analyses of one mussel each and one crab from Katlian Bay. In addition, a PCDD/F TEC of 0.23 ng/kg was reported in a composite sample of five salmon collected from Mountain Point, which was identified as a reference location for Ward Cove (Spannagel 1991).

PCDD/F TECs measured in composite samples of five rockfish from Ward Cove and in five composite samples of five clams each that were exposed to sediments from Ward Cove were all below or similar to background concentrations identified in Alaska and in the contiguous United States (Appendix D). The TEC for rockfish of 0.26 ng/kg was similar to the reference location at Mountain Point of 0.23 ng/kg and the background concentration of 0.25 ng/kg for marine filets in the contiguous United States (Tables D1-2 and D1-4 in Appendix D1).

Estimated fish tissue concentrations of PCDDs/Fs exceeded the maximum background concentration of 1.2 ng/kg identified for fish fillets from freshwater and estuarine waters in the contiguous United States (U.S. EPA 1994d). In addition, the measured TEC for a composite sample of muscle tissue in five crabs of 0.35 ng/kg exceeded the reference

TABLE 6-1. IDENTIFICATION OF CoCs FOR HUMAN HEALTH BASED ON MAXIMUM ESTIMATED OR MEASURED SEAFOOD CONCENTRATIONS

Chemical	Maximum Sediment Concentration[a] (mg/kg dw)	Maximum Seafood Concentration[b] (mg/kg ww)	Oral CSF[c] $(mg/kg\text{-}day)^{-1}$	Oral RfD[c] (mg/kg-day)	Background Concentration (mg/kg ww)	Risk-Based[d] Concentration (mg/kg ww)	Identified as CoC for Human Health
Metals and Organometallic Compounds							
Arsenic[e]	39	0.12	1.5	0.0003	0.15 [e]	0.30	No
Cadmium	7.3	3.7	ND	0.001	NA	19	No
Total mercury (sediments; methylmercury in tissues)	0.7	0.07	ND	0.0001	NA [f]	1.9	No
Total mercury (measured)		0.026			NA [f]	1.9	No
Zinc	396	495	ND	0.3	NA	5,800	No
Organic Compounds							
Phenol	0.91	0.47	ND	0.6	NA	12,000	No
4-Methylphenol	17	8.8	ND	0.005	NA	96	No
PCDD/F (TEC)	4.6×10^{-5}	3.9×10^{-5}	150,000	ND	0.2×10^{-6} [g]	3.0×10^{-6}	Yes
PCDD/F (TEC) (measured tissue data)		0.78×10^{-6} [h]			0.2×10^{-6} [g]	3.0×10^{-6}	No
PAHs							
Carcinogenic PAH	0.41	0.072	7.3	ND	NA	0.42	No
Fluoranthene	2.2	0.39	ND	0.04	NA	5,300	No
Pyrene	1.8	0.32	ND	0.03	NA	4,000	No
Acenaphthene	0.50	0.088	ND	0.06	NA	8,000	No
Anthracene	0.26	0.046	ND	0.3	NA	40,000	No
Fluorene	0.47	0.083	ND	0.04	NA	5,300	No

Note: [ ] - values updated with 1997 data

BSAF - biota-sediment accumulation factor
CoC - chemical of concern
CSF - carcinogenic slope factor
dw - dry weight
EPA - U.S. Environmental Protection Agency
NA - not available
ND - not determined by EPA or not considered to be a carcinogen
PAH - polycyclic aromatic hydrocarbon
PCDD/F - polychlorinated dibenzo-*p*-dioxin and polychlorinated dibenzofuran
RfD - reference dose
RPC - relative potency concentration for carcinogenic PAHs
TEC - toxic equivalent concentration based on data for 2,3,7,8-tetrachlorodibenzo-*p*-dioxin
ww - wet weight

[a] Concentrations are maximum sediment concentrations, except for phenol, PAHs (RPCs), anthracene, and zinc, which exclude higher sediment concentrations identified at locations remote from the site (i.e., Station 23 at the state airplane ramp and Stations 24 and 25 at the cannery; see Figure 4-25). For undetected concentrations, one-half the detection limit was used in the RPC and TEC calculations.

[b] Concentrations estimated using BSAFs (see text and Table 4-5) except data for PCDD/F (TECs) and mercury as indicated. Concentrations for all substances except PAHs were estimates for fish tissues. Higher estimated concentrations of some chemicals in shellfish would be offset by lower (or absent) site-related intake. PAHs were evaluated based on highest estimated shellfish concentrations because PAHs are assumed not to bioaccumulate in fish (ATSDR 1989).

TABLE 6-1. (cont.)

[c] Toxicity values obtained from either the EPA Health Effects Assessment Summary Tables (HEAST) (May 1997) or EPA Integrated Risk Information System (IRIS) (June 1998).

[d] Risk-based concentrations were derived on the basis of consumption of fish and shellfish combined, for all substances except PAHs. Risk-based concentrations for PAHs were based on consumption of shellfish only because PAHs are assumed not to bioaccumulate in fish.

[e] Estimated total arsenic concentration adjusted assuming 10 percent inorganic arsenic (ICF Kaiser 1996). Background concentration was a measured inorganic arsenic concentration reported in Eisler (1994).

[f] Although a maximum background concentration of 1.8 mg/kg was identified in U.S. EPA (1992a), this value was the highest concentration in the data set, which included seafood from industrial areas, and therefore was not included here.

[g] Background concentration from a study near Sitka, Alaska, in Delta Toxicology (1995).

[h] Maximum TEC in mussels (whole body) in EVS (1996). TECs derived using one-half the detection limit for undetected congeners.

concentration in Sitka, Alaska (Delta Technology 1995), and the background concentration in marine fish of 0.25 ng/kg (Schecter et al. 1997). The TEC for a composite sample of hepatopancreas in five crabs of 10 ng/kg and the TECs measured in caged mussel studies in Ward Cove (i.e., concentrations ranged from 0.18 to 0.78 ng/kg wet weight) were close to or higher than the maximum background concentration identified in the reference location near Sitka and in the contiguous United States. Thus, because TECs exceeded background concentrations, they are further evaluated through comparison with risk-based concentrations.

6.2.2 Comparison with Risk-Based Concentrations

Maximum measured tissue concentrations (i.e., data for PCDDs/Fs and mercury) and maximum estimated tissue concentrations were compared with risk-based concentrations derived using the methods described by U.S. EPA (1996b) and site-specific assumptions about consumption of fish and shellfish described above. Table 6-2 shows the algorithm used to estimate risk-based concentrations in tissues through application of the following conservative assumptions:

- A target excess cancer risk of 10^{-5} for carcinogenic effects or a hazard quotient of 1 for noncarcinogenic effects[7] was used consistent with the approach specified in the draft ADEC guidance for risk assessment (ADEC 1998)
- Consumption of seafood at subsistence levels (i.e., 65 g/day of fish and 11 g/day of shellfish for all substances except PAHs)
  - Evaluation of PAHs was based on consumption of 11 g/day of shellfish only, because PAHs do not bioaccumulate in fishes as a result of rapid metabolism (ATSDR 1989)
- Five percent of all fish and shellfish consumed is collected from areas with affected sediments, and all seafood collected from those areas has bioaccumulated chemicals from sediments
- Consumption of seafood from the Ward Cove area for 30 years. (Although a 30-year exposure duration is identified by EPA as the 90th percentile of time that U.S. populations remain in one residence, a 70-year exposure duration was also evaluated in Appendix G because of concerns raised regarding residence time.)

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[7] A 1×10^{-5} cancer risk estimate represents a one in a hundred thousand additional probability that an individual may develop cancer over a lifetime as a result of the exposure conditions evaluated. A hazard quotient is the ratio of the estimated exposure over a specified time to a reference dose assumed to represent a safe exposure level. Where hazard quotients are less than 1, no adverse effects are expected.

TABLE 6-2. RISK-BASED CONCENTRATION ALGORITHM FOR FISH AND SHELLFISH CONSUMPTION

Risk-based concentration (carcinogenic effects) (mg/kg ww) =

$$\frac{TR \times AT_c \times BW}{CF \times EF \times ED \times FI \times IR \times CSF}$$

Risk-based concentration (noncarcinogenic effects) (mg/kg ww) =

$$\frac{THQ \times AT_n \times BW \times RfD}{CF \times EF \times ED \times FI \times IR}$$

where:

TR	=	target risk (unitless)
THQ	=	target hazard quotient (unitless)
CF	=	conversion factor (kg/g)
EF	=	exposure frequency (days/year)
ED	=	exposure duration (years)
FI	=	fraction ingested from contaminated source (unitless)
IR	=	ingestion rate of fish/shellfish (g/day)
BW	=	body weight (kg)
AT	=	averaging time: - carcinogenic effects: 70 years x 365 days/year - noncarcinogenic effects: ED x 365 days/year
CSF	=	carcinogenic slope factor $(mg/kg\text{-}day)^{-1}$ (chemical specific)
RfD	=	reference dose (mg/kg/day) (chemical specific)

Exposure Assumptions[a]

Parameter		
TR	1×10^{-5} [b]	
THQ	1	
CF	1×10^{-3}	
EF	350	
ED	30	
FI	0.05[c]	
BW	70	
	Fish	Shellfish
IR[d]	65	11

[a] Algorithms and exposure assumptions from U.S. EPA (1989e, 1991b), unless otherwise specified.

[b] Based on the draft ADEC (1998) guidance.

[c] Based on best professional judgment.

[d] Ingestion rates represent average seafood consumption rates for a subsistence community in the Ketchikan area (Wolfe 1995, pers. comm.). (See Section 6.1.2).

- All chemicals in fish or shellfish were considered to be completely bioavailable (i.e., complete absorption from the gastrointestinal tract). This assumption would result in an overestimate of risks in instances where chemicals are incompletely absorbed and exposures are lower.

Risk-based concentrations were calculated using a target risk level of 10^{-5} for carcinogens, which ADEC has identified in draft guidance (ADEC 1998) as the basis for screening evaluations. This target risk level is lower than the upper end of the range of cancer risks of 10^{-4} to 10^{-6} identified by EPA and ADEC as the acceptable risk range (U.S. EPA 1990; ADEC 1998). Thus, use of this target risk level incorporates a measure of protection for exposure to carcinogens at the site.

Risk-based concentrations were calculated for all chemicals that had EPA-derived toxicity values. Although some chemicals associated with wood products could not be included in the screening because of the lack of toxicity data, detected compounds (Table A1-5 in Appendix A1) were present at concentrations much lower than risk-based concentrations for other non-chlorinated organic chemicals such as methylphenol, naphthalene, or pyrene (Table 6-1). Thus, human health risks associated with these compounds are expected to be minimal or nonexistent.

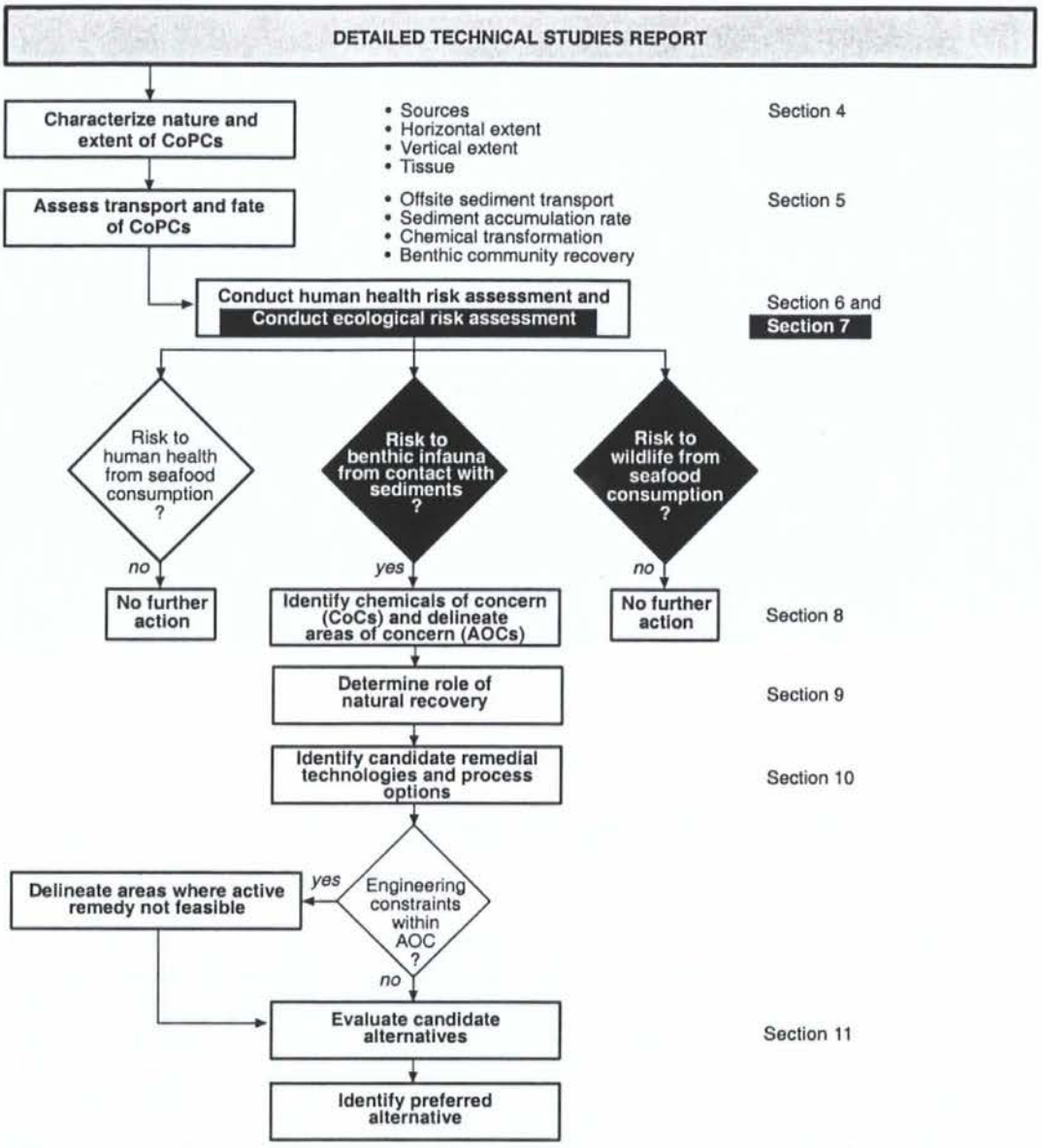
Despite the conservative assumptions used in deriving risk-based concentrations and in estimating tissue concentrations (i.e., use of maximum sediment concentrations and maximum BSAFs [Section 4.4]), estimated tissue concentrations exceeded risk-based concentrations only for PCDD/F TECs. The maximum estimated seafood concentration of 3.9×10^{-5} mg/kg wet weight was approximately 13 times higher than the risk-based concentration of 3.0×10^{-6} mg/kg wet weight and thus PCDDs/Fs would be identified as a CoC on this basis (Table 6-1). In contrast with the estimated PCDD/F TEC in tissue, however, the maximum measured PCDD/F TEC of 0.78×10^{-6} mg/kg wet weight in caged mussels was lower than the risk-based concentration for PCDD/F (TEC). Measured PCDD/F TECs in rockfish, salmon, and clams were near or below background concentrations. Tissue concentrations are a more reliable basis for identifying CoCs than estimated concentrations because of the uncertainty in applying BSAF estimates. In addition, BSAF-derived estimates represent whole-body concentrations, which tend to overestimate concentrations in tissues consumed by people. Thus, given consideration of both the estimated and measured tissue concentrations, no CoCs were identified for human health.

6.3 CONCLUSIONS

The human health risk assessment for Ward Cove sediments included numerous conservative assumptions that are likely to overestimate risks for most people using Ward Cove. These assumptions included an assumed subsistence level seafood consumption rate, use of the area over a 30-year exposure period, and evaluations based on maximum sediment or tissue concentrations. Despite these conservative assumptions, no CoCs were identified for human health. Although there are uncertainties associated with this risk estimate,

assumptions used here tend to overestimate, rather than underestimate risks. (Uncertainties associated with the human health risk assessment are discussed in more detail in Appendices G and H.) Thus, risks, if any, appear to be within levels considered acceptable by regulatory agencies.

7. ECOLOGICAL EVALUATION



In this section, the potential for CoPCs to cause sediment toxicity or to bioaccumulate in the food web of Ward Cove is evaluated. The objective of this evaluation is to identify any potential AOCs in the Cove where sediments may pose risks of adverse effects to ecological receptors. The sediment toxicity assessment is based on concentrations of CoPCs in Ward Cove sediments and results of four kinds of sediment toxicity tests conducted with sensitive and representative benthic macroinvertebrate test species. The food-web assessment focuses on concentrations of CoPCs in Ward Cove sediments and

the potential risks they pose to representative birds and mammals at the top of the site-specific food web. Those risks are predicted using food-web models.

7.1 SEDIMENT TOXICITY ASSESSMENT

The objective of the sediment toxicity assessment is to identify any potential AOCs in Ward Cove that may pose a potential risk of adverse effects to organisms that live within or on the sediments of the Cove. Those organisms are considered at risk of exposure to CoPCs in the Cove because historical studies have documented that CoPC concentrations in sediments are elevated in parts of the Cove.

The sediment toxicity assessment is based primarily on two kinds of information: concentrations of CoPCs in Ward Cove sediments and results of four kinds of sediment toxicity tests conducted with sensitive and representative test species. Both kinds of evaluations are based largely on the information collected during 1996 (28 stations in Ward Cove and the 2 reference stations in Moser Bay) and 1997 (33 stations in Ward Cove and 2 reference stations in Moser Bay). However, historical data collected in 1994 and 1995 (ENSR 1994, 1995b) at 12 NPDES stations sampled in Ward Cove are also compared with the 1996 and 1997 data to evaluate the consistency of the various data sets and to determine whether data collected in 1994 and 1995 (based on the top 2 cm of sediment) differed substantially from data collected in 1996 and 1997 (based on the top 10 cm of sediment).

In this section, the relationship between chemical concentrations and sediment toxicity is evaluated. The distributions of various chemicals in surface sediments throughout Ward Cove are discussed in detail in Section 4.2. Tables 7-1 and 7-2 present information on sediment grain size distribution and conventional CoPCs for 1996 and 1997. Table 7-2 also addresses concentrations of 4-methylphenol because it was the only non-conventional CoPC evaluated in 1997. Table 7-3 presents information on the non-conventional CoPCs evaluated in 1996. Tables 7-4 through 7-7 address additional chemicals that were measured at a subset of stations during 1996 and 1997 to comply with the requirements of the KPC NPDES permit. In those tables, nonionic organic compounds (i.e., 2,3,7,8-TCDD, TCDD TEC, and PAH compounds) are normalized to the organic carbon content of the sediments so that they can be compared more directly with sediment quality values. Finally, Table 7-8 presents information collected in 1996 on compounds commonly associated with pulp mills.

In the remainder of this section, results of the sediment toxicity tests are discussed, the observed CoPC concentrations in Cove sediments are compared with sediment quality values, and results of the sediment chemistry and sediment toxicity evaluations are compared. The delineation of potential AOCs based on the sediment chemistry and sediment toxicity results discussed in this section is presented in Section 8.

TABLE 7-1. SUMMARY OF SEDIMENT GRAIN SIZE AND CONVENTIONAL CoPCs FOR SEDIMENTS IN WARD COVE AND MOSER BAY IN 1996 AND COMPARISON WITH SEDIMENT QUALITY VALUES

Station	Fines[a] (percent)	TOC (percent)	Total Ammonia (mg/kg)	Total Sulfide (mg/kg)	BOD (g/kg)	COD (g/kg)
Ward Cove-Subtidal						
1	53	32 **	310 **	1,700	16 *	480
2	30	14	220 **	1,200	9.9	330
3	24	22	14	5,300	7.3	250
4	64	26	97	6,500	12 *	470
5	31	36 **	67	5,400	10	590 *
6	50	33 **	360 **	2,200	13 *	540
7	69	26	74	1,800	8.7	620 *
8	66	24	100	2,700	12 *	2,400 **
9	56	27	82	4,500	19 *	550
10	58	27	99	5,500	9.8	340
11	26	14	50	1,500	6.4	190
12	53	24	260 **	2,700	10	520
13	77	22	150 **	4,300	8.3	440
14	70	25	130 **	2,200	16 *	190
15	61	25	83	2,700	6.0	490
16	65	31	81	16,000	18 *	620 *
17	18	31	11	27,000	7.6	150
18	6.1	1.1	13	150	1.4	17
19	74	18	44	800	9.6	270
20	77	17	84	420	11	120
21	66	21	88	3,500	6.2	420
22	39	5	21	380	3.5	98
23	67	13	14	1,200	7.9	200
24	60	13	34	670	7.0	190
25	57	11	160 **	1,000	9.2	160
26	81	30	66	2,200	8.5	550
27	57	21	43	4,300	10	330
28	81	20	34	2,400	10	330
Moser Bay-Subtidal						
29	57	4	12	590	2.1	71
30	81	5	11	570	4.5	130
$WCSQV_{(1)}$		**31** [b]	**110** [b]	**NA**	**11** [b]	**550** [b]
$WCSQV_{(2)}$		**31** [b]	**120** [b]	**NA**	**37** [b]	**620** [b]

Note: All concentrations reported on dry weight basis.

- * - concentration exceeds $WCSQV_{(1)}$
- ** - concentration exceeds $WCSQV_{(2)}$
- BOD - biochemical oxygen demand
- COD - chemical oxygen demand
- CoPC - chemical of potential concern
- NA - sediment quality values not available
- TOC - total organic carbon
- $WCSQV_{(1)}$ - Ward Cove sediment quality value analogous to sediment quality standard
- $WCSQV_{(2)}$ - Ward Cove sediment quality value analogous to minimum cleanup level

[a] Fine-grained sediments (silt + clay).

[b] Site-specific sediment quality value.

TABLE 7-2. SUMMARY OF SEDIMENT GRAIN SIZE AND CoPCs FOR SEDIMENTS IN WARD COVE AND MOSER BAY IN 1997 AND COMPARISON WITH SEDIMENT QUALITY VALUES

Station	Fines[a] (percent)	TOC (percent)	Total Ammonia (mg/kg)	Total Sulfide (mg/kg)	BOD (g/kg)	COD (g/kg)	4-Methyl-phenol (µg/kg)
Ward Cove-Subtidal							
2	45	33 **	85	4,500	45 *	12	15,000 **
3	53	30	80	500	46 *	10	6,200 **
4	66	25	150 **	3,700	64 *	13	4,500 **
5	55	38 **	57	2,300	9.2	5.6	16,000 **
7	58	26	120 *	1,900	8.0	10	7,500 **
11	27	19	34	2,300	14 *	16	380
12	35	21	240 **	1,900	6.4	7.8	8,300 **
13	72	22	320 **	2,700	12 *	7.0	1,700 *
16	59	28	40	12,000	13 *	16	1,200
17	59	28	99	50	10	10	570
18	7.5	4.0	13	310	1.6	2.2	26
19	84	17	110	5,500	8.5	11	730
22	34	4.0	19	560	3.5	6.5	24
23	80	9.0	86	3,900	37 *	26	170
25	50	13	120 *	3,800	34 *	30	6,600 **
27	65	20	47	4,500	34 *	12	470
28	56	19	34	4,400	32 *	5.6	802
31	80	21	510 **	11,000	11	13	17,000 **
32	47	23	82	13,000	9.1	7.1	2,700 **
33	18	5.1	23	1,600	1.7	4.5	980
34	55	29	120 *	2,300	10	12	5,100 **
35	59	30	120 *	3,300	14 *	10	460
37	59	31	54	2,700	7.1	8.7	4,400 **
38	46	34 **	260 **	6,700	65 *	15	8,300 **
39	63	23	110	2,700	7.7	8.3	1,300
40	63	23	80	3,800	7.8	11	1,000
41	58	22	58	48	6.4	52	640
42	65	24	82	2,000	6.9	11	5,700 **
43	81	18	110	9,700	7.4	10	1,000
44	69	26	690 **	2,300	13 *	15	9,000 **
45	88	21	170 **	4,800	9.1	12	2,400 **
47	38	26	120 *	3,000	7.1	7.9	1,800 **
48	70	25	300 **	3,900	9.2	19	1,100
Moser Bay-Subtidal							
29	53	3.6	16	240	1.7	3.5	10 *U*
30	91	5.3	18	530	3.0	4.5	15 *U*
Ward Cove-Intertidal							
50	6.2	1.3	3.2	20 *U*	0.7	1.3	10 *U*
51	31	5.1	11	1,000	8.7	6.2	231
$WCSQV_{(1)}$		**31** [b]	**110** [b]	**NA**	**11** [b]	**550** [b]	**1,300** [b]
$WCSQV_{(2)}$		**31** [b]	**120** [b]	**NA**	**37** [b]	**620** [b]	**1,700** [b]

Note: All concentrations reported on dry weight basis.

- * - concentration exceeds $WCSQV_{(1)}$
- ** - concentration exceeds $WCSQV_{(2)}$
- BOD - biochemical oxygen demand
- COD - chemical oxygen demand
- CoPC - chemical of potential concern
- NA - sediment quality values not available
- TOC - total organic carbon
- *U* - undetected at concentration listed
- $WCSQV_{(1)}$ - Ward Cove sediment quality value analogous to sediment quality standard
- $WCSQV_{(2)}$ - Ward Cove sediment quality value analogous to minimum cleanup level

[a] Fine-grained sediments (silt + clay).

[b] Site-specific sediment quality value.

TABLE 7-3. SUMMARY OF CoPCs FOR SEDIMENTS IN WARD COVE AND MOSER BAY IN 1996 AND COMPARISON WITH SEDIMENT QUALITY VALUES

	Metals			Organic Compounds			
Station	Cadmium (mg/kg dry weight)	Total Mercury (mg/kg dry weight)	Zinc (mg/kg dry weight)	Phenol (µg/kg dry weight)	4-Methyl-phenol (µg/kg dry weight)	2,3,7,8-TCDD[a] (µg/kg organic carbon)	TCDD TEC[a,b] (µg/kg organic carbon)
Ward Cove-Subtidal							
1	4.6	0.10	205	240	6,000 **	0.02	0.24
2	2.3	0.10 *U*	135	510 *	11,000 **	0.01 *U*	0.23
3	1.3	0.70 **	214	110	5,600 **	0.01 *U*	0.23
4	4.3	0.20	277	170	2,900 **	0.03	0.46
5	1.3	0.10 *U*	117	150	860	0.02 *U*	0.14
6	4.8	0.10	165	97	8,300 **	0.01 *U*	0.15
7	7.3 **	0.25	197	200 *U*	1,700 *	0.02 *U*	0.46
8	6.1 *	0.20	203	250 *U*	1,400 *	ND	ND
9	5.0	0.10	226	250 *U*	1,400 *	0.01 *U*	0.12
10	2.8	0.10 *U*	270	250 *U*	250 *U*	ND	ND
11	2.4	0.10 *U*	115	200 *U*	200 *U*	0.01 *U*	0.06
12	5.5 *	0.10	200	200 *U*	620	0.01	0.17
13	5.2 *	0.10	142	200 *U*	390	0.01 *U*	0.08
14	6.7 *	0.10	188	200 *U*	1,000	0.02	0.26
15	4.8	0.10	121	200 *U*	220	0.01 *U*	0.14
16	3.7	0.10 *U*	190	360	250 *U*	0.01 *U*	0.07
17	1.0	0.10 *U*	192	250 *U*	250 *U*	0.01 *U*	0.03
18	0.2	0.10 *U*	43	15	20 *U*	0.06 *U*	0.10
19	3.7	0.10	110	250 *U*	250 *U*	0.01 *U*	0.11
20	5.3 *	0.20	147	200 *U*	470	0.01 *U*	0.18
21	5.2 *	0.10	135	250 *U*	250 *U*	0.01 *U*	0.16
22	1.0	0.10 *U*	69	200 *U*	200 *U*	0.02 *U*	0.10
23	2.5	0.20	159	46	49	0.02 *U*	0.06
24	3.5	0.20	242	250 *U*	250 *U*	0.02 *U*	0.22
25	3.7	0.10	340	130	1,700 *	0.02 *U*	0.21
26	4.0	0.10	144	200 *U*	200 *U*	0.01 *U*	0.14
27	4.7	0.10	133	200 *U*	200 *U*	0.03 *U*	0.05
28	2.6	0.10 *U*	171	200 *U*	200 *U*	ND	ND
Moser Bay-Subtidal							
29	0.33	0.10 *U*	78	20 *U*	20 *U*	ND	ND
30	1.4	0.10 *U*	70	20 *U*	20 *U*	0.02 *U*	0.03
SQS/WCSQV$_{(1)}$	**5.1** [c]	**0.41** [c]	**410** [c]	**420** [c]	**1,300** [d]	**NA**	**NA**
MCUL/WCSQV$_{(2)}$	**6.7** [c]	**0.58** [c]	**960** [c]	**1,200** [c]	**1,700** [d]	**NA**	**NA**

Note: * - concentration exceeds sediment quality standard (SQS)
** - concentration exceeds minimum cleanup level (MCUL)
CoPC - chemical of potential concern
NA - sediment quality values not available
ND - no data
TCDD - tetrachlorodibenzo-*p*-dioxin
TEC - toxic equivalent concentration
TOC - total organic carbon
U - undetected at concentration listed
WCSQV$_{(1)}$ - Ward Cove sediment quality value analogous to sediment quality standard
WCSQV$_{(2)}$ - Ward Cove sediment quality value analogous to minimum cleanup level

[a] Concentrations are normalized to station-specific TOC concentrations, except that a TOC concentration of 10 percent was used for all station-specific values that were ≥ 10 percent.

[b] Detection limits are included in the sum at half their value.

[c] Washington State sediment management standard.

[d] Site-specific sediment quality value.

TABLE 7-4. SUMMARY OF NPDES CHEMICALS IN WARD COVE SEDIMENTS IN 1996 AND COMPARISON WITH WASHINGTON STATE SEDIMENT MANAGEMENT STANDARDS

Station	AVS (mg/kg)	Arsenic (mg/kg)	Methyl-mercury (µg/kg)	Benzoic Acid (µg/kg)	EOX (mg/kg)
Ward Cove-Subtidal					
2	2,200	18	0.6	990 **	10 *U*
3	2,800	16	0.8	500 *U*	10 *U*
4	2,400	29	10	1,600 **	10 *U*
5	2,000	8.5	0.6	500 *U*	10 *U*
11	1,500	17	3.5	500 *U*	10 *U*
13	320	33	6.9	500 *U*	10 *U*
16	13,000	19	1.0	500 *U*	10 *U*
18	240	2.7	0.8	100 *U*	10 *U*
22	540	11	5.4	500 *U*	10 *U*
23	2,100	29	9.5	500 *U*	10 *U*
25	4,200	24	0.5	500 *U*	10 *U*
27	3,200	26	3.1	500 *U*	10 *U*
SQS	**NA**	**57**	**NA**	**650**	**NA**
MCUL	**NA**	**93**	**NA**	**650**	**NA**

Note: All concentrations reported on dry weight basis.

- * - concentration exceeds sediment quality standard (SQS)
- ** - concentration exceeds minimum cleanup level (MCUL)
- AVS - acid-volatile sulfide
- EOX - extractable organic halides
- NA - sediment standards not available
- NPDES - National Pollutant Discharge Elimination System
- *U* - undetected at concentration listed

TABLE 7-5. SUMMARY OF NPDES CHEMICALS IN WARD COVE SEDIMENTS IN 1997 AND COMPARISON WITH WASHINGTON STATE SEDIMENT MANAGEMENT STANDARDS

Station	AVS (mg/kg dry weight)	Arsenic (mg/kg dry weight)	Cadmium (mg/kg dry weight)	Methyl-mercury (ng/kg dry weight)	Total Mercury (mg/kg dry weight)	Zinc (mg/kg dry weight)	Benzoic Acid (µg/kg dry weight)	EOX (mg/kg dry weight)	Phenol (µg/kg dry weight)	2,3,7,8-TCDD[a] (µg/kg organic carbon)	TCDD TEC[a,b] (µg/kg organic carbon)
Ward Cove-Subtidal											
2	1,600	23	3.0	0.43	0.2 *U*	200	100 *U*	20	910 *	0.02	0.22
3	2,500	25	3.6	1.2	0.2 *U*	220	100 *U*	23	200	0.01	0.31
4	4,500	31	4.8	1.3	0.2 *U*	400	870 **	10 *U*	220	0.02	0.45
5	3,700	8.7	1.5	0.55	0.2 *U*	170	100 *U*	10 *U*	910 *	0.01	0.17
11	3,000	17	2.6	0.65	0.2 *U*	100	340	27	53	0.01	0.09
13	4,300	29	4.4	3.6	0.2 *U*	140	540	10 *U*	150	0.02 *U*	0.20
16	17,000	18	2.5	0.54	0.2 *U*	180	400	10 *U*	100	0.01	0.12
18	580	3.6	0.26	0.28	0.2 *U*	39	150	10 *U*	12	0.02 *U*	0.03
22	680	11	0.78	3.4	0.2 *U*	62	63	10 *U*	17	0.02 *U*	0.22
23	3,900	19	2.3	14.3	0.2 *U*	130	270	10 *U*	48	0.01 *U*	0.16
25	5,800	24	5.1	0.22	0.2 *U*	530 *	100 *U*	79	990 *	0.01	0.20
27	5,300	34	5.0	3.6	0.2 *U*	170	600	10 *U*	57	0.01	0.17
Ward Cove-Intertidal											
50	ND	ND	0.14	ND	0.2 *U*	64	62	10 *U*	10 *U*	ND	ND
51	ND	ND	0.48	ND	0.2 *U*	72	120	10 *U*	37	ND	ND
SQS	**NA**	**57**	**5.1**	**NA**	**0.41**	**410**	**650**	**NA**	**420**	**NA**	**NA**
MCUL	**NA**	**93**	**6.7**	**NA**	**0.58**	**960**	**650**	**NA**	**1,200**	**NA**	**NA**

Note: * - concentration exceeds sediment quality standard (SQS)
** - concentration exceeds minimum cleanup level (MCUL)
AVS - acid-volatile sulfide
CoPC - chemical of potential concern
EOX - extractable organic halides
NA - sediment standards not available
ND - no data
NPDES - National Pollutant Discharge Elimination System
TCDD - tetrachlorodibenzo-*p*-dioxin
TEC - toxic equivalent concentration
TOC - total organic carbon
U - undetected at concentration listed

[a] Concentrations are normalized to station-specific TOC concentrations, except that a TOC concentration of 10 percent was used for all station-specific values that were ≥ 10 percent.

[b] Detection limits are included in the sum at half their value.

TABLE 7-6. SUMMARY OF PAH CONCENTRATIONS IN WARD COVE SEDIMENTS IN 1996 AND COMPARISON WITH WASHINGTON STATE SEDIMENT MANAGEMENT STANDARDS[a]

Station	Naphthalene	Acenaphthylene	Acenaphthene	Fluorene	Phenanthrene	Anthracene	2-Methyl-naphthalene	Fluoranthene	Pyrene
Ward Cove-Subtidal									
2	0.86	1.0 *U*	0.68	0.64	4	0.62	0.87	6.3	3.2
3	4.4	1.0 *U*	5.0	4.7	11	2.6	2.8	19	14
4	2.0	0.34	1.7	1.7	6.7	1.9	1.4	13	8.3
5	0.49	1.0 *U*	0.60	0.67	2.7	0.62	0.74	6.9	2.3
11	0.24	1.0 *U*	1.0 *U*	0.20	1.5	0.41	0.22	3.4	2.3
13	0.54	1.0 *U*	1.0 *U*	0.20	1.3	0.34	0.25	2.7	1.7
16	0.12	0.50 *U*	0.32	0.34	0.97	0.49	0.15	3.3	1.9
18	0.09	0.91 *U*	0.91 *U*	0.91 *U*	0.55	0.27	0.91 *U*	1.4	0.7
22	2.2 *U*	0.26	2.2 *U*	0.26	2.4	0.72	2.2 *U*	4.8	4.3
23	0.20	1.1	0.34	0.99	8.5	3.6	0.20	10	12
25	0.24	1.0	0.37	1.1	9.0	3.8	0.22	15	15
27	0.17	1.0 *U*	1.0 *U*	0.21	1.2	0.40	0.18	3.0	2.2
SQS	**99**	**66**	**16**	**23**	**100**	**220**	**38**	**160**	**1,000**
MCUL	**170**	**66**	**57**	**79**	**480**	**1,200**	**64**	**1,200**	**1,400**

TABLE 7-6. (cont.)

Station	Benz[a]-anthracene	Chrysene	Benzo[b]-fluoranthene	Benzo[k]-fluoranthene	Benzo[a]-pyrene	Indeno [1,2,3-cd]-pyrene	Dibenz[a,h]-anthracene	Benzo[ghi]-perylene
Ward Cove-Subtidal								
2	1.1	1.3	0.79	0.52	0.56	0.40	1.0 *U*	0.19
3	4.8	4.5	2.2	1.5	2.2	1.1	0.22	0.79
4	3.5	4.1	2.4	1.7	1.7	1.4	0.39	0.90
5	1.6	1.3	1.0	0.61	0.65	0.36	1.0 *U*	0.19
11	1.6	1.0	0.69	0.51	0.67	0.51	1.0 *U*	0.31
13	0.8	1.0	0.62	0.48	0.46	0.33	1.0 *U*	0.30
16	0.9	1.0	0.50	0.36	0.40	0.25	0.06	0.16
18	0.3	0.36	0.27	0.91 *U*	0.91 *U*	0.09	0.91 *U*	0.09
22	2.2	2.4	1.3	1.6	1.4	0.80	2.2 *U*	0.70
23	7.9	9.5	5.1	4.4	6.2	3.5	0.49	2.5
25	10	13	6.9	5.3	7.5	5.2	0.73	2.9
27	1.1	1.4	0.82	0.54	0.60	0.46	1.0 *U*	0.30
SQS	**110**	**110**	**230** [b]	**230** [b]	**99**	**34**	**12**	**31**
MCUL	**270**	**460**	**450** [b]	**450** [b]	**210**	**88**	**33**	**78**

Note: All concentrations reported as mg/kg organic carbon.

* - concentration exceeds sediment quality standard (SQS)

** - concentration exceeds minimum cleanup level (MCUL)

PAH - polycyclic aromatic hydrocarbon

TOC - total organic carbon

U - undetected at concentration listed

[a] Concentrations are normalized to station-specific TOC concentrations, except that a TOC concentration of 10 percent was used for all station-specific values that were ≥ 10 percent.

[b] Sum of benzo[b]fluoranthene and benzo[k]fluoranthene.

TABLE 7-7. SUMMARY OF PAH CONCENTRATIONS IN WARD COVE SEDIMENTS IN 1997 AND COMPARISON WITH WASHINGTON STATE SEDIMENT MANAGEMENT STANDARDS[a]

Station	Naphthalene	Acenaphthylene	Acenaphthene	Fluorene	Phenanthrene	Anthracene	2-Methyl-naphthalene	Fluoranthene	Pyrene
Ward Cove-Subtidal									
2	1.4	0.20 *U*	0.95	1.1	4.8	1.0	1.5	5.5	4.2
3	2.5	0.20 *U*	2.3	2.6	9.0	2.3	1.7	14	12
4	3.1	0.20 *U*	2.6	3.0	9.2	2.5	2.8	22	18
5	1.9	0.20 *U*	1.4	1.4	3.9	0.70	2.0	5.6	4.4
11	0.37	0.20 *U*	0.20 *U*	0.20 *U*	1.0	0.36	0.46	2.0	1.5
13	1.4	0.20 *U*	0.24	0.38	2.2	0.53	0.63	3.3	2.6
16	0.54	0.20 *U*	0.82	1.1	3.1	0.86	0.79	4.2	3.7
18	0.25 *U*	0.25 *U*	0.25 *U*	0.25 *U*	0.40	0.25 *U*	0.25 *U*	0.74	0.57
22	0.28	0.28	0.25 *U*	0.35	2.4	0.80	0.25	5.7	6.0
23	0.23	0.22	0.22 *U*	0.34	2.6	0.68	0.29	5.1	4.8
25	0.52	0.35	0.42	0.92	5.5	3.25	0.60	9.6	8.3
27	0.51	0.20 *U*	0.31	0.52	2.2	0.84	0.69	3.9	3.6
28	0.31	0.10 *U*	0.33	0.45	1.2	0.46	0.38	3.9	3.1
47	2.0 *U*	2.0 *U*	2.0 *U*	2.0 *U*	3.1	2.0 *U*	2.0 *U*	4.7	4.6
Ward Cove-Intertidal									
50	0.76 *U*	0.76 *U*	0.76 *U*	0.76 *U*	3.2	0.76 *U*	0.76 *U*	11	9.1
51	0.28	0.20 *U*	0.28	0.36	1.7	0.65	0.26	5.1	4.0
SQS	**99**	**66**	**16**	**23**	**100**	**220**	**38**	**160**	**1,000**
MCUL	**170**	**66**	**57**	**79**	**480**	**1,200**	**64**	**1,200**	**1,400**

TABLE 7-7. (cont.)

Station	Benz[a]-anthracene	Chrysene	Benzo[b]-fluoranthene	Benzo[k]-fluoranthene	Benzo[a]-pyrene	Indeno [1,2,3-cd]-pyrene	Dibenz[a,h]-anthracene	Benzo[ghi]-perylene
Ward Cove-Subtidal								
2	1.6	1.4	1.9	0.68	0.80	0.40	0.20 *U*	0.91
3	5.1	5.4	6.7	2.5	2.4	1.2	0.20 *U*	0.61
4	6.6	4.8	5.3	1.8	1.9	0.20 *U*	0.20 *U*	0.64
5	0.92	0.93	0.20 *U*	0.20 *U*	0.32	0.20 *U*	0.20 *U*	0.41
11	0.58	0.49	0.77	0.26	0.45	0.24	0.20 *U*	0.20 *U*
13	0.97	1.3	1.5	0.49	0.63	0.34	0.20 *U*	0.50
16	1.2	1.6	0.81	0.31	0.42	0.23	0.20 *U*	0.20 *U*
18	0.25 *U*	0.30	0.32	0.25 *U*	0.25 *U*	0.25 *U*	0.25 *U*	0.25 *U*
22	3.1	3.7	3.7	1.2	2.6	1.9	0.35	1.6
23	2.3	3.0	3.0	0.97	1.9	1.3	0.24	0.94
25	6.7	5.9	7.4	2.5	3.9	2.3	0.20 *U*	1.6
27	1.7	1.9	2.6	0.86	1.4	0.72	0.20 *U*	0.64
28	1.1	1.0	1.3	0.44	0.54	0.10 *U*	0.10 *U*	0.10 *U*
47	4.6	2.1	2.0 *U*	2.0 *U*	2.0 *U*	2.0 *U*	2.0 *U*	2.0 *U*
Ward Cove-Intertidal								
50	2.1	3.6	3.6	1.4	1.7	0.83	0.76 *U*	0.76 *U*
51	1.6	1.9	2.1	0.77	0.91	0.49	0.20 *U*	0.38
SQS	**110**	**110**	**230** [b]	**230** [b]	**99**	**34**	**12**	**31**
MCUL	**270**	**460**	**450** [b]	**450** [b]	**210**	**88**	**33**	**78**

Note: All concentrations reported as mg/kg organic carbon.

* - concentration exceeds sediment quality standard (SQS)

** - concentration exceeds minimum cleanup level (MCUL)

PAH - polycyclic aromatic hydrocarbon

TOC - total organic carbon

U - undetected at concentration listed

[a] Concentrations are normalized to station-specific TOC concentrations, except that a TOC concentration of 10 percent was used for all station-specific values that were ≥ 10 percent.

[b] Sum of benzo[b]fluoranthene and benzo[k]fluoranthene.

TABLE 7-8. SUMMARY OF PULP MILL COMPOUNDS IN SEDIMENTS OFFSHORE FROM THE KPC FACILITY IN 1996

Chemical	Station				
	25 m Offshore		60 m Offshore		130 m Offshore
	2	4	7	9	16
Resin Acids and Fatty Acids					
Abietic acid	65	45	150	28	18
Dehydroabietic acid	38	34	150	20	12
12-Chlorodehydroabietic acid	3	4.7	22	2.9	7.2 *U*
14-Chlorodehydroabietic acid	1.5 *U*	1.7 *U*	23	1.8 *U*	7.2 *U*
Dichlorodehydroabietic acid	1.5 *U*	2.1	14	1.8 *U*	7.2 *U*
9,10-Dichlorostearic acid	1.5 *U*	1.7 *U*	6.5 *U*	1.8 *U*	7.2 *U*
Pimaric acid	1.5 *U*	1.7 *U*	6.5 *U*	1.8 *U*	7.2 *U*
Isopimaric acid	6.5	6.2	22	4.3	7.2 *U*
Linoleic acid	1.5 *U*	1.7 *U*	6.5 *U*	1.8 *U*	7.2 *U*
Oleic/Linolenic acid	7.2	21	79	10	7.2 *U*
Chlorinated Phenols					
4-Chlorophenol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
2,4-Dichlorophenol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
2,6-Dichlorophenol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
2,4,5-Trichlorophenol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
2,4,6-Trichlorophenol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
2,3,4,6-Tetrachlorophenol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
Pentachlorophenol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
Chlorinated Guaiacols					
4-Chloroguaiacol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
3,4-Dichloroguaiacol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
4,5-Dichloroguaiacol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
4,6-Dichloroguaiacol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
3,4,5-Trichloroguaiacol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
3,4,6-Trichloroguaiacol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
4,5,6-Trichloroguaiacol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
Tetrachloroguaiacol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
Chlorinated Catechols					
4-Chlorocatechol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
3,4-Dichlorocatechol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
3,6-Dichlorocatechol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
4,5-Dichlorocatechol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
3,4,5-Trichlorocatechol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
3,4,6-Trichlorocatechol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
Tetrachlorocatechol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
Vanillins					
5-Chlorovanillin	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
6-Chlorovanillin	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
5,6-Dichlorovanillin	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
Additional Compounds					
2-Chlorosyringaldehyde	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
2,6-Dichlorosyringaldehyde	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*
Trichlorosyringol	1.6 *U*	2.0 *U*	1.9 *U*	2.3 *U*	0.80 *U*

Note: All concentrations reported as mg/kg dry weight.

U - undetected at concentration listed

7.1.1 Reference Area Evaluation

As discussed in Section 2, Moser Bay was selected as the reference area for the Phase 1 and Phase 2 investigations of Ward Cove sediments. Moser Bay was selected as the reference area for several reasons:

- Moser Bay is located near Ward Cove (within 25 km) and is similar to the Cove with respect to size, shoreline configuration, and presence of a tributary at its head (Figure 1-1)
- There are no major sources of hazardous substances in Moser Bay
- The shoreline of Moser Bay is largely undeveloped, and anthropogenic uses of the bay are primarily limited to boating and recreational fishing
- There is no potential for Moser Bay to be influenced by CoPCs from Ward Cove
- Moser Bay is used as the reference area for the bioaccumulation component of the KPC NPDES monitoring program for Ward Cove (ENSR 1995a; EVS 1996)
- Moser Bay was used as the reference area for Ward Cove historical assessments of the sediment toxicity and benthic macroinvertebrate communities in Ward Cove (EVS 1992)
- All of the historical data on sediment chemistry, sediment toxicity, bioaccumulation, and benthic communities in Moser Bay (summarized in PTI [1996]) indicate that the bay is relatively uncontaminated by anthropogenic releases and is therefore appropriate for use as a reference area.

In addition to the general characteristics and historical data for Moser Bay described above, the data on sediment chemistry and sediment toxicity collected in 1996 and 1997 during the Phase 1 and Phase 2 investigations confirm the appropriateness of Moser Bay as a reference area for Ward Cove. As shown in Tables 7-1 and 7-2, the range of percent fine-grained sediment found in Moser Bay (53–91) bracketed the range found at most of the 61 stations sampled in Ward Cove in 1996 and 1997. In addition, Tables 7-1 through 7-3 document that concentrations of all CoPCs in Moser Bay were low. Finally, results for all four sediment toxicity tests indicate that little toxicity was found for sediments from Moser Bay (Tables 7-9 and 7-10). For example, values of survival for both amphipod tests (91–99 percent) were considerably greater than the performance criterion of 75 percent for valid reference areas (Ecology 1995), and values of percent normal survival for the echinoderm embryo test (73–86 percent) were greater than the performance criterion of 65 percent (Michelsen 1996). Although individual growth rate of *Neanthes* sp. at Station 29 was reduced relative to the value found at Station 30, it was greater than the performance criterion of 0.40 mg/day (Ecology 1995).

TABLE 7-9. SUMMARY OF SEDIMENT TOXICITY RESULTS FOR WARD COVE AND MOSER BAY IN 1996 AND COMPARISON WITH SEDIMENT QUALITY VALUES

Station	*Rhepoxynius abronius* Survival (percent)	*Leptocheirus plumulosus* Survival (percent)	*Neanthes* sp. Individual Growth Rate (mg/day)	*Dendraster excentricus* Normal Survival (percent)	*Dendraster excentricus* Embryo Normality (percent)
Ward Cove					
1	50 (32.2)**	93 (4.5)	0.59 (0.12)	51 (19.0)**	85 (11.1)*
2	7 (10.9)**	94 (4.2)	0.64 (0.08)	55 (10.1)**	93 (5.5)
3	90 (7.9)	93 (5.7)	0.54 (0.06)	51 (25.6)**	88 (11.9)*
4	64 (15.2)*	93 (6.7)	0.62 (0.11)	56 (19.5)**	87 (9.6)*
5	25 (19.0)**	98 (2.7)	0.57 (0.04)	48 (28.1)**	74 (26.6)*
6	5 (8.7)**	88 (6.7)	0.62 (0.11)	54 (21.4)**	92 (7.1)
7	90 (7.9)	99 (2.2)	0.61 (0.08)	61 (13.5)*	86 (12.4)*
8	43 (22.8)**	89 (13.9)	0.68 (0.16)	58 (13.9)**	89 (11.1)*
9	54 (17.8)**	92 (7.6)	0.63 (0.10)	43 (23.0)**	92 (6.8)[a]
10	75 (14.6)	96 (4.2)	0.67 (0.16)	50 (13.2)**	97 (1.7)
11	94 (8.2)	97 (4.5)	0.54 (0.11)	47 (23.7)**	95 (3.4)[a]
12	3 (2.7)**	93 (10.9)	0.63 (0.07)	46 (18.8)**	92 (2.0)
13	36 (10.8)**	95 (6.1)	0.56 (0.19)	52 (14.6)**	96 (3.2)
14	60 (20.9)**	98 (4.5)	0.70 (0.14)	64 (26.0)*	93 (6.6)
15	67 (13.5)*	94 (6.5)	0.66 (0.08)	67 (8.9)*	97 (1.8)
16	30 (15.4)**	98 (2.7)	0.68 (0.11)	52 (17.2)**	97 (1.8)
17	88 (11.5)	94 (6.5)	0.51 (0.10)	54 (30.4)**	95 (3.8)[a]
18	95 (5.0)	96 (4.2)	0.55 (0.07)	58 (13.4)**	94 (4.6)
19	48 (18.9)**	100 (--)	0.65 (0.06)	79 (15.0)	94 (5.8)
20	67 (16.4)*	97 (4.5)	0.59 (0.09)	72 (18.2)	96 (2.5)
21	82 (16.0)	96 (4.2)	0.63 (0.07)	80 (9.3)	98 (1.2)
22	84 (11.9)	92 (12.6)	0.57 (0.10)	80 (13.3)	94 (7.6)
23	94 (6.5)	94 (4.2)	0.64 (0.10)	59 (18.9)*	95 (5.3)
24	89 (8.2)	96 (6.5)	0.57 (0.07)	71 (16.4)*	89 (12.5)
25	3 (4.5)**	96 (5.5)	0.74 (0.09)	58 (24.2)**	94 (5.8)[a]
26	96 (4.2)	93 (4.5)	0.58 (0.10)	75 (9.2)	93 (4.4)
27	85 (6.1)	98 (2.7)	0.65 (0.10)	72 (23.2)	95 (3.2)[a]
28	69 (24.9)*	96 (5.5)	0.63 (0.10)	67 (8.6)*	94 (2.1)
Moser Bay					
29	91 (4.2)	97 (2.7)	0.48 (0.09)	83 (17.6)	97 (2.7)
30	93 (6.7)	99 (2.2)	0.72 (0.12)	86 (8.3)	97 (2.8)

Note: Mean values are presented with standard deviations in parentheses.

* - toxicity response is less than sediment quality standard (SQS) or, for *Dendraster excentricus* normality, response is significantly less ($P \leq 0.05$) than the pooled results for Moser Bay

** - toxicity response is less than minimum cleanup level (MCUL)

[a] Results are calculated from four replicate samples based on an outlier analysis discussed in the text.

TABLE 7-10. SUMMARY OF SEDIMENT TOXICITY RESULTS FOR WARD COVE AND MOSER BAY IN 1997 AND COMPARISON WITH SEDIMENT QUALITY VALUES

Station	*Rhepoxynius abronius* Survival (percent)	*Dendraster excentricus* Normal Survival (percent)	*Dendraster excentricus* Embryo Normality (percent)
Ward Cove			
2	9 (17.5)**	43 (20.6)**	91 (6.9)
3	65 (10.8)** [a]	53 (22.6)*	96 (0.8)
4	38 (28.4)**	56 (22.0)*	93 (4.9)
5	39 (22.5)**	53 (12.5)*	95 (3.3)
7	58 (15.7)**	59 (15.2)*	96 (3.8)
11	83 (7.6)	55 (12.8)*	96 (4.0)
12	14 (11.9)**	43 (14.4)**	94 (5.6)
13	15 (22.6)**	48 (5.4)**	97 (1.9)
16	89 (4.2)	32 (21.5)**	91 (9.5)
17	43 (39.9)**	57 (16.1)*	94 (4.0)
18	90 (7.1)	50 (23.1)**	97 (2.4)[a]
19	59 (12.9)**	61 (13.5)*	96 (1.9)
22	84 (13.4)	78 (14.0)	99 (1.1)
23	79 (18.8)	63 (22.6)	94 (4.7)
25	10 (14.1)**	56 (17.0)*	93 (2.4)
27	75 (17.3)[a]	38 (18.7)**	95 (3.2)[a]
28	73 (16.6)* [a]	58 (14.8)*	94 (6.9)
31	3 (4.5)**	28 (12.8)**	95 (4.5)
32	28 (32.5)**	54 (15.2)*	98 (2.4)
33	77 (11.0)	28 (11.9)**	95 (7.9)
34	39 (10.3)** [a]	50 (9.6)**	94 (5.2)
35	75 (17.0)	44 (9.5)**	97 (2.5)
37	65 (15.4)**	68 (17.0)	98 (2.5)
38	0 (0)**	50 (27.7)**	90 (9.5)
39	41 (11.1)** [a]	68 (14.1)	98 (1.7)
40	75 (5.8)[a]	76 (14.9)	97 (4.0)
41	90 (6.1)	41 (19.9)**	97 (3.7)
42	68 (16.8)*	57 (9.0)*	97 (1.8)
43	72 (15.3)*	59 (6.8)*	97 (4.3)
44	1 (2.2)**	52 (13.6)*	96 (1.7)
45	54 (37.0)**	48 (12.5)**	92 (7.2)
47	73 (16.1)*	49 (10.0)**	97 (3.5)
48	5 (7.1)**	56 (6.1)*	97 (2.6)
Moser Bay			
29	96 (2.2)	74 (11.4)	97 (2.1)
30	96 (4.2)	73 (16.9)	98 (1.1)

Note: Mean values are presented with standard deviations in parentheses.

* - toxicity response is less than sediment quality standard (SQS) or, for *Dendraster excentricus* normality, response is significantly less ($P \leq 0.05$) than the pooled results for Moser Bay

** - toxicity response is less than minimum cleanup level (MCUL)

[a] Results are calculated from four replicate samples based on an outlier analysis discussed in the text.

In summary, the general characteristics of Moser Bay and the results of evaluations of sediment chemistry, sediment toxicity, bioaccumulation, and benthic macroinvertebrate communities in Moser Bay during historical studies and/or the Phase 1 and Phase 2 investigations all support the appropriateness of Moser Bay as a reference area for Ward Cove.

7.1.2 Sediment Toxicity Evaluation

As discussed in Section 2, the four sediment toxicity tests used to characterize sediments in Ward Cove were as follows:

- The 10-day amphipod test using *Rhepoxynius abronius*
- The 10-day amphipod test using *Leptocheirus plumulosus*
- The 96-hour echinoderm embryo test using the sand dollar *Dendraster excentricus*
- The 20-day juvenile polychaete test using *Neanthes* sp.

The endpoint for the two amphipod tests was percent survival after the 10-day exposure period, and the endpoint for the juvenile polychaete test was growth (i.e., estimated by biomass) after the 20-day exposure period.

The primary endpoint for the echinoderm embryo test was percent normal survival following the 96-hour exposure period. However, an error component exists in the calculation of that endpoint because the expected density of embryos in each test chamber at the end of testing is *estimated* on the basis of the mean density found in the replicated seawater controls (U.S. EPA 1995). A second endpoint for this test is percent normality of surviving embryos. Because this determination is based on *known* numbers of organisms, it is not affected by the same error component as the endpoint based on percent normal survival. For the present study, the normality endpoint was used in Section 8 to corroborate the endpoint based on percent normal survival for stations at which the latter endpoint was the only environmental indicator (chemical or biological) that identified a potential problem at a station. Additional information on the performance of the echinoderm test and interpretation of test results is provided in U.S. EPA (1999a).

The remainder of this section presents the detailed results of the sediment toxicity evaluation of Ward Cove sediments.

7.1.2.1 Results of Sediment Toxicity Tests

Results of the sediment toxicity tests used to evaluate sediments in Ward Cove in 1996 and 1997 are presented in Tables 7-9 and 7-10 and Figure 7-1. Because the standard deviations for the amphipod test based on *Rhepoxynius abronius* in 1997 were higher in

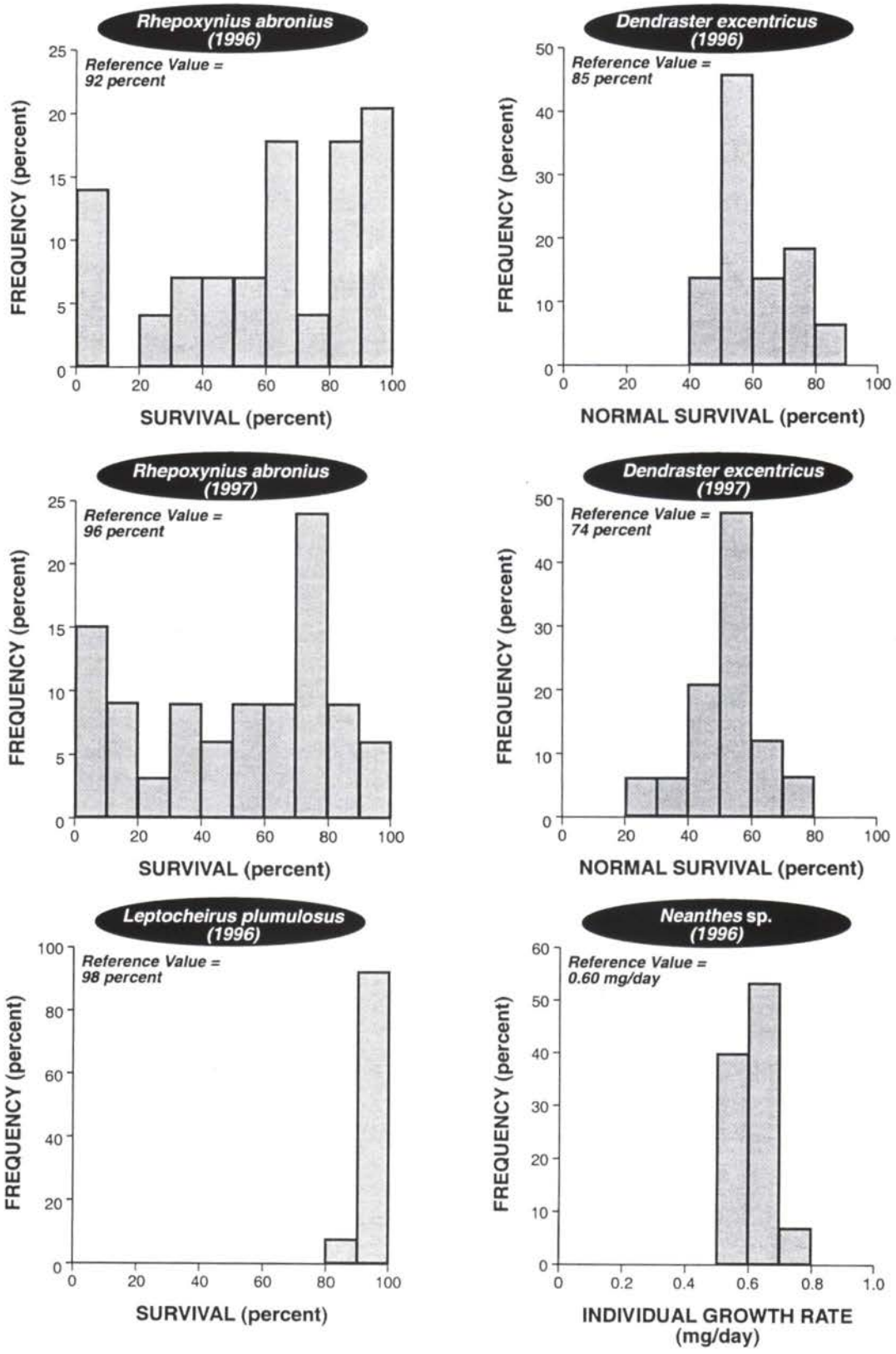


Figure 7-1. Distribution of results for the toxicity tests used in 1996 and 1997.

some cases than the values found in 1996, the data on individual replicates for 1997 were inspected. It was found that in some cases, survival in a single unrepresentative (or outlier) replicate was much lower than survival in the remaining four replicates. For such cases, an outlier analysis recommended by U.S. EPA (1994e) was conducted using the *t*-test for comparing a single value (i.e., the outlier) with the mean of a sample (i.e., the mean of the remaining four replicates). Results from six stations (i.e., Stations 3, 27, 28, 34, 39, and 40) were subjected to outlier analysis, and significant differences ($P \leq 0.05$) were found between the outlier replicate and the mean of the remaining replicates in all cases. The toxicity responses for those six stations were therefore based on the four remaining replicates.

Unusually high standard deviations (i.e., >15 percent) were also found for some results of the normality endpoint of the echinoderm embryo tests in 1996 and 1997. Results from the affected stations (i.e., Stations 5, 9, 11, 17, 25, and 27 in 1996; Stations 18 and 27 in 1997) were therefore subjected to the same kind of outlier analysis described above for the amphipod test. Significant differences ($P \leq 0.05$) were found between the outlier replicate and the mean of the remaining four replicates for seven of the eight stations (i.e., all except Station 5 in 1996). The toxicity responses for five stations were therefore based on the four remaining replicates.

The results for each kind of sediment toxicity test conducted in Ward Cove in 1996 and 1997 are described below.

Amphipod Test Based on *Rhepoxynius abronius*—Survival in Ward Cove ranged from 3 percent at Stations 12 and 25 to 96 percent at Station 26 in 1996 (Table 7-9) and from 0 percent at Station 38 to 90 percent at Stations 18 and 41 in 1997 (Table 7-10). In both years, amphipod survival exhibited a broad distribution across the response range (Figure 7-1). Mean survival for all stations in the Cove was 61 percent in 1996 and 51 percent in 1997. Survival in Moser Bay ranged from 91 to 96 percent, with a mean value of 92 percent in 1996 and 96 percent in 1997.

Amphipod Test Based on *Leptocheirus plumulosus*—In 1996, survival in Ward Cove ranged from 88 percent at Station 6 to 100 percent at Station 19 (Table 7-9) and exhibited a narrow distribution with all but 2 of the 28 values falling between 90 and 100 percent (Figure 7-1). Mean survival for all stations in the Cove was 95 percent. Survival in Moser Bay ranged from 97 to 99 percent, with a mean value of 98 percent.

Juvenile Polychaete Test Based on *Neanthes* sp.—In 1996, individual growth rate in Ward Cove ranged from 0.51 mg/day at Station 17 to 0.74 mg/day at Station 25 (Table 7-9) and exhibited a narrow distribution with all but 2 of the 28 values falling between 0.50 and 0.70 mg/day (Figure 7-1). Mean growth rate for all stations in

the Cove was 0.62 mg/day. Individual growth rate in Moser Bay ranged from 0.48 to 0.72 mg/day, with a mean value of 0.60 mg/day.

Echinoderm Embryo Test Based on *Dendraster excentricus*—Normal survival in Ward Cove ranged from 43 percent at Station 9 to 80 percent at Stations 21 and 22 in 1996 (Table 7-9) and from 28 percent at Stations 31 and 33 to 78 percent at Station 22 in 1997 (Table 7-10). In both years, normal survival exhibited a moderately broad distribution over the response range (Figure 7-1). Mean normal survival for all stations in the Cove was 60 percent in 1996 and 52 percent in 1997. Normal survival in Moser Bay ranged from 73 to 86 percent, with a mean value of 85 percent in 1996 and 74 percent in 1997.

Echinoderm embryo normality in Ward Cove ranged from 74 percent at Station 5 to 98 percent at Station 21 in 1996 (Table 7-9) and from 90 percent at Station 38 to 99 percent at Station 22 in 1997 (Table 7-10). Mean normality for all stations in the Cove was 92 percent in 1996 and 95 percent in 1997. Normality in Moser Bay ranged from 97 to 98 percent, with a mean value of 97 percent in 1996 and 98 percent in 1997.

The results described above indicate that of the four toxicity tests used to evaluate sediments from Ward Cove, only the amphipod test based on *Rhepoxynius abronius* and the echinoderm embryo test (i.e., based on normal survival) exhibited responses that were substantially adverse compared to the responses found for Moser Bay. By contrast, the responses exhibited by the amphipod test using *Leptocheirus plumulosus*, the juvenile polychaete test, and the embryo normality endpoint for the echinoderm test generally were similar to the responses found for Moser Bay. In most cases, the responses based on the normality endpoint for the echinoderm embryo test were also similar to the responses found for Moser Bay.

7.1.2.2 Significance of Toxicity Test Results

The significance of all of the toxicity test results except those for the echinoderm embryo normality endpoint was determined using the methods described for the Washington State sediment management standards (Ecology 1995) and included more recent modifications specified by Washington State regulatory agencies (Ecology et al. 1995; Michelsen 1996; Michelsen and Shaw 1996). The methods identify two degrees of progressively adverse effects. The lower degree of adverse effects determines compliance with the SQSs and is used to evaluate whether sediments may be toxic and therefore warrant further investigation. The higher degree of adverse effects determines compliance with the minimum cleanup levels (MCULs) and is used in cleanup evaluations. Because the Washington State sediment management standards do not address echinoderm embryo normality, neither an SQS nor an MCUL could be developed for that endpoint. Instead, the endpoint was tested only for statistical significance ($P \leq 0.05$) using the same statistical procedures specified by the sediment management standards for the other endpoints. In addition, to ensure that the test response was sufficiently adverse, a screening

value of 90 percent was used. The screening value is the minimum allowable normality for acceptable negative controls (Fox and Littleton 1994).

Compliance with the SQS and MCUL values is typically determined by statistically comparing the mean toxicity response at each test station with the mean response in a reference area using Student's *t*-test and a one-tailed pairwise significance level of $P \leq 0.05$. However, because of the relatively high degree of variability often found among replicate samples for the echinoderm embryo test, significance for this test is determined at $P \leq 0.10$ to provide increased statistical power to discriminate adverse effects (Michelsen 1996).

In addition to being significantly different from the reference response based on statistical criteria, the test response must be sufficiently adverse, as determined by the following screening values:

- **Amphipod Test**
  - **SQS:** Mean survival must be less than 75 percent (i.e., if mean survival is greater than 75 percent or mean mortality is less than 25 percent, the SQS cannot be exceeded regardless of the results of the statistical comparisons)
  - **MCUL:** Mean survival must be less than the mean reference value minus 30 percent (i.e., if mean survival is greater than 70 percent of the reference value or mean mortality is less than 30 percent of the reference value, the MCUL cannot be exceeded regardless of the results of the statistical comparisons)
- **Juvenile Polychaete Test**
  - **SQS:** Mean individual growth rate must be less than 70 percent of the mean reference value (i.e., if mean growth rate is greater than 70 percent of the mean reference value, the SQS cannot be exceeded regardless of the results of the statistical comparisons)
  - **MCUL:** Mean individual growth rate must be less than 50 percent of the mean reference value (i.e., if mean growth rate is greater than 50 percent of the mean reference value, the MCUL cannot be exceeded regardless of the results of the statistical comparisons)
- **Echinoderm Embryo Test (i.e., based on normal survival)**
  - **SQS:** Mean normal survival must be less than 85 percent of the mean reference value (i.e., if mean normal survival is greater than 85 percent of the mean reference value, the SQS

cannot be exceeded regardless of the results of the statistical comparisons), based on data normalized to the seawater control

- **MCUL:** Mean normal survival must be less than 70 percent of the mean reference value (i.e., if mean normal survival is greater than 70 percent of the mean reference value, the MCUL cannot be exceeded regardless of the results of the statistical comparisons), based on data normalized to the seawater control.

Using the above criteria, the SQS and MCUL screening values for the present study are as follows:

- **Amphipod Test**
  - **SQS:** 75 percent (for both amphipod tests)
  - **MCUL:** 62 percent (*Rhepoxynius abronius* in 1996), 66 percent (*Rhepoxynius abronius* in 1997), 69 percent (*Leptocheirus plumulosus*)
- **Juvenile Polychaete Test**
  - **SQS:** 0.42 mg/day
  - **MCUL:** 0.30 mg/day
- **Echinoderm Embryo Test**
  - **SQS:** 72 percent (in 1996), 63 percent (in 1997)
  - **MCUL:** 59 percent (in 1996), 52 percent (in 1997).

In developing the reference conditions for comparisons with toxicity results for Ward Cove, the data from both stations sampled in Moser Bay were pooled (separately for 1996 and 1997) and the mean and standard deviation of the pooled data set were used as estimates of reference conditions for each year. Data from both stations were used in this analysis because the two stations bracketed the range of sediment grain size distributions found throughout most of Ward Cove. For example, percent fine-grained sediment at the 44 stations (61 values) sampled in Ward Cove in 1996 and 1997 ranged from 6 to 81 percent, with all but 15 of the values ranging from 50 to 81 percent (Tables 7-1 and 7-2). Because percent fine-grained sediment at the two reference stations ranged from 53 to 91 percent, they bracketed the range of grain size distributions found throughout most of Ward Cove. The use of multiple reference stations also increased the sample size for reference conditions from 5 replicate samples (based on a single station) to 10 replicate samples (based on both stations). This increased sample size increased the statistical power of the comparisons of toxicity responses between Ward Cove stations and the reference conditions.

Decision trees for evaluating adverse effects for the four toxicity tests used in Ward Cove in 1996 are presented in Figures 7-2 through 7-5. They are based on similar figures presented by Gries and Waldow (1995) but were modified for site-specific use in Ward Cove using the data collected in the Phase 1 and Phase 2 investigations. These evaluations have the following major steps:

- Evaluate performance of the negative control sample
- Evaluate performance of the reference area (i.e., Moser Bay) sample
- Apply the screening value for SQS exceedance to each Ward Cove sample
- Conduct statistical comparisons with Moser Bay samples for each Ward Cove sample that does not pass the SQS screening evaluation
- If significant toxicity is found for a Ward Cove sample, apply the screening value for MCUL exceedance
- If significant toxicity is not found for a Ward Cove sample, evaluate data variability (i.e., standard deviation) to ensure there is adequate statistical power to discriminate an effect
- If data variability is too high, evaluate statistical power
- If statistical power is too low, determine whether outlier values can be removed from the analysis or evaluate the results qualitatively.

Summaries of the evaluations used to statistically compare results from Ward Cove and Moser Bay for the amphipod test based on *Rhepoxynius abronius* and the echinoderm embryo test are presented in Tables 7-11 through 7-14. Similar tables are not presented for the amphipod test based on *Leptocheirus plumulosus* and *Neanthes* sp. because the results for all samples exceeded the SQS screening values, obviating the need for statistical comparisons. For *Neanthes* sp., the screening value was exceeded by all samples regardless of whether it was based on the mean growth rate at both Moser Bay stations (screening value = 0.42 mg/day) or, more conservatively, only on the greater of the two growth rates (screening value = 0.50 mg/day).

The statistical comparisons summarized in Tables 7-11 through 7-16 for each station in Ward Cove and the reference conditions in Moser Bay were conducted as follows:

- The data were tested for normality (using the Wilk-Shapiro test) and homogeneous variances (using the *F*-test for variances)
- If the data passed both the normality and variance tests, significance was determined using Student's *t*-test and a one-tailed hypothesis (significance levels of $P \leq 0.05$ and $P \leq 0.10$ were used for the amphipod and echinoderm tests, respectively)

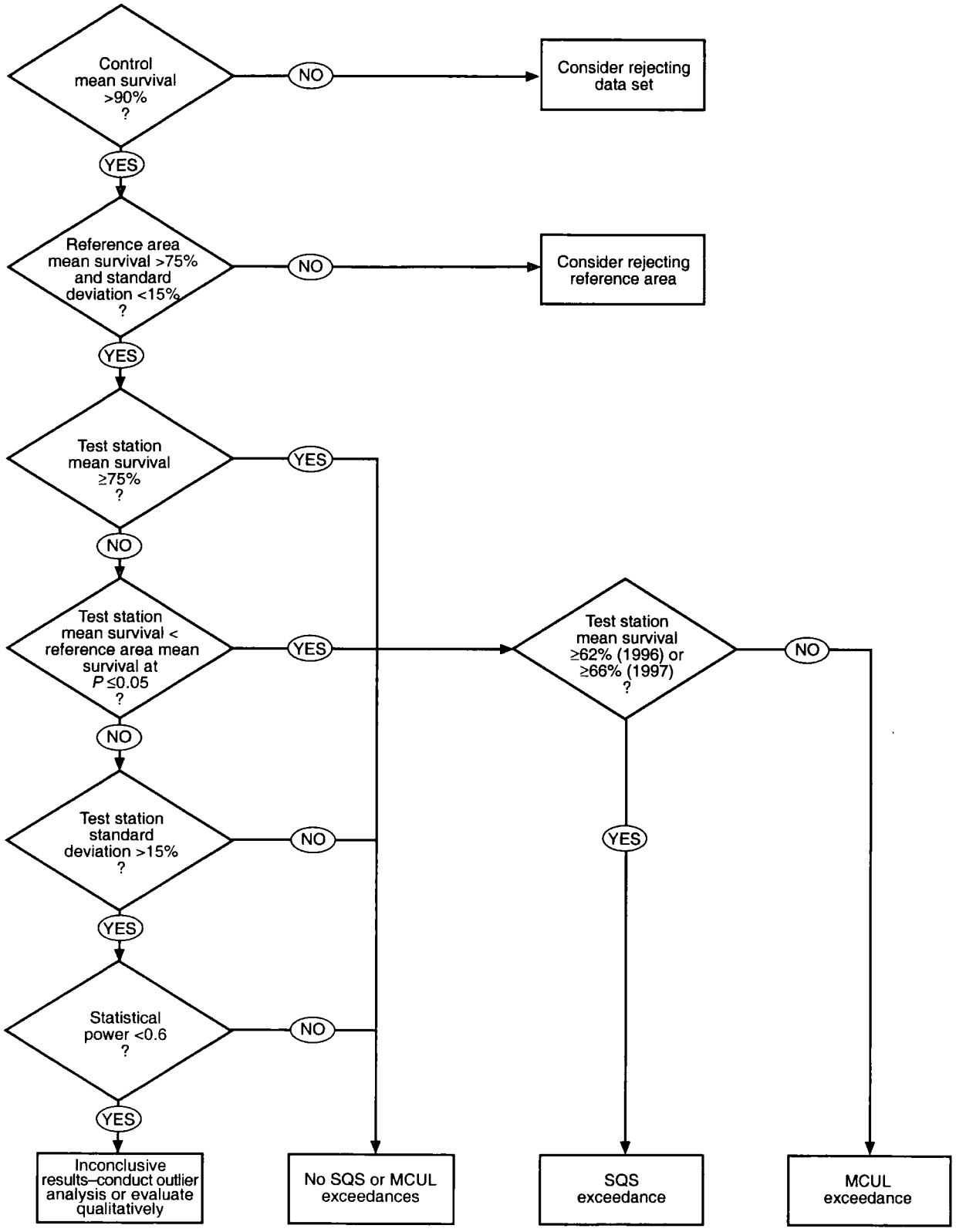


Figure 7-2. Decision tree for evaluating SQS and MCUL exceedances based on the amphipod test using *Rhepoxynius abronius*.

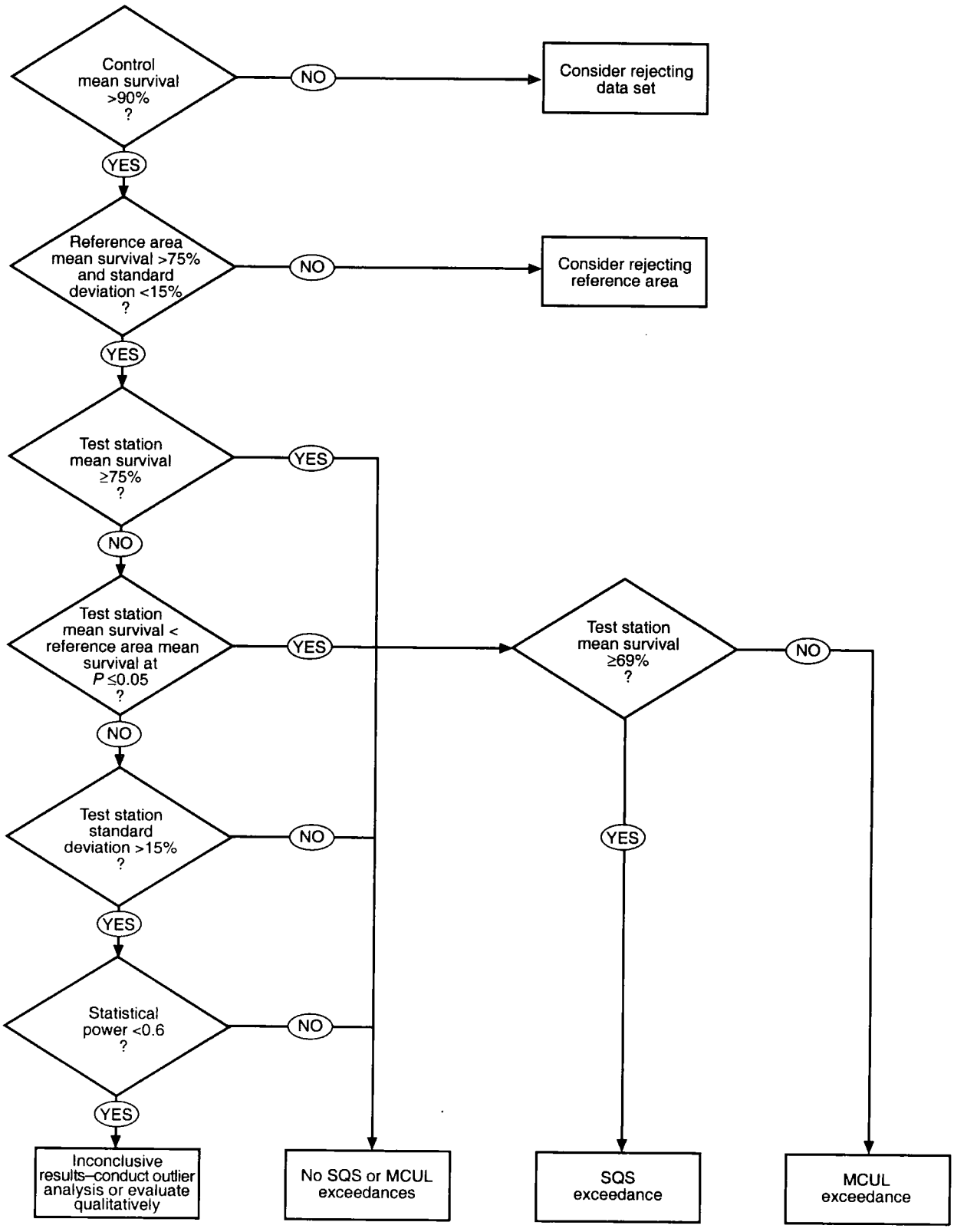


Figure 7-3. Decision tree for evaluating SQS and MCUL exceedances based on the amphipod test using *Leptocheirus plumulosus*.

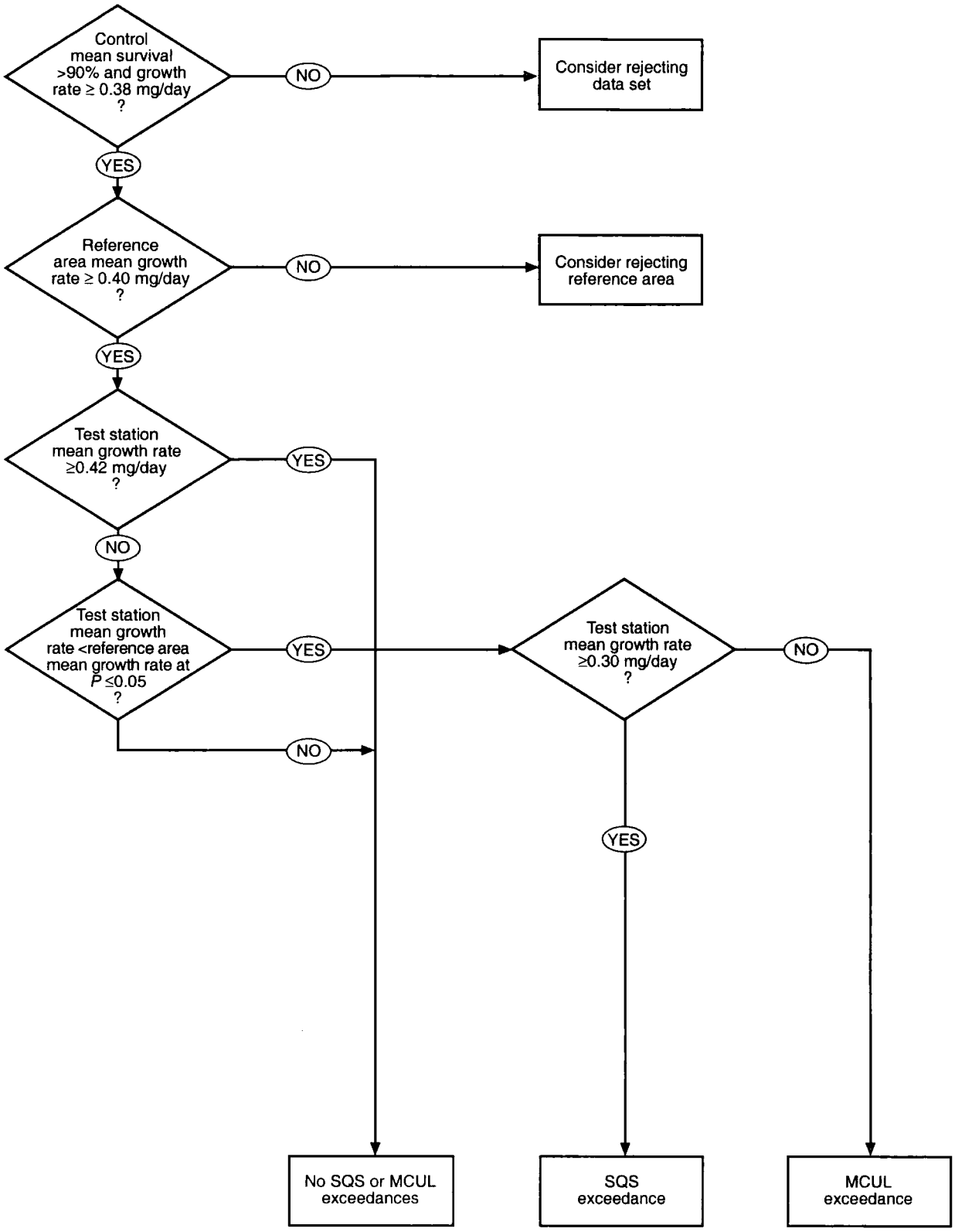


Figure 7-4. Decision tree for evaluating SQS and MCUL exceedances based on the juvenile polychaete test using *Neanthes* sp.

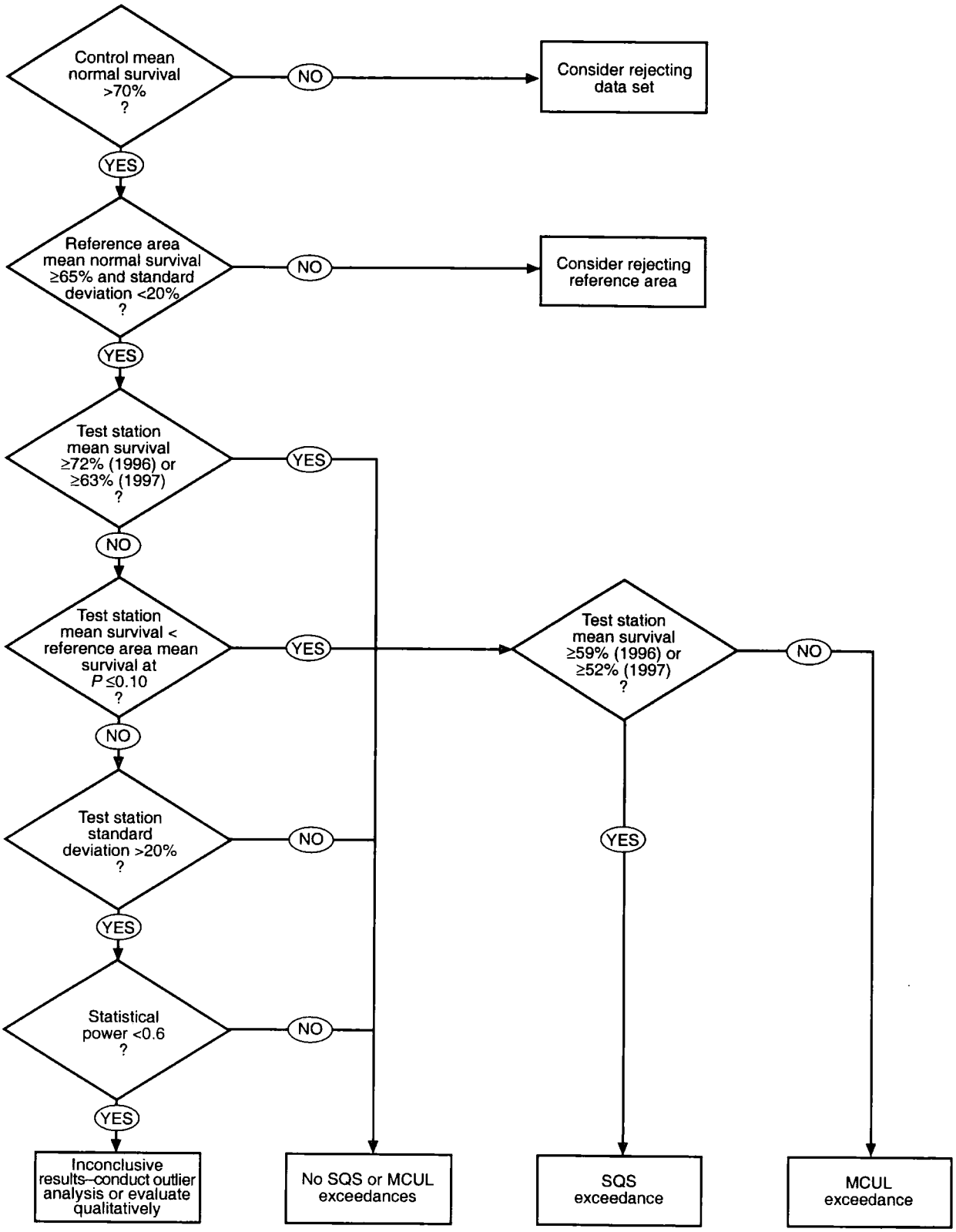


Figure 7-5. Decision tree for evaluating SQS and MCUL exceedances based on the echinoderm embryo test using *Dendraster excentricus.*

TABLE 7-11. SUMMARY OF EVALUATIONS OF *Rhepoxynius abronius* SURVIVAL FOR WARD COVE IN 1996

Station	Mean Survival (percent)	Standard Deviation (percent)	Normality[a]	Homog. Variance[b]	Arcsine Transform	Normality[a]	Homog. Variance[b]	Mann-Whitney *U*-Test	Signif.	High Variance[c]	Exceedance[d] SQS	Exceedance[d] MCUL
1	50	32.2	*	*	yes	ns	ns	--	*	--	yes	yes
2	7	11.0	*	*	yes	*	ns	yes	*	--	yes	yes
3	90 [e]	7.9	--	--	--	--	--	--	--	--	no	no
4	64	15.2	*	*	yes	ns	ns	--	*	--	yes	no
5	25	19.0	*	*	yes	ns	ns	--	*	--	yes	yes
6	5	8.7	*	ns	yes	*	ns	yes	*	--	yes	yes
7	90 [e]	7.9	--	--	--	--	--	--	--	--	no	no
8	43	22.8	*	*	yes	ns	ns	--	*	--	yes	yes
9	54	17.8	*	*	yes	ns	ns	--	*	--	yes	yes
10	75 [e]	14.6	--	--	--	--	--	--	--	--	no	no
11	94 [e]	8.2	--	--	--	--	--	--	--	--	no	no
12	3	2.7	*	ns	yes	*	*	yes	*	--	yes	yes
13	36	10.8	*	*	yes	ns	ns	--	*	--	yes	yes
14	60	20.9	*	*	yes	ns	ns	--	*	--	yes	yes
15	67	13.5	*	*	yes	ns	ns	--	*	--	yes	yes
16	30	15.4	*	*	yes	ns	ns	--	*	--	yes	no
17	88 [e]	11.5	--	--	--	--	--	--	--	--	yes	yes
18	95 [e]	5.0	--	--	--	--	--	--	--	--	no	no
19	48	18.9	*	*	yes	ns	ns	--	*	--	no	no
20	67	16.4	*	*	yes	ns	ns	--	*	--	yes	yes
21	82 [e]	16.0	--	--	--	--	--	--	--	--	yes	no
22	84 [e]	11.9	--	--	--	--	--	--	--	--	no	no
23	94 [e]	6.5	--	--	--	--	--	--	--	--	no	no
24	89 [e]	8.2	--	--	--	--	--	--	--	--	no	no
25	3	4.5	*	ns	yes	*	*	yes	*	--	no	no
26	96 [e]	4.2	--	--	--	--	--	--	--	--	yes	yes
27	85 [e]	6.1	--	--	--	--	--	--	--	--	no	no
28	69	24.9	*	*	yes	ns	ns	--	*	--	no	no
											yes	no

Note: MCUL - minimum cleanup level
ns - $P > 0.05$
SQS - sediment quality standard
* - $P \leq 0.05$
-- - determination not necessary due to results of a previous determination

[a] Normality was evaluated using the Wilk-Shapiro test.

[b] Homogeneous variance was evaluated using the *F*-test for variances.

[c] High variances are those >15 percent for test results not significantly different ($P > 0.05$) from reference conditions.

[d] Only the SQS was exceeded if a significant ($P \leq 0.05$) test result was <75 percent but ≥62 percent. Both the SQS and MCUL were exceeded if a significant test result was <62 percent.

[e] Because mean survival was ≥ 75 percent, the SQS was not exceeded and statistical testing was not necessary.

TABLE 7-12. SUMMARY OF EVALUATIONS OF *Rhepoxynius abronius* SURVIVAL FOR WARD COVE IN 1997

Station	Mean Survival (percent)	Standard Deviation (percent)	Normality[a]	Homog. Variance[b]	Arcsine Transform	Normality[a]	Homog. Variance[b]	Mann-Whitney *U*-Test	Signif.	High Variance[c]	Exceedance[d] SQS	Exceedance[d] MCUL
2	9	17.5	*	*	yes	*	*	yes	*	--	yes	yes
3	65	10.8	*	*	yes	ns	ns	--	*	--	yes	yes
4	38	28.4	*	*	yes	*	*	yes	*	--	yes	yes
5	39	22.5	*	*	yes	*	*	yes	*	--	yes	yes
7	58	15.7	*	*	yes	*	ns	yes	*	--	yes	yes
11	83 [e]	7.6	--	--	--	--	--	--	--	--	no	no
12	14	11.9	*	*	yes	*	ns	yes	*	--	yes	yes
13	15	22.6	*	*	yes	*	*	yes	*	--	yes	yes
16	89 [e]	4.2	--	--	--	--	--	--	--	--	no	no
17	43	39.9	*	*	yes	*	*	yes	*	--	yes	yes
18	90 [e]	7.1	--	--	--	--	--	--	--	--	no	no
19	59	12.9	*	*	yes	ns	ns	--	*	--	yes	yes
22	84 [e]	13.4	--	--	--	--	--	--	--	--	no	no
23	79 [e]	18.8	--	--	--	--	--	--	--	--	no	no
25	10	14.1	*	*	yes	*	*	yes	*	--	yes	yes
27	75 [e]	17.3	--	--	--	--	--	--	--	--	no	no
28	73	16.6	*	*	yes	*	ns	yes	*	--	yes	no
31	3	4.5	*	ns	yes	*	ns	yes	*	--	yes	yes
32	28	32.5	*	*	yes	*	*	yes	*	--	yes	yes
33	77 [e]	11.0	--	--	--	--	--	--	--	--	no	no
34	39	10.3	*	*	yes	*	ns	yes	*	--	yes	yes
35	75 [e]	17.0	--	--	--	--	--	--	--	--	no	no
37	65	15.4	*	*	yes	*	ns	yes	*	--	yes	yes
38	0	0	*	*	yes	*	*	yes	*	--	yes	yes
39	41	11.1	*	*	yes	*	ns	yes	*	--	yes	yes
40	75 [e]	5.8	--	--	--	--	--	--	--	--	no	no
41	90 [e]	6.1	--	--	--	--	--	--	--	--	no	no
42	68	16.8	*	*	yes	ns	ns	--	*	--	yes	no
43	72	15.3	*	*	yes	ns	ns	--	*	--	yes	no
44	1	2.2	*	ns	yes	*	ns	yes	*	--	yes	yes
45	54	37.0	*	*	yes	*	*	yes	*	--	yes	yes
47	73	16.1	*	*	yes	*	ns	yes	*	--	yes	no
48	5	7.1	*	*	yes	*	ns	yes	*	--	yes	yes

Note: MCUL - minimum cleanup level
ns - $P > 0.05$
SQS - sediment quality standard
* - $P \leq 0.05$
-- - determination not necessary due to results of a previous determination

[a] Normality was evaluated using the Wilk-Shapiro test.

[b] Homogeneous variance was evaluated using the *F*-test for variances.

[c] High variances are those >15 percent for test results not significantly different ($P > 0.05$) from reference conditions.

[d] Only the SQS was exceeded if a significant ($P \leq 0.05$) test result was <75 percent but ≥66 percent. Both the SQS and MCUL were exceeded if a significant test result was <66 percent.

[e] Because mean survival was ≥ 75 percent, the SQS was not exceeded and statistical testing was not necessary.

TABLE 7-13. SUMMARY OF EVALUATIONS OF *Dendraster excentricus* NORMAL SURVIVAL FOR WARD COVE IN 1996

Station	Mean Normal Survival (percent)	Standard Deviation (percent)	Normality[a]	Homog. Variance[b]	Arcsine Transform	Normality[a]	Homog. Variance[b]	Mann-Whitney *U*-Test	Signif.	High Variance[c]	Exceedance[d] SQS	Exceedance[d] MCUL
1	51	18.9	ns	ns	--	--	--	--	*	--	yes	yes
2	55	10.1	ns	ns	--	--	--	--	*	--	yes	yes
3	51	25.6	ns	*	yes	ns	ns	--	*	--	yes	yes
4	56	19.5	ns	ns	--	--	--	--	*	--	yes	yes
5	48	28.1	*	*	yes	ns	ns	--	*	--	yes	yes
6	54	21.4	ns	ns	--	--	--	--	*	--	yes	yes
7	61	13.5	ns	ns	--	--	--	--	*	--	yes	no
8	58	13.9	ns	ns	--	--	--	--	*	--	yes	yes
9	43	23.0	ns	ns	--	--	--	--	*	--	yes	yes
10	50	13.2	ns	ns	--	--	--	--	*	--	yes	yes
11	47	23.7	ns	ns	--	--	--	--	*	--	yes	yes
12	46	18.9	ns	ns	--	--	--	--	*	--	yes	yes
13	52	14.6	ns	ns	--	--	--	--	*	--	yes	yes
14	64	26.0	ns	*	yes	ns	ns	--	--	--	yes	no
15	67	8.9	ns	ns	--	--	--	--	*	--	yes	no
16	52	17.1	ns	ns	--	--	--	--	*	--	yes	yes
17	54	30.4	*	*	yes	ns	ns	--	*	--	yes	yes
18	58	13.4	ns	ns	--	--	--	--	*	--	yes	yes
19	79[e]	15.0	--	--	--	--	--	--	--	--	no	no
20	72[e]	18.2	--	--	--	--	--	--	--	--	no	no
21	80[e]	9.3	--	--	--	--	--	--	--	--	no	no
22	80[e]	13.3	--	--	--	--	--	--	--	--	no	no
23	59	18.9	ns	ns	--	--	--	--	*	--	yes	no
24	71	16.4	ns	ns	--	--	--	--	*	--	yes	no
25	58	24.2	*	ns	yes	ns	ns	--	*	--	yes	yes
26	75[e]	9.2	--	--	--	--	--	--	--	--	no	no
27	72[e]	23.2	--	--	--	--	--	--	--	--	no	no
28	67	8.6	ns	ns	--	--	--	--	*	--	yes	no

Note: MCUL - minimum cleanup level
ns - $P > 0.05$
SQS - sediment quality standard
* - $P \leq 0.05$
-- - determination not necessary due to results of a previous determination

[a] Normality was evaluated using the Wilk-Shapiro test.

[b] Homogeneous variance was evaluated using the *F*-test for variances.

[c] High variances are those >20 percent for test results not significantly different ($P > 0.10$) from reference conditions.

[d] Only the SQS was exceeded if a significant ($P \leq 0.10$) test result was <72 percent but ≥59 percent. Both the SQS and MCUL were exceeded if a significant test result was <59 percent.

[e] Because mean survival was ≥ 72 percent, the SQS was not exceeded and statistical testing was not necessary.

TABLE 7-14. SUMMARY OF EVALUATIONS OF *Dendraster excentricus* NORMAL SURVIVAL FOR WARD COVE IN 1997

Station	Mean Normal Survival (percent)	Standard Deviation (percent)	Normality[a]	Homog. Variance[b]	Arcsine Transform	Normality[a]	Homog. Variance[b]	Mann-Whitney *U*-Test	Signif.	High Variance[c]	Exceedance[d] SQS	Exceedance[d] MCUL
2	43	20.6	ns	*	yes	ns	ns	--	*	--	yes	yes
3	53	22.6	ns	ns	--	--	--	--	*	--	yes	no
4	56	22.0	ns	ns	--	--	--	--	*	--	yes	no
5	53	12.5	ns	ns	--	--	--	--	*	--	yes	no
7	59	15.2	ns	ns	--	--	--	--	*	--	yes	no
11	55	12.8	ns	ns	--	--	--	--	*	--	yes	no
12	43	14.4	ns	ns	--	--	--	--	*	--	yes	yes
13	48	5.4	ns	*	yes	ns	*	yes	*	--	yes	yes
16	32	21.5	ns	ns	--	--	--	--	*	--	yes	yes
17	57	16.1	ns	ns	--	--	--	--	*	--	yes	no
18	50	23.1	*	ns	yes	*	ns	yes	*	--	yes	yes
19	61	13.5	ns	ns	--	--	--	--	*	--	yes	no
22	78 [e]	14.0	--	--	--	--	--	--	--	--	no	no
23	63 [e]	22.6	--	--	--	--	--	--	--	--	no	no
25	56	17.0	ns	ns	--	--	--	--	*	--	yes	no
27	38	18.7	ns	ns	--	--	--	--	*	--	yes	yes
28	58	14.8	ns	ns	--	--	--	--	*	--	yes	no
31	28	12.8	ns	ns	--	--	--	--	*	--	yes	yes
32	54	15.2	ns	ns	--	--	--	--	*	--	yes	no
33	28	11.9	ns	ns	--	--	--	--	*	--	yes	yes
34	50	9.6	ns	ns	--	--	--	--	*	--	yes	yes
35	44	9.5	ns	ns	--	--	--	--	*	--	yes	yes
37	68 [e]	17.0	--	--	--	--	--	--	--	--	no	no
38	50	27.7	*	*	yes	ns	ns	--	*	--	yes	yes
39	68 [e]	14.1	--	--	--	--	--	--	--	--	no	no
40	76 [e]	14.9	--	--	--	--	--	--	--	--	no	no
41	41	19.9	*	*	yes	*	ns	yes	*	--	yes	yes
42	57	9.0	ns	ns	--	--	--	--	*	--	yes	no
43	59	6.8	ns	ns	--	--	--	--	*	--	yes	no
44	52	13.6	ns	ns	--	--	--	--	*	--	yes	no
45	48	12.5	ns	ns	--	--	--	--	*	--	yes	yes
47	49	10.0	ns	ns	--	--	--	--	*	--	yes	yes
48	56	6.1	ns	ns	--	--	--	--	*	--	yes	no

Note: MCUL - minimum cleanup level
ns - $P > 0.05$
SQS - sediment quality standard
* - $P \leq 0.05$
-- - determination not necessary due to results of a previous determination

[a] Normality was evaluated using the Wilk-Shapiro test.

[b] Homogeneous variance was evaluated using the *F*-test for variances.

[c] High variances are those >20 percent for test results not significantly different ($P > 0.10$) from reference conditions.

[d] Only the SQS was exceeded if a significant ($P \leq 0.10$) test result was <63 percent but ≥52 percent. Both the SQS and MCUL were exceeded if a significant test result was <52 percent.

[e] Because mean survival was ≥ 63 percent, the SQS was not exceeded and statistical testing was not necessary.

TABLE 7-15. SUMMARY OF EVALUATIONS OF *Dendraster excentricus* NORMALITY FOR WARD COVE IN 1996

Station	Mean Normality (percent)	Standard Deviation (percent)	Normality[a]	Homog. Variance[b]	Arcsine Transform	Normality[a]	Homog. Variance[b]	Mann-Whitney *U*-Test	Significant Effect?
1	85	11.1	*	*	yes	ns	ns	--	*
2	93 [c]	5.5	--	--	--	--	--	--	--
3	88	11.9	*	*	yes	*	*	yes	*
4	87	9.6	*	*	yes	ns	ns	--	*
5	74	26.6	*	*	yes	*	*	yes	*
6	92 [c]	7.1	--	--	--	--	--	--	--
7	86	12.4	*	*	yes	*	*	yes	*
8	89	11.1	*	*	yes	*	*	yes	*
9	92 [c]	6.8	--	--	--	--	--	--	--
10	97 [c]	1.7	--	--	--	--	--	--	--
11	95 [c]	3.4	--	--	--	--	--	--	--
12	92 [c]	2.0	--	--	--	--	--	--	--
13	96 [c]	3.2	--	--	--	--	--	--	--
14	93 [c]	6.6	--	--	--	--	--	--	--
15	97 [c]	1.8	--	--	--	--	--	--	--
16	97 [c]	1.8	--	--	--	--	--	--	--
17	95 [c]	3.8	--	--	--	--	--	--	--
18	94 [c]	4.6	--	--	--	--	--	--	--
19	94 [c]	5.8	--	--	--	--	--	--	--
20	96 [c]	2.5	--	--	--	--	--	--	--
21	98 [c]	1.2	--	--	--	--	--	--	--
22	94 [c]	7.6	--	--	--	--	--	--	--
23	95 [c]	5.3	--	--	--	--	--	--	--
24	89	12.6	*	*	yes	*	*	yes	--
25	94 [c]	5.8	--	--	--	--	--	--	--
26	93 [c]	4.4	--	--	--	--	--	--	--
27	95 [c]	3.2	--	--	--	--	--	--	--
28	94 [c]	2.1	--	--	--	--	--	--	--

Note: ns - $P > 0.05$, comparisonwise

* - $P \leq 0.05$, comparisonwise

-- - determination not necessary due to results of a previous determination

[a] Normality was evaluated using the Wilk-Shapiro test.

[b] Homogeneous variance was evaluated using the *F*-test for variances.

[c] Significance was not determined because value was ≥90 percent (i.e., the minimum allowable value for negative controls).

TABLE 7-16. SUMMARY OF EVALUATIONS OF *Dendraster excentricus* NORMALITY FOR WARD COVE IN 1997

Station	Mean Normality[a] (percent)	Standard Deviation (percent)	Normality[b]	Homog. Variance[c]	Arcsine Transform	Normality[b]	Homog. Variance[c]	Mann-Whitney *U*-Test	Significant Effect?
2	91	6.9	--	--	--	--	--	--	--
3	96	0.8	--	--	--	--	--	--	--
4	93	4.9	--	--	--	--	--	--	--
5	95	3.3	--	--	--	--	--	--	--
7	96	3.8	--	--	--	--	--	--	--
11	96	4.0	--	--	--	--	--	--	--
12	94	5.6	--	--	--	--	--	--	--
13	97	1.9	--	--	--	--	--	--	--
16	91	9.5	--	--	--	--	--	--	--
17	94	4.0	--	--	--	--	--	--	--
18	97	2.4	--	--	--	--	--	--	--
19	96	1.9	--	--	--	--	--	--	--
22	99	1.1	--	--	--	--	--	--	--
23	94	4.7	--	--	--	--	--	--	--
25	93	2.4	--	--	--	--	--	--	--
27	95	3.2	--	--	--	--	--	--	--
28	94	6.9	--	--	--	--	--	--	--
31	95	4.5	--	--	--	--	--	--	--
32	98	2.4	--	--	--	--	--	--	--
33	95	7.9	--	--	--	--	--	--	--
34	94	5.2	--	--	--	--	--	--	--
35	97	2.5	--	--	--	--	--	--	--
37	98	2.5	--	--	--	--	--	--	--
38	90	9.5	--	--	--	--	--	--	--
39	98	1.7	--	--	--	--	--	--	--
40	97	4.0	--	--	--	--	--	--	--
41	97	3.7	--	--	--	--	--	--	--
42	97	1.8	--	--	--	--	--	--	--
43	97	4.3	--	--	--	--	--	--	--
44	96	1.7	--	--	--	--	--	--	--
45	92	7.2	--	--	--	--	--	--	--
47	97	3.5	--	--	--	--	--	--	--
48	97	2.6	--	--	--	--	--	--	--

Note: ns - $P > 0.05$, comparisonwise
* - $P \leq 0.05$, comparisonwise
-- - determination not necessary due to results of a previous determination

[a] Significance was not tested because value was ≥90 percent (i.e., the minimum allowable value for negative controls).

[b] Normality was evaluated using the Wilk-Shapiro test.

[c] Homogeneous variance was evaluated using the *F*-test for variances.

- If the data failed one or both of the normality and variance tests, an arcsine transformation was applied to the survival data and the transformed data were again tested for normality and homogeneous variances using the Wilk-Shapiro test and the *F*-test, respectively
- If the data passed both the normality and variance tests, significance was determined for the transformed data using Student's *t*-test
- If the data failed one or both of the normality and variance tests, significance was determined using the nonparametric Mann-Whitney *U*-test, which does not assume normality or homogeneous variances
- If mean survival in the Ward Cove sample was determined to be significantly different from mean survival in Moser Bay, the value of mean survival was compared with the screening value for MCUL exceedance
- If mean survival in the significant Ward Cove sample was greater than the screening value for MCUL exceedance, it was classified as an SQS exceedance only
- If mean survival in the significant Ward Cove sample was less than the screening value for MCUL exceedance, it was classified as both an SQS and an MCUL exceedance
- If mean survival in the Ward Cove sample was not significantly different from mean survival in Moser Bay, the sample variance was compared with the screening value for data variability
- If the sample variability was less than the screening value, the nonsignificant Ward Cove sample was classified as not exceeding SQS or MCUL values
- If the sample variability was greater than the screening value, a power analysis would have been conducted, but was not required for any of the Ward Cove samples.

Summaries of the significance determinations for the toxicity results found in Ward Cove in 1996 and 1997 are presented in Tables 7-9 and 7-10. As discussed previously, no sediment samples exceeded test-specific SQS or MCUL values for the amphipod test based on *Leptocheirus plumulosus* or the juvenile polychaete test. This lack of significance of test responses for those two tests is consistent with the relatively low levels of adverse responses found throughout Ward Cove for both tests, compared to the responses found in Moser Bay (previous section).

SQS and MCUL values were exceeded in Ward Cove for the amphipod test using *Rhepoxynius abronius* and the echinoderm embryo test in both 1996 and 1997 (Tables 7-9 and 7-10). For the amphipod test, the SQS was exceeded at 16 of 28 stations (57 percent) in 1996 and at 23 of 33 stations (70 percent) in 1997. The MCUL for the

amphipod test was exceeded at 12 stations (43 percent) in 1996 and 19 stations (58 percent) in 1997. For the echinoderm embryo test, the SQS was exceeded at 22 of the 28 stations (79 percent) in 1996 and at 28 of 33 stations (85 percent) in 1997. The MCUL for the echinoderm test was exceeded at 16 stations (57 percent) in 1996 and at 14 stations (42 percent) in 1997.

The spatial patterns of SQS and MCUL exceedances for the amphipod test using *Rhepoxynius abronius* and the echinoderm embryo test are presented in Figures 7-6 and 7-7. For the amphipod test, SQS and MCUL exceedances were found at most stations near the KPC facility and downcurrent from the facility midway along the northern shoreline of the Cove. SQS and MCUL exceedances were also found near the fish cannery. For the echinoderm embryo test, SQS and MCUL exceedances were also found at most stations near the KPC facility and downcurrent from the facility along the northern shoreline of the Cove. However, in contrast to the amphipod test, exceedances were found upcurrent from the KPC facility and downcurrent along the northern shoreline as far as the mouth of the Cove. SQS and MCUL exceedances for the echinoderm embryo test were also found near the fish cannery.

7.1.2.3 Development of Site-Specific Sediment Quality Values

Sediment quality values were used to identify stations in Ward Cove at which potential sediment toxicity would be predicted based on observed concentrations of various benthic CoPCs. In general, site-specific sediment quality values are preferable to the Washington State sediment management standards because they factor in site-specific bioavailability, matrix effects, and the synergistic and antagonistic effects associated with the mixtures of chemicals in Ward Cove. Site-specific sediment quality values for Ward Cove (WCSQVs) were therefore developed for several of the benthic CoPCs that lacked Washington State sediment management standards (i.e., TOC, total ammonia, BOD, and COD) as well as one benthic CoPC for which a sediment management standard exists (i.e., 4-methylphenol). Although sediment management standards are not available for total sulfide, 2,3,7,8-TCDD, or TCDD TEC, WCSQVs were not developed for those CoPCs because the toxicological significance of bulk sediment concentrations of total sulfide is questionable and the primary ecological concern for 2,3,7,8-TCDD and TCDD TEC is bioaccumulation in the food web, rather than direct toxicity to benthic macroinvertebrates.

WCSQVs were not developed for most benthic CoPCs having Washington State sediment management standards (i.e., arsenic, cadmium, total mercury, zinc, benzoic acid, and phenol) because the concentration ranges found in Ward Cove were not substantially higher than the sediment management standards and because the range of concentrations found in Ward Cove was much less than the range of concentrations used to generate the standards for Puget Sound. Given the relatively low concentration ranges found for those chemicals in Ward Cove, it is unlikely that meaningful refinements to the sediment management standards could be made. By contrast, the range of concentrations of 4-methylphenol in Ward Cove was much higher than the sediment management standards and the concentrations typically found in Puget Sound. It therefore was possible to develop

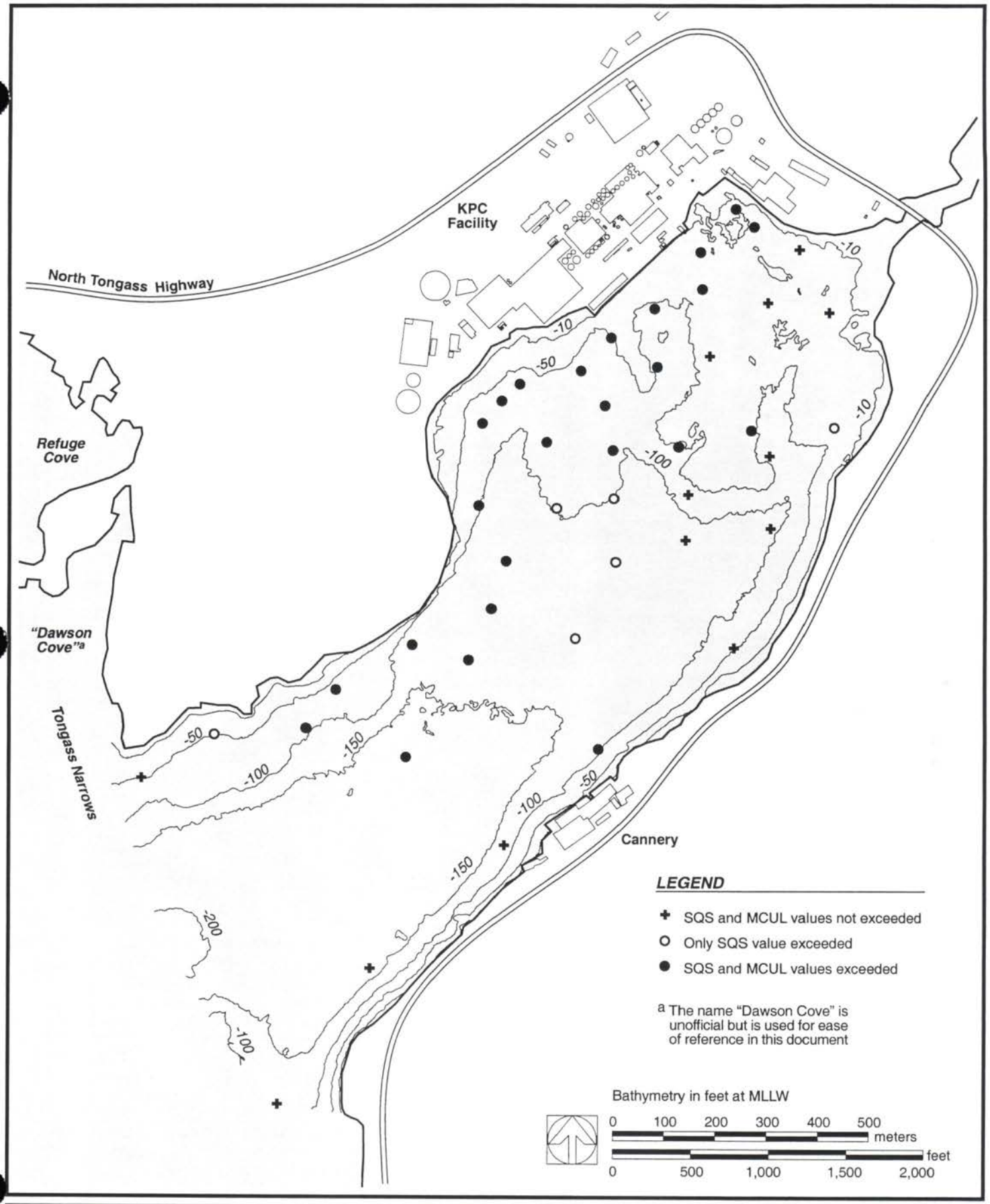


Figure 7-6. Distribution of exceedances of SQS and MCUL values for the amphipod test in Ward Cove in 1996 and 1997.

8600BOW.001 1602 05/11/99 WA

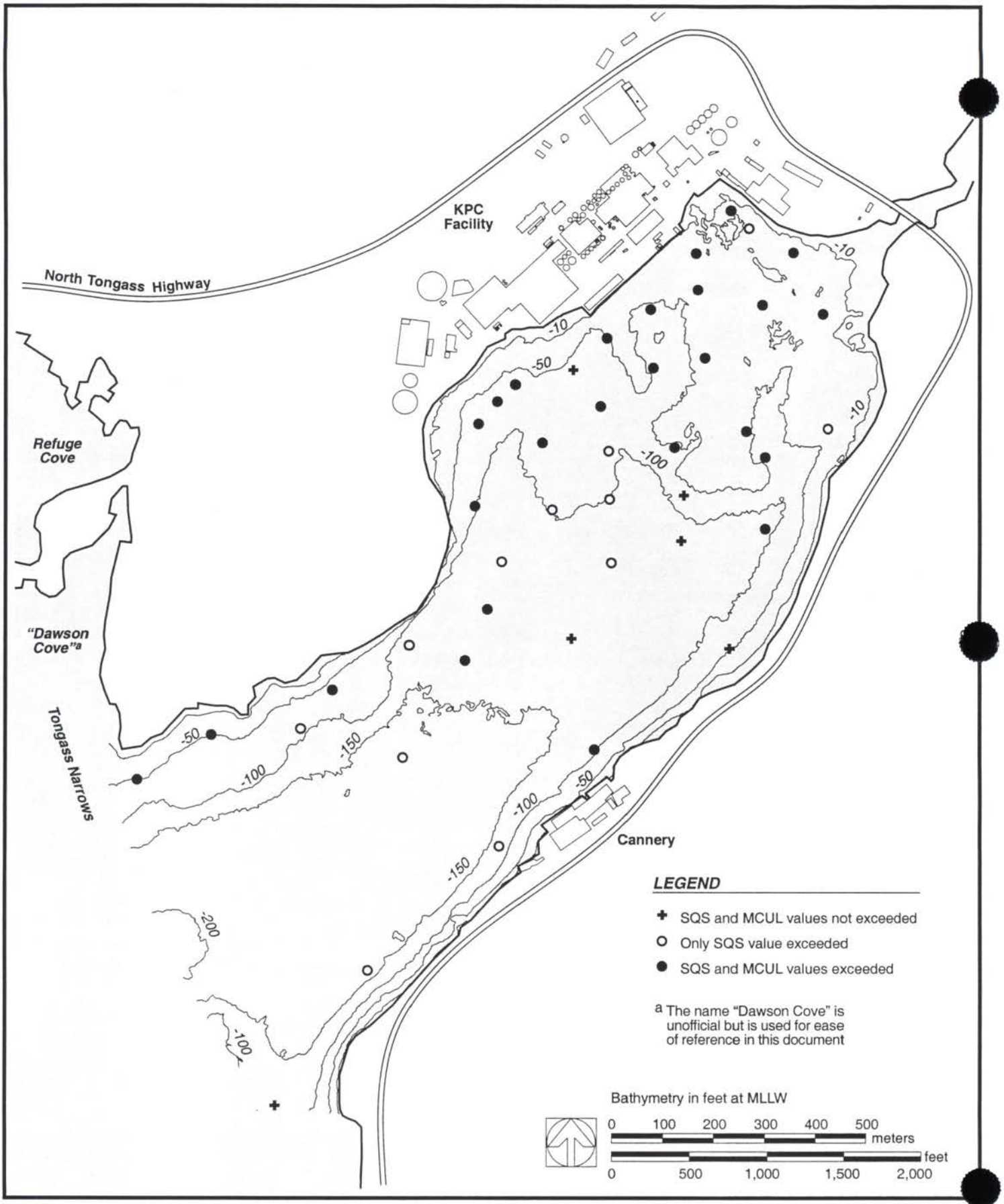


Figure 7-7. Distribution of exceedances of SQS and MCUL values for the echinoderm embryo test in Ward Cove in 1996 and 1997.

site-specific sediment quality values that reflected the site-specific conditions found in Ward Cove.

The site-specific WCSQVs were developed using the primary methods employed to determine the Washington State sediment management standards. Most of the standards were determined using the apparent effects threshold (AET) approach (Barrick et al. 1988). A chemical-specific AET value is defined as the concentration above which adverse biological effects are always observed for a particular data set. AET values can be developed for a range of biological indicators. The AET approach has been endorsed by EPA's Science Advisory Board (U.S. EPA 1989b) as a valid method for developing site-specific sediment quality values. The method is considered environmentally protective when it is based on a range of sensitive and representative biological indicators, such as those used to determine compliance with the Washington State sediment management standards and those used in the present study. As for sediment toxicity, the Washington State sediment management standards identify two kinds of sediment quality values for each chemical (SQS and MCUL values) that are used in the same manner as the analogous values for sediment toxicity. The SQS is based on the lowest AET value for a range of biological indicators, whereas the MCUL is based on the second lowest AET value observed for the indicators.

In addition to the AET approach, several other approaches have been used in the United States and Canada to develop sediment quality values. A comparison of various sediment quality values for metals, PAH compounds, and total PCBs is provided in Appendix J. There are numerous reviews of the advantages and disadvantages of the various approaches (e.g., Chapman 1989; Adams et al. 1992; Giesy and Hoke 1990; MacDonald et al. 1992; Persaud et al. 1992; U.S. EPA 1992b).

For the Ward Cove sediment assessment, two kinds of site-specific WCSQVs were developed. The $WCSQV_{(1)}$ (analogous to SQS) was based on the lowest AET values for all of the sediment toxicity tests evaluated in Ward Cove in 1996 and 1997 and is therefore analogous to the SQS. The $WCSQV_{(2)}$ (analogous to MCUL) was based on the second lowest AET value for the four toxicity tests and is therefore analogous to the MCUL.

Summaries of the data sets used to develop the AET values that were used as the WCSQVs for TOC, total ammonia, BOD, COD, and 4-methylphenol are provided in Tables 7-17 through 7-21. Although AET values were not calculated for total sulfide and TCDD TEC, matched chemical concentrations and sediment toxicity results for those two CoPCs are presented in Tables 7-22 and 7-23 for informational purposes only. In developing the WCSQVs, only matched sets of chemical and toxicity results (based on subsamples from the same homogenized sediment samples) were used from the stations sampled during 1996 and 1997. Therefore, when field replicates were collected at a station for separate chemical analyses (i.e., for quality assurance purposes), WCSQVs were based only on the replicate that was also subsampled for toxicity testing.

Two of the primary requirements for developing valid AET values are that the data be collected from a reasonable number of stations and that the data be broadly distributed

TABLE 7-17. SUMMARY OF RESULTS USED TO DETERMINE AET VALUES FOR TOC[a]

1996				1997			
Station	Concentration[b] (percent dry weight)	Amphipod Test	Echinoderm Test	Station	Concentration[b] (percent dry weight)	Amphipod Test	Echinoderm Test
5	36	X	X	5	38	X	X
6	33	X	X	38	34	X	X
1	32	X	X	2	33	X	X
16	31	X	X	37	31	X	-- [d]
17	31	-- [c]	X	3	30	X	X
26	30	--	--	35	30	--	X
9	27	X	X	34	29	X	X
10	27	--	X	16	28	--	X
4	26	X	X	17	28	X	X
7	26	--	X	47	26	X	X
14	25	X	X	44	26	X	X
15	25	X	X	7	26	X	X
8	24	X	X	48	25	X	X
12	24	X	X	4	25	X	X
3	22	--	X	42	24	X	X
13	22	X	X	39	23	X	--
21	21	--	--	40	23	--	--
27	21	--	--	32	23	X	X
28	20	X	X	13	22	X	X
19	18	X	--	41	22	--	X
20	17	X	--	31	21	X	X
2	14	X	X	12	21	X	X
11	14	--	X	45	21	X	X
23	13	--	X	27	20	--	X
24	13	--	X	11	19	--	X
25	11	X	X	28	19	X	X
22	5	--	--	43	18	X	X
18	1	--	X	19	17	X	X
				25	13	X	X
				23	9	--	--
				33	5	--	X
				18	4	--	X
				22	4	--	--

Note: AET - apparent effects threshold
TOC - total organic carbon
X - toxicity response was less than the sediment quality standard (SQS), indicating that an adverse effect was present
-- - toxicity response was greater than the SQS, indicating that no adverse response was present

[a] Chemical concentrations are also presented in Tables 7-1 and 7-2 and toxicity responses and associated SQS comparisons are presented in Tables 7-9 and 7-10.

[b] Concentrations are listed in rank order.

[c] AET for the amphipod test.

[d] AET for the echinoderm test.

TABLE 7-18. SUMMARY OF RESULTS USED TO DETERMINE AET VALUES FOR TOTAL AMMONIA[a]

1996				1997			
Station	Concentration[b] (mg/kg dry weight)	Amphipod Test	Echinoderm Test	Station	Concentration[b] (mg/kg dry weight)	Amphipod Test	Echinoderm Test
6	360	X	X	44	690	X	X
1	310	X	X	31	510	X	X
12	260	X	X	13	320	X	X
2	220	X	X	48	300	X	X
25	160	X	X	38	260	X	X
13	150	X	X	12	240	X	X
14	130	X	X	45	170	X	X
8	100	X	X	4	150	X	X
10	99	--	X	35	120	--[c]	X
4	97	X	X	34	120	X	X
21	88	--	--	47	120	X	X
20	84	X	--	7	120	X	X
15	83	X	X	25	120	X	X
9	82	X	X	39	110	X	--[d]
16	81	X	X	43	110	X	X
7	74	--	X	19	110	X	X
5	67	X	X	17	99	X	X
26	66	--	--	23	86	--	--
11	50	--	X	2	85	X	X
19	44	X	--	42	82	X	X
27	43	--	--	32	82	X	X
24	34	--	X	3	80	X	X
28	34	X	X	40	80	--	--
22	21	--	--	41	58	--	X
3	14	--	X	5	57	X	X
23	14	--	X	37	54	X	--
18	13	--	X	27	47	--	X
17	11	--	X	16	40	--	X
				11	34	--	X
				28	34	X	X
				33	23	--	X
				22	19	--	--
				18	13	--	X

Note: AET - apparent effects threshold

X - toxicity response was less than the sediment quality standard (SQS), indicating that an adverse effect was present

-- - toxicity response was greater than the SQS, indicating that no adverse response was present

[a] Chemical concentrations are also presented in Tables 7-1 and 7-2 and toxicity responses and associated SQS comparisons are presented in Tables 7-9 and 7-10.

[b] Concentrations are listed in rank order.

[c] AET for the amphipod test.

[d] AET for the echinoderm test.

TABLE 7-19. SUMMARY OF RESULTS USED TO DETERMINE AET VALUES FOR BOD[a]

1996				1997			
Station	Concentration[b] (g/kg dry weight)	Amphipod Test	Echinoderm Test		Concentration[b] (g/kg dry weight)	Amphipod Test	Echinoderm Test
9	19	X	X	38	65	X	X
16	18	X	X	4	64	X	X
1	16	X	X	3	46	X	X
14	16	X	X	2	45	X	X
6	13	X	X	23	37	-- [c]	-- [d]
4	12	X	X	25	34	X	X
8	12	X	X	27	34	--	X
20	11	X	-- [e]	28	32	X	X
5	10	X	X	11	14	--	X
12	10	X	X	35	14	--	X
27	10	--	--	16	13	--	X
28	10	X	X	44	13	X	X
2	9.9	X	X	13	12	X	X
10	9.8	--	X	31	11	X	X
19	9.6	X	--	34	10	X	X
25	9.2	X	X	17	10	X	X
7	8.7	--	X	48	9.2	X	X
26	8.5	--	--	5	9.2	X	X
13	8.3	X	X	32	9.1	X	X
23	7.9	--	X	45	9.1	X	X
17	7.6	--	X	19	8.5	X	X
3	7.3	--	X	7	8.0	X	X
24	7.0	--	X	40	7.8	--	--
11	6.4	--	X	39	7.7	X	--
21	6.2	--	--	43	7.4	X	X
15	6.0	X	X	47	7.1	X	X
22	3.5	--	--	37	7.1	X	--
18	1.4	--	X	42	6.9	X	X
				12	6.4	X	X
				41	6.4	--	X
				22	3.5	--	--
				33	1.7	--	X
				18	1.6	--	X

Note: AET - apparent effects threshold
BOD - biochemical oxygen demand
X - toxicity response was less than the sediment quality standard (SQS), indicating that an adverse effect was present
-- - toxicity response was greater than the SQS, indicating that no adverse response was present

[a] Chemical concentrations are also presented in Tables 7-1 and 7-2 and toxicity responses and associated SQS comparisons are presented in Tables 7-9 and 7-10.

[b] Concentrations are listed in rank order.

[c] AET for the amphipod test.

[d] This no-effect concentration was not used to set the AET because it is considered a chemical anomaly (i.e., it is more than three times greater than the next highest no-effect concentration).

[e] AET for the echinoderm test.

TABLE 7-20. SUMMARY OF RESULTS USED TO DETERMINE AET VALUES FOR COD[a]

1996				1997			
Station	Concentration[b] (g/kg dry weight)	Amphipod Test	Echinoderm Test	Station	Concentration[b] (g/kg dry weight)	Amphipod Test	Echinoderm Test
8	2,400	X	X	41	52	--	X
7	620	-- [c]	X	25	30	X	X
16	620	X	X	23	26	--	--
5	590	X	X	48	19	X	X
9	550	X	X	16	16	--	X
26	550	--	-- [d]	11	16	--	X
6	540	X	X	44	15	X	X
12	520	X	X	38	15	X	X
15	490	X	X	31	13	X	X
1	480	X	X	4	13	X	X
4	470	X	X	45	12	X	X
13	440	X	X	34	12	X	X
21	420	--	--	2	12	X	X
10	340	--	X	27	12	--	X
2	330	X	X	19	11	X	X
27	330	--	--	42	11	X	X
28	330	X	X	40	11	--	--
19	270	X	--	35	10	--	X
3	250	--	X	3	10	X	X
23	200	--	X	43	10	X	X
11	190	--	X	17	10	X	X
14	190	X	X	7	10	X	X
24	190	--	X	37	8.7	X	--
25	160	X	X	39	8.3	X	--
17	150	--	X	47	7.9	X	X
20	120	X	--	12	7.8	X	X
22	98	--	--	32	7.1	X	X
18	17	--	X	13	7.0	X	X
				22	6.5	--	--
				5	5.6	X	X
				28	5.6	X	X
				33	4.5	--	X
				18	2.2	--	X

Note: AET - apparent effects threshold
COD - chemical oxygen demand
X - toxicity response was less than the sediment quality standard (SQS), indicating that an adverse effect was present
-- - toxicity response was greater than the SQS, indicating that no adverse response was present

[a] Chemical concentrations are also presented in Tables 7-1 and 7-2 and toxicity responses and associated SQS comparisons are presented in Tables 7-9 and 7-10.

[b] Concentrations are listed in rank order.

[c] AET for the amphipod test.

[d] AET for the echinoderm test.

TABLE 7-21. SUMMARY OF RESULTS USED TO DETERMINE AET VALUES FOR 4-METHYLPHENOL[a]

1996				1997			
Station	Concentration[b] (µg/kg dry weight)	Amphipod Test	Echinoderm Test	Station	Concentration[b] (µg/kg dry weight)	Amphipod Test	Echinoderm Test
2	11,000	X	X	31	17,000	X	X
6	8,300	X	X	5	16,000	X	X
1	6,000	X	X	2	15,000	X	X
3	5,600	-- [c]	X	44	9,000	X	X
4	2,900	X	X	12	8,300	X	X
7	1,700	-- [d]	X	38	8,300	X	X
25	1,700	X	X	7	7,500	X	X
8	1,400	X	X	25	6,600	X	X
9	1,400	X	X	3	6,200	X	X
14	1,000	X	X	42	5,700	X	X
5	860	X	X	34	5,100	X	X
12	620	X	X	4	4,500	X	X
20	470	X	--	37	4,400	X	-- [c]
13	390	X	X	32	2,700	X	X
10	250 *U*	--	X	45	2,400	X	X
16	250 *U*	X	X	47	1,800	X	X
17	250 *U*	--	X	13	1,700	X	X
19	250 *U*	X	--	39	1,300	X	-- [e]
21	250 *U*	--	--	16	1,240	--	X
24	250 *U*	--	X	48	1,100	X	X
15	220	X	X	40	1,000	--	--
11	200 *U*	--	X	43	1,000	X	X
22	200 *U*	--	--	33	980	--	X
26	200 *U*	--	--	28	802	X	X
27	200 *U*	--	--	19	730	X	X
28	200 *U*	X	X	41	640	--	X
23	49	--	X	17	570	X	X
18	20 *U*	--	X	27	472	--	X
				35	460	--	X
				11	380	--	X
				23	168	--	--
				18	26	--	X
				22	24	--	--

Note: AET - apparent effects threshold

X - toxicity response was less than the sediment quality standard (SQS), indicating that an adverse effect was present

-- - toxicity response was greater than the SQS, indicating that no adverse response was present

[a] Chemical concentrations are also presented in Tables 7-1 and 7-2 and toxicity responses and associated SQS comparisons are presented in Tables 7-9 and 7-10.

[b] Concentrations are listed in rank order.

[c] This no-effect concentration was not used to set the AET because it is considered a chemical anomaly (i.e., it is more than three times greater than the next highest no-effect concentration).

[d] AET for the amphipod test.

[e] AET for the echinoderm test.

TABLE 7-22. SUMMARY OF CORRESPONDING CHEMICAL CONCENTRATIONS AND SEDIMENT TOXICITY RESULTS FOR TOTAL SULFIDE[a]

1996				1997			
Station	Concentration[b] (mg/kg dry weight)	Amphipod Test	Echinoderm Test	Station	Concentration[b] (mg/kg dry weight)	Amphipod Test	Echinoderm Test
17	27,000	--	X	32	13,000	X	X
16	16,000	X	X	16	12,000	--	X
4	6,500	X	X	31	11,000	X	X
10	5,500	--	X	43	9,700	X	X
5	5,400	X	X	38	6,700	X	X
3	5,300	--	X	19	5,500	X	X
9	4,500	X	X	45	4,800	X	X
13	4,300	X	X	2	4,500	X	X
27	4,300	--	--	27	4,500	--	X
21	3,500	--	--	28	4,400	X	X
8	2,700	X	X	48	3,900	X	X
12	2,700	X	X	23	3,900	--	--
15	2,700	X	X	25	3,800	X	X
28	2,400	X	X	40	3,800	--	--
6	2,200	X	X	4	3,700	X	X
14	2,200	X	X	35	3,300	--	X
26	2,200	--	--	47	3,000	X	X
7	1,800	--	X	13	2,700	X	X
1	1,700	X	X	39	2,700	X	--
11	1,500	--	X	37	2,700	X	--
2	1,200	X	X	44	2,300	X	X
23	1,200	--	X	34	2,300	X	X
25	1,000	X	X	5	2,300	X	X
19	800	X	--	11	2,300	--	X
24	670	--	X	42	2,000	X	X
20	420	X	--	12	1,900	X	X
22	380	--	--	7	1,900	X	X
18	150	--	X	33	1,600	--	X
				22	560	--	--
				3	500	X	X
				18	310	--	X
				17	50	X	X
				41	48	--	X

Note: X - toxicity response was less than the sediment quality standard (SQS), indicating that an adverse effect was present

-- - toxicity response was greater than the SQS, indicating that no adverse response was present

[a] Chemical concentrations are also presented in Tables 7-1 and 7-2 and toxicity responses and associated SQS comparisons are presented in Tables 7-9 and 7-10.

[b] Concentrations are listed in rank order.

TABLE 7-23. SUMMARY OF CORRESPONDING CHEMICAL CONCENTRATIONS AND SEDIMENT TOXICITY RESULTS FOR TCDD TEC[a]

1996				1997			
Station	Concentration[b] (µg/kg) organic carbon)	Amphipod Test	Echinoderm Test	Station	Concentration[b] (µg/kg) organic carbon)	Amphipod Test	Echinoderm Test
7	0.46	--	X	4	0.45	X	X
4	0.46	X	X	3	0.31	X	X
14	0.26	X	X	22	0.22	--	--
1	0.24	X	X	2	0.22	X	X
2	0.23	X	X	13	0.20	X	X
3	0.23	--	X	25	0.20	X	X
24	0.22	--	X	27	0.17	--	X
25	0.21	X	X	5	0.17	X	X
20	0.18	X	--	23	0.16	--	--
12	0.17	X	X	16	0.12	--	X
21	0.16	--	--	11	0.09	--	X
6	0.15	X	X	18	0.03	--	X
15	0.14	X	X				
26	0.14	--	--				
5	0.14	X	X				
9	0.12	X	X				
19	0.11	X	--				
18	0.10	--	X				
22	0.10	--	--				
13	0.08	X	X				
16	0.07	X	X				
23	0.06	--	X				
11	0.06	--	X				
27	0.05	--	--				
17	0.03	--	X				

Note: TCDD - tetrachlorodibenzo-*p*-dioxin
TEC - toxic equivalent concentration
TOC - total organic carbon
X - toxicity response was less than the sediment quality standard (SQS), indicating that an adverse effect was present
-- - toxicity response was greater than the SQS, indicating that no adverse response was present

[a] Chemical concentrations are also presented in Tables 7-1 and 7-2 and toxicity responses and associated SQS comparisons are presented in Tables 7-9 and 7-10.

[b] Concentrations are listed in rank order. Concentrations are normalized to station-specific TOC concentrations, except that a TOC concentration of 10 percent was used for all station-specific values that were ≥10 percent. Detection limits are included in the sum at half their value.

along a relatively large concentration range for each chemical. The data collected during 1996 and 1997 satisfied both of those requirements. Although it was originally recommended that AET values be developed using a minimum of 50 stations when the AET approach was first developed (Barrick et al. 1988), subsequent experience with the approach has shown that valid AET values can be developed with fewer stations, especially if the stations are broadly distributed across a relatively large concentration range. The development of WCSQVs for Ward Cove was therefore considered appropriate because they were developed using data from 37–61 stations that were broadly distributed across relatively large concentrations ranges (Tables 7-1 to 7-3).

A third consideration in developing AET values is that unusual substrate types such as slag do not reduce the bioavailability of the chemicals of interest, resulting in AET values that are unrealistically high. The presence of wood debris in many of the sediment samples may have reduced the bioavailability of TOC, resulting in AET values that are too high. However, as described in Section 2.3, the influence of larger woody debris on the laboratory analytical results was minimized because that kind of debris was removed from samples when they were collected and processed in the field.

A fourth consideration in developing AET values is that the data represent a good combination of impacted and unaffected stations. As shown in Tables 7-9 and 7-10, 39 of the total of 61 stations (64 percent) evaluated using the amphipod test based on *Rhepoxynius abronius* in both 1996 and 1997 were identified as impacted, whereas the remaining 22 stations (36 percent) were not considered impacted. For the echinoderm embryo test, 50 of the total of 61 stations (82 percent) evaluated in both 1996 and 1997 were identified as impacted, whereas only 11 stations (18 percent) were not considered impacted. These results indicate that the amphipod test was represented by a relatively good combination of impacted and unaffected stations, with approximately two-thirds of the stations falling into the impacted category. By contrast, the combination of stations for the echinoderm test was less desirable, with more than seven times as many impacted stations as unaffected stations. However, the observed similarities in many of the AET values for amphipod and echinoderm tests indicate that both tests often produced similar results despite their differences with respect to the combination of impacted and unaffected stations. This pattern indicates that the less desirable station combination for the echinoderm test did not appear to substantially affect its ability to generate meaningful AET values.

A fifth consideration in developing AET values is that an AET value should not be based on a chemically anomalous sample. As described by Gries and Waldow (1995), a chemically anomalous sample is one in which there is a threefold difference in chemical concentration between the concentration of the sample that set the AET value (i.e., the highest concentration of the unaffected stations) and the second highest concentration of the unaffected stations. If a chemical anomaly is found, the anomalous concentration is excluded from the AET calculation and the AET value is set by the second highest chemical concentration of the unaffected stations, provided that concentration is not also considered anomalous. In the present study, chemical anomalies were found for BOD (Table 7-19) and 4-methylphenol (Table 7-21).

As discussed in part above, review of the data sets used to generate AET values for the present study indicate that the data sets were adequate for developing valid AET values for all five CoPCs (Tables 7-17 through 7-21). Furthermore, the relatively close similarities between many of the AET values for the CoPCs indicate that the two toxicity tests appeared to be responding in a similar manner to the CoPCs in many cases and tend to confirm that the AET values are meaningful estimates of the site-specific toxicity of each CoPC in Ward Cove.

The CoPC concentrations observed in Ward Cove sediments in 1996 and 1997 are compared with sediment quality values in Tables 7-1 through 7-3. For stations at which multiple samples were evaluated for sediment chemical concentrations, only results for the first replicate sample are presented, because toxicity testing was conducted only on those replicates. As shown in Tables 7-4 and 7-7, none of the CoPCs measured at the two intertidal stations in Ward Cove in 1997 exceeded sediment quality values. Because Washington State sediment management standards exist for a subset of the substances measured only for compliance with the KPC NPDES permit, comparisons were also made between the concentrations of those substances observed in Ward Cove and the appropriate SQS and MCUL values (Tables 7-4 through 7-7). Concentrations of pulp mill compounds in Ward Cove sediments are also evaluated in this section.

Total Organic Carbon—The $WCSQV_{(1)}$ and $WCSQV_{(2)}$ for TOC were both 31 percent and were exceeded at three stations in 1996 (Stations 1, 5, and 6) and at three stations in 1997 (Stations 2, 5, and 38). All of those stations were located immediately offshore from the KPC facility near Outfall 001 or Outfall 002.

Total Ammonia—The $WCSQV_{(1)}$ for total ammonia (110 mg/kg) was exceeded at 7 stations in 1996 and at 13 stations in 1997. The $WCSQV_{(2)}$ (120 mg/kg) was exceeded at seven stations in 1996 and at eight stations in 1997. Exceedances of the $WCSQV_{(2)}$ generally were confined to areas immediately offshore from the KPC facility (Stations 1, 2, 4, 6, 31, and 38) and downcurrent from the facility midway along the northern shoreline of the Cove (Stations 12, 13, 14, 44, 45, and 48). The MCUL was also exceeded immediately offshore from the cannery (Station 25).

Biochemical Oxygen Demand—The $WCSQV_{(1)}$ for BOD (11 g/kg) was exceeded at seven stations in 1996 and at 13 stations in 1997. The $WCSQV_{(2)}$ (37 g/kg) was not exceeded at any station in 1996, but was exceeded at four stations in 1997 (Stations 2, 3, 4, and 38). All of the stations at which the $WCSQV_{(2)}$ was exceeded were located immediately offshore from the KPC facility.

Chemical Oxygen Demand—The $WCSQV_{(1)}$ for COD (550 g/kg) was exceeded at four stations in 1996, but was not exceeded at any station in 1997. The

$WCSQV_{(2)}$ (620 g/kg) was exceeded at only one station in 1996. That station (Station 8) was located approximately 150 m offshore from the KPC facility.

4-Methylphenol—The $WCSQV_{(1)}$ for 4-methylphenol (1,300 μg/kg) was exceeded at 9 stations in 1996 and at 17 stations in 1997. The $WCSQV_{(2)}$ (1,700 μg/kg) was exceeded at 5 stations in 1996 and at 16 stations in 1997. Exceedances of the $WCSQV_{(2)}$ generally were confined to areas offshore from the KPC facility (Stations 1–7, 31, 32, 34, 37, and 38) and downcurrent from the facility along the northern shoreline of the Cove (Stations 12, 42, 44, 45, and 47). The $WCSQV_{(2)}$ was also exceeded immediately offshore from the cannery (Station 25).

Cadmium—The SQS for cadmium (5.1 mg/kg) was exceeded at seven stations in 1996, but was not exceeded at any station in 1997. The MCUL (6.7 mg/kg) was exceeded at only one station in 1996. That station (Station 7) was located approximately 150 m offshore from the KPC facility.

Total Mercury—The SQS (0.41 mg/kg) and MCUL (0.58 mg/kg) for total mercury were exceeded at only one station in 1996 and were not exceeded at any station in 1997. The station with the exceedance (Station 3) was located immediately offshore from the KPC facility.

Zinc—The SQS (410 mg/kg) and MCUL (960 mg/kg) for zinc were not exceeded at any station in Ward Cove in 1996, but the SQS was exceeded at one station in 1997. That station (Station 25) was located immediately offshore from the cannery.

Phenol—The SQS for phenol (510 μg/kg) was exceeded at one station in 1996 (Station 2) and at three stations in 1997 (Stations 2, 5, and 25). The MCUL (1,200 μg/kg) was not exceeded at any station in either year.

NPDES Substances—The SQS (57 mg/kg) and MCUL (93 mg/kg) values for arsenic were not exceeded at any station in Ward Cove in 1996 or 1997. The SQS and MCUL values for benzoic acid are equal (650 mg/kg) and were exceeded at only two stations in 1996 (Stations 2 and 4) and at one station in 1997 (Station 4), which were all located immediately offshore from the KPC facility. None of the PAH compounds exceeded their respective SQS or MCUL values in either year, when normalized to TOC content of the sediment (Tables 7-6 and 7-7). The TOC normalization procedure was consistent with the method described in Section 4.4.1, where the maximum TOC concentration was set at 10 percent regardless of whether the station-specific TOC content exceeded that value.

Pulp Mill Compounds—No historical data are available on concentrations of pulp mill compounds in Ward Cove sediments. In addition, no sediment quality values are available with which the concentrations of pulp mill compounds found in Ward Cove in the Phase 1 investigation can be compared.

Only seven pulp mill compounds were detected in sediments from Ward Cove in 1996 (Table 7-8). All of the detected compounds are resin acids or fatty acids. Although sediment quality values are not available for any of the detected pulp mill compounds, an estimate of their potential toxicity in Ward Cove can be made by examining the results of the toxicity tests for Station 7, the location of the highest concentrations of pulp mill compounds measured in the Cove. Significant responses were not found in three of the toxicity tests conducted on sediments for Station 7 (Table 7-9). Survival in both amphipod tests was 90 percent or greater, and growth rate for the juvenile polychaete test (0.61 mg/day) was similar to the mean growth rate observed for Moser Bay (0.60 mg/day). Although normal survival in the echinoderm embryo test (61 percent) was significantly different ($P \leq 0.10$) from the mean value observed for Moser Bay (85 percent), only the SQS criterion was exceeded. These results indicate that although pulp mill compounds are present in sediments near the KPC facility, it is unlikely that they are a major source of toxicity in those sediments.

In summary, the observed exceedances of sediment quality values for CoPCs and other substances in Ward Cove were largely confined to within 300–400 m offshore from the KPC facility and downcurrent from the facility midway along the northern shoreline of the Cove. Exceedances of sediment quality values for several CoPCs were also found immediately offshore from the cannery.

7.1.2.4 Comparison of Toxicity and Chemistry Results

Potential relationships between results of the two toxicity tests that exhibited adverse responses in Ward Cove (i.e., the amphipod test using *Rhepoxynius abronius* and the echinoderm embryo test) and each CoPC were evaluated using the Spearman rank correlation coefficient (Tables 7-24 and 7-25). The correlation analysis was used as one kind of exploratory evaluation to identify the CoPCs that exhibited the strongest associations with the sediment toxicity results.

As for any test of correlation, a significant result should not be considered conclusive evidence of a cause-and-effect relationship between the two test variables. Correlation analysis is simply an evaluation of the "association" between two variables. Instead of a cause-and-effect relationship, a significant result may be due to both variables responding similarly to an unmeasured variable or it may be due simply to a random association between the variables. Regression analysis is more appropriate than correlation analysis for evaluating cause-and-effect relationships between independent and dependent variables. However, regression analysis is most appropriate when the experimenter controls the independent variable (as in a laboratory dose-response experiment). Although correlation cannot prove causation, a significant correlation can infer that a potential

TABLE 7-24. CORRELATIONS BETWEEN CoPC CONCENTRATIONS IN WARD COVE SEDIMENTS AND TOXICITY TEST RESPONSES IN 1996

CoPC	Number of Stations[a]	Amphipod Test		Echinoderm Embryo Test	
		Spearman r_s	*P* value	Spearman r_s	*P* value
TOC	28	-0.26 ns	0.089	-0.44 *	0.0089
Total ammonia	28	-0.80 *	<0.000001	-0.36 *	0.031
Total sulfide	28	-0.17 ns	0.194	-0.46 *	0.0067
BOD	28	-0.60 *	0.00037	-0.35 *	0.035
COD	28	-0.37 *	0.028	-0.31 ns	0.055
Cadmium	28	-0.32 *	0.047	0.11 ns	0.283
Total mercury	19	0.49 *	0.017	-0.06 ns	0.403
Zinc	28	-0.20 ns	0.156	-0.36 *	0.028
Phenol	10	-0.43 ns	0.107	-0.43 ns	0.107
4-Methylphenol	17	-0.28 ns	0.149	-0.29 ns	0.138
TCDD TEC	25	-0.22 ns	0.132	-0.02 ns	0.512

Note: * - $P \leq 0.05$ (one-tailed)
ns - $P > 0.05$
BOD - biochemical oxygen demand
COD - chemical oxygen demand
CoPC - chemical of potential concern
TCDD - tetrachlorodibenzo-*p*-dioxin
TEC - toxic equivalent concentration
TOC - total organic carbon

[a] Undetected values were not included for total mercury, phenol, and 4-methylphenol.

TABLE 7-25. CORRELATIONS BETWEEN CoPC CONCENTRATIONS IN WARD COVE SEDIMENTS AND TOXICITY TEST RESPONSES IN 1997

CoPC	Number of Stations	Amphipod Test		Echinoderm Embryo Test	
		Spearman r_s	*P* value	Spearman r_s	*P* value
TOC	33	-0.39 *	0.023	-0.14 ns	0.440
Total ammonia	33	-0.77 *	<0.000001	-0.14 ns	0.423
Total sulfide	33	-0.28 ns	0.119	-0.08 ns	0.647
BOD	33	-0.35 *	0.043	-0.13 ns	0.461
COD	33	-0.15 ns	0.397	-0.13 ns	0.472
4-Methylphenol	17	-0.77 *	<0.000001	-0.22 ns	0.224

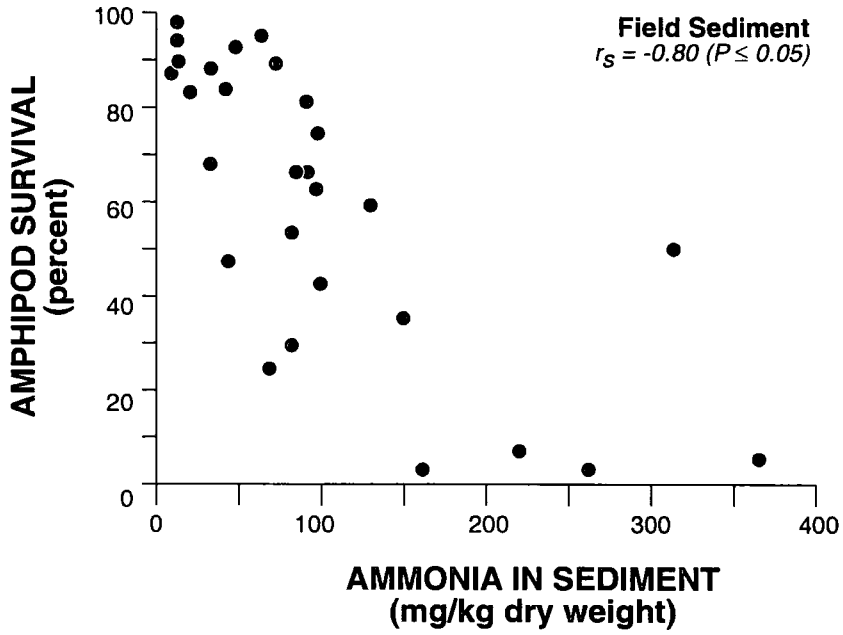
Note: * - $P \leq 0.05$ (one-tailed)
ns - $P > 0.05$
BOD - biochemical oxygen demand
COD - chemical oxygen demand
CoPC - chemical of potential concern
TOC - total organic carbon

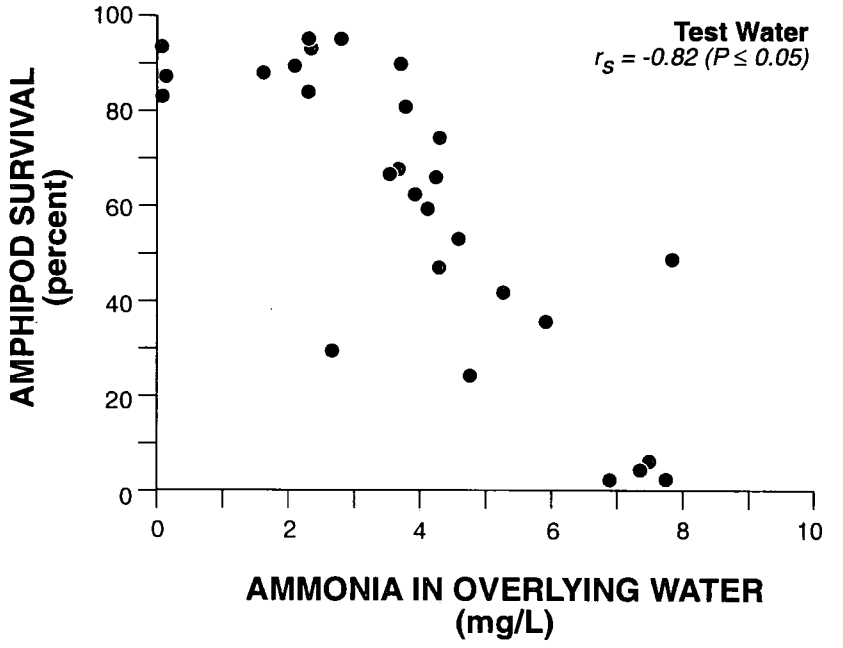
cause-and-effect relationship may exist between two variables and thereby allow more focused evaluations to be designed in which the experimenter has more control over the independent variable and can account for other potentially confounding variables. The correlation analysis conducted in the present study was used only to infer which CoPCs may warrant further evaluation with respect to the observed sediment toxicity.

For total mercury, phenol, and 4-methylphenol, only detected concentrations were used in the correlation analysis for the 1996 data (all values were detected in the 1997 data set), because undetected concentrations were not considered useful for evaluating gradients in CoPC concentrations that may be related to gradients in toxicity responses. The correlation analysis focused only on the toxicity-based CoPCs because they were generally found at concentrations greater than screening values at multiple stations in Ward Cove and therefore are present at levels at which potential adverse effects are predicted. In addition, those CoPCs were measured at numerous stations throughout the Cove, potentially covering large concentration gradients that are conducive to correlation analysis. Other chemicals measured during 1996 and 1997 (NPDES chemicals and pulp mill compounds) were not included in the correlation analysis because they generally were not present at concentrations at which adverse effects would be predicted and they were measured at only a limited number of stations.

The Spearman test was used as the measure of association between CoPC concentrations and toxicity results because it is a nonparametric (or distribution-free) test that is nearly as powerful as parametric correlation tests (Siegel 1956). Because the Spearman test is nonparametric, it does not require the data to meet the parametric assumptions of normality and homogeneous variance. In addition, it does not assume a linear relationship, as many parametric tests do. The Spearman test evaluates any kind of monotonic relationship (e.g., linear, curvilinear). The test is also less affected by outliers than are parametric tests.

The results of the correlation analysis for the 1996 data are presented in Table 7-24. Significant negative correlations ($P \leq 0.05$) were found between amphipod survival and five CoPCs (total ammonia, BOD, COD, cadmium, and total mercury) and between echinoderm embryo normal survival and five CoPCs (TOC, total ammonia, total sulfide, BOD, and zinc). The two variables that exhibited the strongest correlation ($r_s = -0.80$; $P \leq 0.000001$) were amphipod survival and sediment concentrations of total ammonia (Figure 7-8). The coefficient of determination for the relationship between amphipod survival and total ammonia (i.e., $r^2 = 0.64$) was relatively high, indicating that the relationship accounted for a substantial portion (64 percent) of the variability in the data set. The coefficients of determination for all of the other relationships presented in Table 7-24 indicate that none of them explained more than 40 percent of the variability in their respective data sets. Scatter plots for all CoPCs and all four sediment toxicity tests evaluated in 1996 are presented in Figures I-1 through I-11 in Appendix I to document the general lack of strong negative associations between most CoPC concentrations and toxicity responses.





r_S = *Spearman rank correlation coefficient*

Figure 7-8. Comparison of amphipod (*Rhepoxynius abronius*) survival and concentrations of ammonia in field sediment and Day-10 test water in 1996.

The results of the correlation analysis for the 1997 data are presented in Table 7-25. Significant negative correlations ($P \leq 0.05$) were found between amphipod survival and four CoPCs (TOC, total ammonia, BOD, and 4-methylphenol). By contrast, no significant correlation was found between echinoderm embryo normal survival and any of the CoPCs. The strongest correlations ($r_s = -0.77$; $P \leq 0.000001$) were found between amphipod survival and sediment concentrations of total ammonia (Figure 7-9) and 4-methylphenol (Figure 7-10). The coefficients of determination for those relationships (i.e., $r^2 = 0.59$) were relatively high, indicating that the relationships accounted for a substantial portion (59 percent) of the variability in the data set. The coefficients of determination for all of the other relationships presented in Table 7-25 indicate that none of them explained more than 15 percent of the variability in the respective data sets. Scatter plots for all CoPCs and the two sediment toxicity tests evaluated in 1997 are presented in Figures I-12 through I-14 in Appendix I to document the general lack of strong negative associations between most CoPC concentrations and toxicity responses.

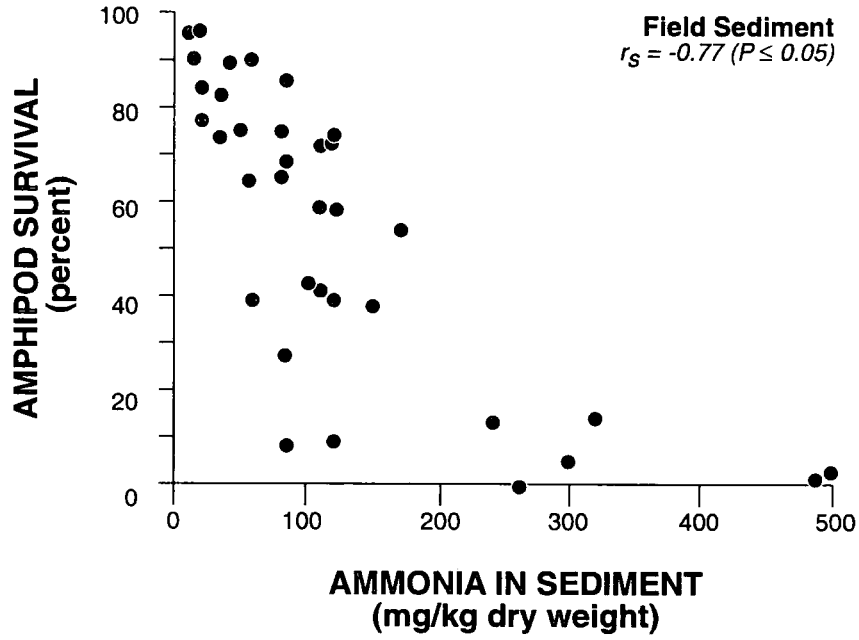
Because survival of *Rhepoxynius abronius* can be affected by sediment grain-size distribution, the potential relationship between amphipod survival and percent fines was evaluated. The correlations between those two variables in 1996 ($r_s = 0.001$) and 1997 ($r_s = -0.19$) were not significant ($P > 0.05$), indicating that amphipod survival was not strongly influenced by the grain-size distributions found for Ward Cove sediments during either year.

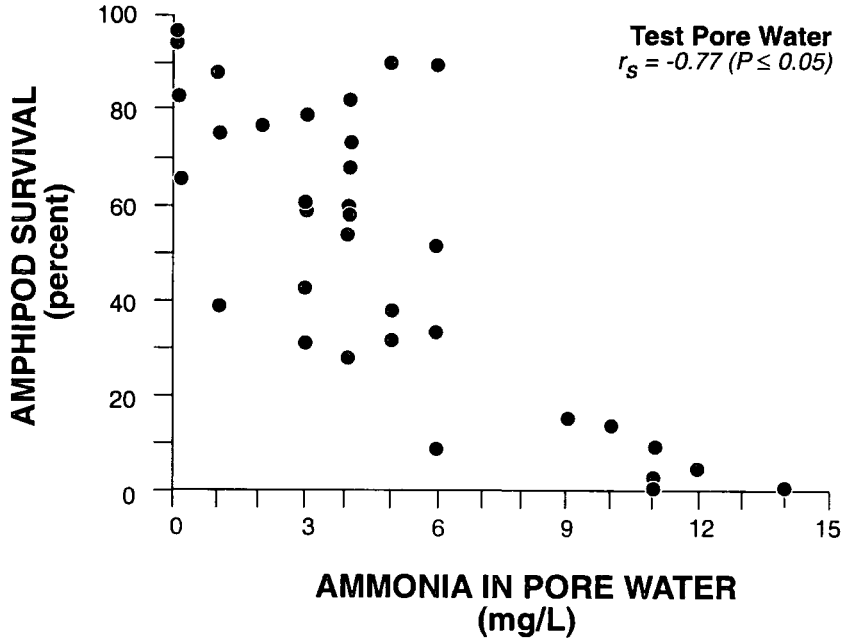
The strong negative relationship between *Rhepoxynius abronius* survival and total ammonia in Ward Cove sediments was further evaluated by comparing amphipod survival with concentrations of total ammonia in the overlying water (1996) and pore water (1997) of the toxicity test chambers on the final day of the 10-day exposure period using the Spearman rank correlation coefficient (Figures 7-8 and 7-9). Significant correlations ($P \leq 0.05$) were found between the two variables in both cases (Figures 7-8 and 7-9). The strength of the correlations was similar to the strength of the correlation between amphipod survival and total ammonia concentrations in sediments in both years. Porewater concentrations of sulfide were also measured in the toxicity test chambers in 1997 and also showed a significant ($P \leq 0.05$) negative correlation with amphipod survival (Figure 7-11).

In summary, the correlation analysis indicated that at least three CoPCs may be related in some manner to the observed patterns of amphipod survival in sediments from Ward Cove and warrant further evaluation. The three CoPCs are ammonia, sulfide, and 4-methylphenol, and they are evaluated in greater detail in Section 7.1.4. Normal survival of echinoderm embryos did not exhibit a strong relationship with sediment concentrations of any of the CoPCs in either year.

7.1.3 Summary and Historical Comparison of NPDES Data

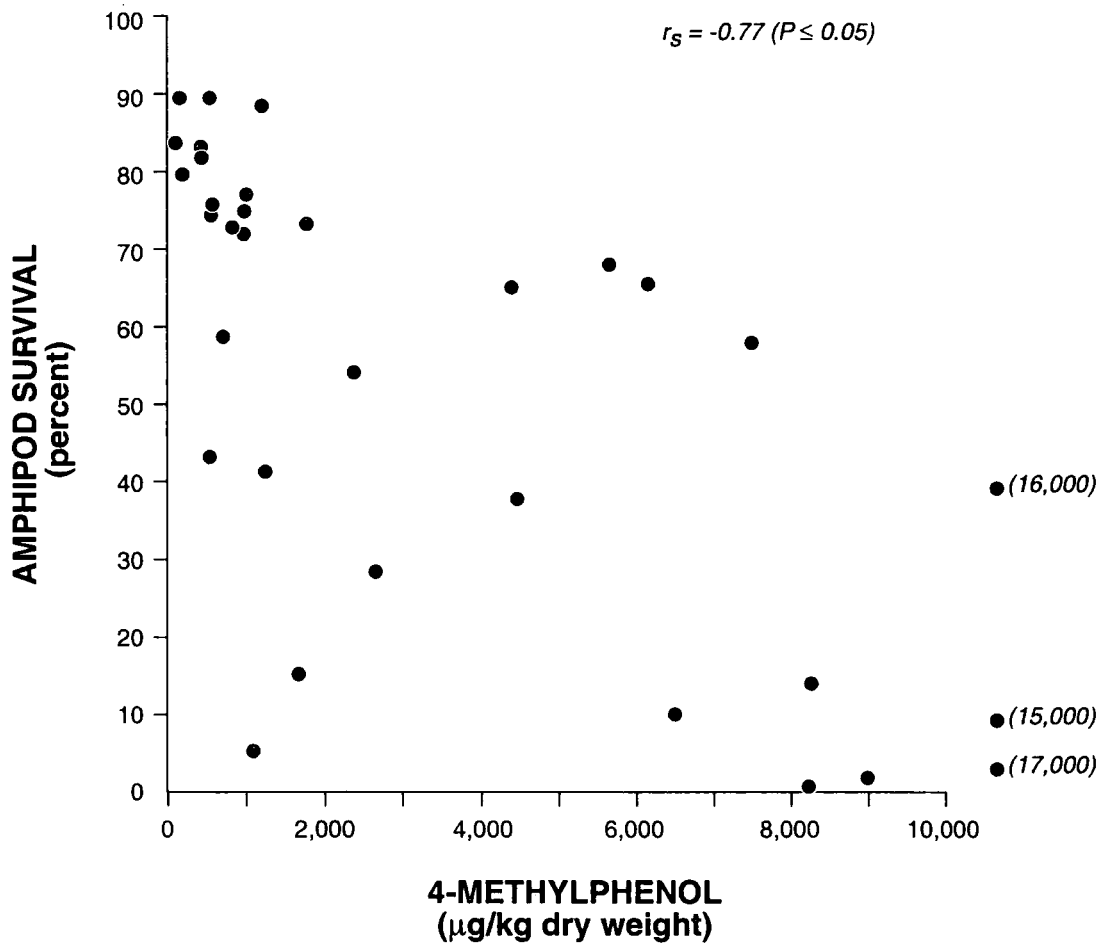
In this section, the data on sediment chemistry and toxicity collected in the Phase 1 and Phase 2 investigations in 1996 and 1997 are compared with the corresponding data





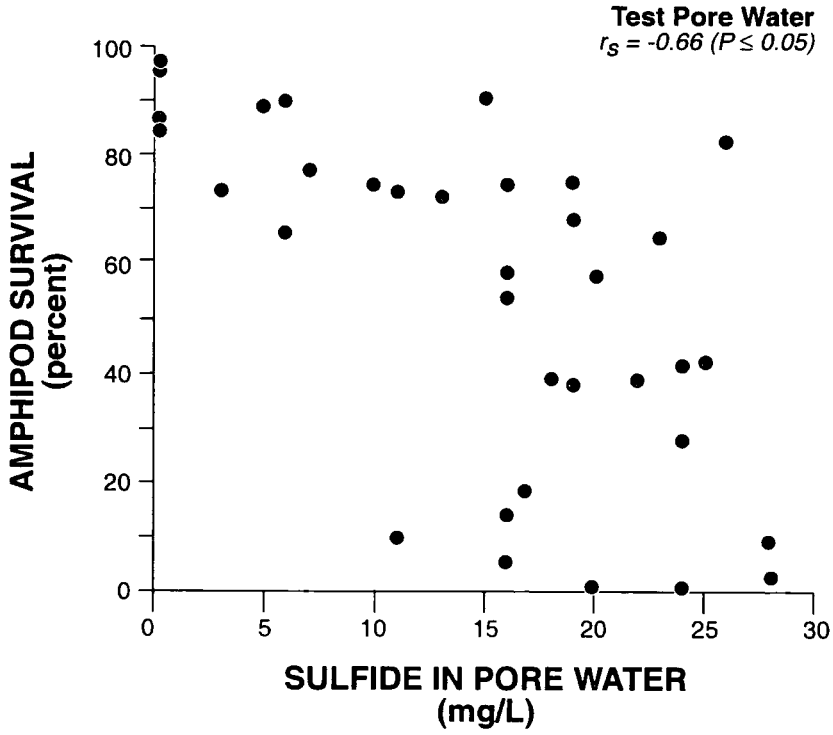
r_s = *Spearman rank correlation coefficient*

Figure 7-9. Comparison of amphipod (*Rhepoxynius abronius*) survival and concentrations of ammonia in field sediment and Day-10 test pore water in 1997.



r_S = *Spearman rank correlation coefficient*

Figure 7-10. Comparison of amphipod (*Rhepoxynius abronius*) survival and concentrations of 4-methylphenol in field sediment in 1997.



r_S = *Spearman rank correlation coefficient*

Figure 7-11. Comparison of amphipod (*Rhepoxynius abronius*) survival and concentrations of sulfide in Day-10 test pore water in 1997.

collected in 1994 and 1995 as part of the KPC NPDES sediment monitoring program (ENSR 1994, 1995b). Comparisons were made for the 12 NPDES stations sampled in Ward Cove during multiple years. The primary objective of these comparisons was to evaluate the consistency of the chemical and toxicity results among years and to determine whether substance concentrations and toxicity responses for the top 2 cm of sediment (evaluated in 1994 and 1995) differed from concentrations and responses for the top 10 cm of sediment (evaluated in 1996 and 1997). In particular, the comparisons were made to determine whether the subsurface sediments included in the 10-cm interval would result in greater chemical concentrations or toxicity responses than the top 2 cm of sediment. Greater concentrations or responses in subsurface sediments would suggest that the surface sediments may be overlying more contaminated sediments. Evaluation of the upper 2-cm horizon would therefore provide a less conservative estimate of potential ecological risks for those receptors exposed to subsurface sediments. As part of this analysis, chemical concentrations measured in all three studies were compared to the sediment quality values described previously.

In comparing chemical concentrations and toxicity responses among years, it is important to recognize that station locations likely varied somewhat as the result of the normal navigational variability associated with reoccupying stations in the field and the site-specific variability of the sediment characteristics found in Ward Cove. Because many areas of the Cove contain wood debris and are difficult to sample, stations were sometimes repositioned slightly to obtain acceptable sediment samples. In addition, because some variables such as AVS and total sulfide may vary seasonally, some observed differences among years for these chemicals may have been due to the different seasons in which sampling was conducted in 1994 (November), 1995 (December), 1996 (May–June), and 1997 (August).

In addition to allowing comparisons among years to be evaluated, the presentation of all of the chemical and toxicity results collected at the 12 NPDES stations in 1996 serves as a summary of the data collected to satisfy the sediment monitoring requirements of the KPC NPDES permit for 1996 and 1997.

7.1.3.1 Chemical Concentrations

The comparisons of chemical concentrations among years at the 12 NPDES stations in Ward Cove are presented in Tables 7-26 through 7-29. The concentrations observed in the top 10 cm in 1996 and 1997 generally were similar to or less than the concentrations observed in the top 2 cm during historical studies in 1994 and 1995. However, the following major exceptions were found:

- **Methylmercury:** In 1996, the concentrations of methylmercury at Stations 4 and 22 (5.4–10 μg/kg) were considerably greater than the respective historical values (1.1–2.9 μg/kg).

TABLE 7-26. CONVENTIONAL ANALYTES MEASURED AT NPDES STATIONS IN WARD COVE SEDIMENTS BETWEEN 1994 AND 1997 AND COMPARISON WITH WARD COVE SEDIMENT QUALITY VALUES

	Fines (percent)				TOC (percent)				AVS (mg/kg)			
Station	1994	1995	1996	1997	1994	1995	1996	1997	1994	1995	1996	1997
Upstream												
18	25	25	6.1	7.5	7	8.7	1	4	2,100	3,100	240	580
Facility												
5	ND	40	31	55	ND	30	36 **	38 **	ND	12,000	2,000	3,700
4	47	46	64	66	33 **	30	26	25	6,600	9,600	2,400	4,500
3	54	46	24	53	31	27	22	30	8,500	9,300	2,800	2,500
2	41	39	30	45	36 **	32 **	14	33 **	3,500	9,900	2,200	1,600
Downstream												
13	50	55	77	72	27	21	22	22	3,300	2,500	320	4,300
11	52	57	26	27	29	19	14	19	3,000	3,700	1,500	3,000
Offshore												
16	ND	37	65	59	ND	21	31	28	ND	6,200	13,000	17,000
27	ND	43	66	65	ND	22	21	20	ND	2,700	3,200	5,300
Cannery												
25	60	56	46	50	11	12	11	13	3,200	3,400	4,200	5,800
Outer Cove												
23	36	56	67	80	11	11	13	9	1,600	3,500	2,100	3,900
Tongass Narrows												
22	41	21	39	34	2	1.5	5	4	62	40	540	680
$WCSQV_{(1)}$		**NA**				**31**				**NA**		
$WCSQV_{(2)}$		**NA**				**31**				**NA**		

TABLE 7-26. (cont.)

Station	Total Sulfide (mg/kg) 1994	1995	1996	1997	BOD (g/kg) 1994	1995	1996	1997	COD (g/kg) 1994	1995	1996	1997
Upstream												
18	2,800	1,050	150	310	0.13	8.1	1.4	1.6	110	150	17	2.2
Facility												
5	ND	4,600	5,400	2,300	ND	11	10	9.2	ND	570 *	590 *	5.6
4	5,000	5,600	6,500	3,700	4.3	13 *	12 *	64 **	360	560 *	470	13
3	6,500	5,700	5,300	500	4.5	17 *	7.3	46 **	390	650 **	250	10
2	2,300	2,600	1,200	4,500	0.75	16 *	9.9	45 **	380	810 **	330	12
Downstream												
13	2,000	1,900	4,300	2,700	1.4	8.6	8.3	12 *	410	770 **	440	7.0
11	1,600	2,100	1,500	2,300	0.7	8.4	6.4	14 *	390	440	190	16
Offshore												
16	ND	1,800	16,000	12,000	ND	8.6	18 *	13 *	ND	360	620 *	16
27	ND	1,400	4,300	4,500	ND	9.3	10	34 *	ND	490	330	12
Cannery												
25	2,700	1,600	1,000	3,800	3.3	7.5	9.2	34 *	190	360	160	30
Outer Cove												
23	580	1,200	1,200	3,900	0.35	5.8	7.9	37 *	110	230	200	26
Tongass Narrows												
22	47	20 *U*	380	560	0.13	0.72	3.5	3.5	36	42	98	6.5
$WCSQV_{(1)}$		**NA**				**11**				**550**		
$WCSQV_{(2)}$		**NA**				**37**				**620**		

Note: All concentrations reported on dry weight basis.

* - concentration exceeds $WCSQV_{(1)}$

** - concentration exceeds $WCSQV_{(2)}$

AVS - acid-volatile sulfide

BOD - biochemical oxygen demand

COD - chemical oxygen demand

NA - not available

ND - no data

NPDES - National Pollutant Discharge Elimination System

TOC - total organic carbon

U - undetected at concentration listed

$WCSQV_{(1)}$ - Ward Cove sediment quality value analogous to sediment quality standard

$WCSQV_{(2)}$ - Ward Cove sediment quality value analogous to minimum cleanup level

TABLE 7-27. METALS CONCENTRATIONS MEASURED AT NPDES STATIONS IN WARD COVE SEDIMENTS BETWEEN 1994 AND 1997 AND COMPARISON WITH WASHINGTON STATE SEDIMENT MANAGEMENT STANDARDS

	Arsenic (mg/kg)				Cadmium (mg/kg)				Total Mercury (mg/kg)			
Station	1994	1995	1996	1997	1994	1995	1996	1997	1994	1995	1996	1997
Upstream												
18	7.6	13	2.7	3.6	1.2	1.6	0.2	0.26	ND	0.2 *U*	0.1 *U*	0.2 *U*
Facility												
5	ND	27	8.5	8.7	ND	4.0	1.3	1.5	ND	0.2 *U*	0.1 *U*	0.2 *U*
4	18	18	29	31	5.2 *	3.7	4.3	4.8	ND	0.2 *U*	0.2	0.2 *U*
3	25	30	16	25	4.4	5.2 *	1.3	3.6	ND	0.2 *U*	0.7 **	0.2 *U*
2	26	35	18	23	3.5	5.6 *	2.3	3.0	ND	0.2 *U*	0.1 *U*	0.2 *U*
Downstream												
13	39	40	33	29	6.6 *	6.7 *	5.2 *	4.4	ND	0.2 *U*	0.1	0.2 *U*
11	24	22	17	17	4.0	5.7 *	2.4	2.6	ND	0.2	0.1 *U*	0.2 *U*
Offshore												
16	ND	30	19	18	ND	4.2	3.7	2.5	ND	0.2 *U*	0.1 *U*	0.2 *U*
27	ND	39	26	34	ND	5.4 *	4.7	5.0	ND	0.2 *U*	0.1	0.2 *U*
Cannery												
25	35	37	24	24	5.3 *	6.9 **	3.7	5.1	ND	0.2	0.1	0.2 *U*
Outer Cove												
23	20	31	29	19	1.2	3.0	2.5	2.3	ND	0.2	0.2	0.2 *U*
Tongass Narrows												
22	5.2	6.5	11	11	1.4	1.6	1.0	0.8	ND	0.2 *U*	0.1 *U*	0.2 *U*
SQS		**57**				**5.1**				**0.41**		
MCUL		**93**				**6.7**				**0.58**		

TABLE 7-27. (cont.)

Station	Methylmercury (µg/kg) 1994	1995	1996	1997	Zinc (mg/kg) 1994	1995[a]	1996	1997
Upstream								
18	1.4	1.4	0.8	0.3	110	1,700	43	39
Facility								
5	ND	1.3	0.6	0.6	ND	630	120	170
4	2.9	1.9	10	1.3	470 *	2,100	280	400
3	1.8	2.1	0.8	1.2	450 *	2,100	210	220
2	1.2	0.8	0.6	0.4	220	430	140	200
Downstream								
13	2.7	4.6	6.9	3.6	190	2,800	140	140
11	5.0	6.4	3.5	0.7	150	680	120	100
Offshore								
16	ND	1.5	1.0	0.5	ND	540	190	180
27	ND	3.9	3.1	3.6	ND	850	130	170
Cannery								
25	3.4	6.4	0.5	0.2	340	1,000	340	530 *
Outer Cove								
23	5.9	19	9.5	14.3	100	1,700	160	130
Tongass Narrows								
22	0.4	1.1	5.4	3.4	60	770	69	62
SQS		**NA**				**410**		
MCUL		**NA**				**960**		

Note: All concentrations reported on dry weight basis.

Data shown represent NPDES stations only. Data for 1994 and 1995 are from ENSR (1994, 1995b). Data for 1996 and 1997 are a subset of the data from the present investigation for corresponding NPDES stations. Complete data from the present investigation can be found in Appendix A.

* - concentration exceeds sediment quality standard (SQS)
** - concentration exceeds minimum cleanup level (MCUL)
NA - SQS and MCUL values not available
ND - no data
NPDES - National Pollutant Discharge Elimination System
U - undetected at concentration listed

[a] The 1995 data set for zinc is considered unreliable because all 1995 concentrations are inconsistent with the concentrations found in 1994, 1996, and 1997.

TABLE 7-28. CONCENTRATIONS OF SELECTED ORGANIC COMPOUNDS MEASURED AT NPDES STATIONS IN WARD COVE SEDIMENTS BETWEEN 1994 AND 1997

Station	Phenol (µg/kg dry weight)				4-Methylphenol (µg/kg dry weight)				Benzoic Acid (µg/kg dry weight)			
	1994	1995	1996	1997	1994	1995	1996	1997	1994	1995	1996	1997
Upstream												
18	42	36	15	12	100	160	20 *U*	26	190	300	100 *U*	151
Facility												
5	ND	106	150	910	ND	840	860	16,000 **	ND	100 *U*	500 *U*	100 *U*
4	210	340	170	220	2,000 **	3,300 **	2,900 **	4,500 **	480	26	1,600 **	870 **
3	800 *	50	110	200	1,500 *	430	5,600 **	6,200 **	430	340	500 *U*	100 *U*
2	820 *	400	510 *	910 *	9,100 **	15,000 **	11,000 **	15,000 **	550	760 **	990 **	100 *U*
Downstream												
13	800 *	680 *	200 *U*	150	1,200	2,500 **	390	1,700 *	580	400	500 *U*	540
11	36	36	200 *U*	53	110	390	200 *U*	380	420	240	500 *U*	340
Offshore												
16	ND	720 *	360	100	ND	450	250 *U*	1,200	ND	420	500 *U*	400
27	ND	900 *	200 *U*	57	ND	590	200 *U*	480	ND	340	500 *U*	600
Cannery												
25	96	1,600 **	130	990 *	2,100 **	1,200	1,700 *	6,600 **	390	340	500 *U*	100 *U*
Outer Cove												
23	68	14	46	48	34	82	49	170	120	240	500 *U*	270
Tongass Narrows												
22	20 *U*	20 *U*	200 *U*	17	20 *U*	20 *U*	200 *U*	24	99	100 *U*	500 *U*	63
SQS/WCSQV(1)		**420** [c]				**1,300** [d]				**650** [c]		
MCUL/WCSQV(2)		**1,200** [c]				**1,700** [d]				**650** [c]		

TABLE 7-28 (cont.)

Station	2,3,7,8-TCDD[a,b] (μg/kg organic carbon)				TCDD TEC[a,b] (μg/kg organic carbon)			
	1994	1995	1996	1997	1994	1995	1996	1997
Upstream								
18	0.007 *U*	0.004	0.06 *U*	0.02 *U*	0.03	0.06	0.10	0.03
Facility								
5	ND	0.02	0.02 *U*	0.01	ND	0.34	0.14	0.17
4	0.01 *U*	0.03	0.03	0.02	0.24	0.62	0.46	0.45
3	0.03	0.02	0.01 *U*	0.01	0.44	0.62	0.23	0.31
2	0.03 *U*	0.02	0.01 *U*	0.02	0.18	0.42	0.23	0.22
Downstream								
13	0.02 *U*	0.02	0.01 *U*	0.02 *U*	0.15	0.25	0.08	0.20
11	0.004 *U*	0.01	0.01 *U*	0.01	0.05	0.17	0.06	0.09
Offshore								
16	ND	0.01	0.01 *U*	0.01	ND	0.09	0.07	0.12
27	ND	0.01	0.03 *U*	0.01	ND	0.14	0.05	0.17
Cannery								
25	0.01 *U*	0.01	0.02 *U*	0.01	0.08	0.18	0.21	0.20
Outer Cove								
23	0.004 *U*	0.01	0.02 *U*	0.01 *U*	0.03	0.15	0.06	0.16
Tongass Narrows								
22	0.04 *U*	0.01 *U*	0.02 *U*	0.02 *U*	0.05	0.14	0.10	0.22
SQS/WCSQV$_{(1)}$		**NA**				**NA**		
MCUL/WCSQV$_{(2)}$		**NA**				**NA**		

Note: Data shown represent NPDES stations only. Data for 1994 and 1995 are from ENSR (1994, 1995b). Data for 1996 and 1997 are a subset of the data from the present investigation for corresponding NPDES stations. Complete data from the present investigation can be found in Appendix A.

* - concentration exceeds the sediment quality standard (SQS) or WCSQV$_{(1)}$
** - concentration exceeds the minimum cleanup level (MCUL) or WCSQV$_{(2)}$
NA - SQS and MCUL values not available
ND - no data
NPDES - National Pollutant Discharge Elimination System
TCDD - tetrachlorodibenzo-*p*-dioxin
TEC - toxic equivalent concentration
U - undetected at concentration listed
WCSQV$_{(1)}$ - Ward Cove sediment quality value analogous to sediment quality standard
WCSQV$_{(2)}$ - Ward Cove sediment quality value analogous to minimum cleanup level

TABLE 7-28 (cont.)

[a] Detection limits are included in the sum at half their value.

[b] Concentrations are normalized to station-specific TOC concentrations, except that a TOC concentration of 10 percent was used for all station-specific values that were ≥ 10 percent.

[c] Washington State sediment management standard.

[d] Site-specific sediment quality value.

TABLE 7-29. CONCENTRATIONS OF PAH COMPOUNDS MEASURED AT NPDES STATIONS IN WARD COVE SEDIMENTS BETWEEN 1994 AND 1997

Station	Naphthalene				Acenaphthylene				Acenaphthene				Fluorene			
	1994	1995	1996	1997	1994	1995	1996	1997	1994	1995	1996	1997	1994	1995	1996	1997
Upstream																
18	11	12	1	10 *U*	10 *U*	10 *U*	10 *U*	10 *U*	10 *U*	10 *U*	10 *U*	10 *U*	10 *U*	12	10 *U*	10 *U*
Facility																
5	ND	10 *U*	49	190	ND	10 *U*	100 *U*	20 *U*	ND	60	60	140	ND	120	67	140
4	97	360	200	310	28	20	34	20 *U*	110	260	170	260	140	240	170	300
3	110	200	440	250	24	14	100 *U*	20 *U*	160	170	500	230	160	240	470	260
2	33	100 *U*	86	140	19 *U*	100 *U*	100 *U*	20 *U*	35	100 *U*	68	95	39	100 *U*	64	110
Downstream																
13	25	44	54	140	25 *U*	22	100 *U*	20 *U*	25 *U*	22	100 *U*	24	25 *U*	32	20	38
11	22 *U*	26	24	37	22 *U*	14	100 *U*	20 *U*	22 *U*	14	100 *U*	20 *U*	22 *U*	28	20	20 *U*
Offshore																
16	ND	16	12	54	ND	10 *U*	50 *U*	20 *U*	ND	10 *U*	32	82	ND	10 *U*	34	110
27	ND	10 *U*	17	51	ND	10 *U*	100 *U*	20 *U*	ND	10 *U*	100 *U*	31	ND	10 *U*	21	52
Cannery																
25	18	34	24	52	67	46	100	35	25	26	37	42	48	46	110	92
Outer Cove																
23	13	24	20	21	17	32	110	20	12 *U*	20	34	20 *U*	55	44	99	31
Tongass Narrows																
22	10 *U*	10 *U*	100 *U*	11	10 *U*	10 *U*	12	11	10 *U*	10 *U*	100 *U*	10 *U*	10 *U*	10 *U*	12	14

TABLE 7-29. (cont.)

Station	Phenanthrene				Anthracene				2-Methyl-naphthalene				Fluoranthene			
	1994	1995	1996	1997	1994	1995	1996	1997	1994	1995	1996	1997	1994	1995	1996	1997
Upstream																
18	28	88	6	16	11	10 *U*	3	10 *U*	10 *U*	10 *U*	10 *U*	10 *U*	73	10 *U*	15	30
Facility																
5	ND	320	270	390	ND	130	62	70	ND	46	74	200	ND	280	690	560
4	530	1,000	670	920	180	320	190	250	110	150	140	280	820	1,600	1,300	2,200
3	560	680	1,100	900	120	240	260	230	120	72	280	170	870	1,100	1,900	1,400
2	170	500	350	480	21	100 *U*	62	100	64	100 *U*	87	150	230	1,000	630	560
Downstream																
13	130	260	130	220	25 *U*	44	34	53	120	46	25	63	150	240	270	330
11	74	100	150	101	22 *U*	28	41	36	45	32	22	46	110	220	340	200
Offshore																
16	ND	98	97	310	ND	28	49	86	ND	46	15	79	ND	220	330	420
27	ND	60	120	220	ND	10 *U*	40	84	ND	10 *U*	18	69	ND	92	300	390
Cannery																
25	310	360	900	550	110	140	380	330	20	32	22	60	520	740	1,500	960
Outer Cove																
23	170	300	850	230	190	88	360	61	29	18	20	26	210	500	1,000	460
Tongass Narrows																
22	37	44	110	97	13	16	33	32	10 *U*	10 *U*	100 *U*	10	61	90	220	230

TABLE 7-29. (cont.)

Station	Pyrene				Benz[a]-anthracene				Chrysene				Benzo[b]-fluoranthene			
	1994	1995	1996	1997	1994	1995	1996	1997	1994	1995	1996	1997	1994	1995	1996	1997
Upstream																
18	55	66	8	23	21	78	3	10 *U*	24	98	4	12	20	70	3	13
Facility																
5	ND	220	230	440	ND	150	160	92	ND	190	130	93	ND	96	95	20 *U*
4	520	1,100	830	1,800	240	480	350	660	340	680	410	480	210	440	240	530
3	580	800	1,400	1,200	260	460	480	510	280	660	450	540	200	320	220	670
2	130	520	320	420	53	240	110	160	66	220	130	140	51	100 *U*	79	190
Downstream																
13	75	200	170	260	28	62	77	97	38	170	100	130	33	100	62	150
11	72	180	230	150	31	46	160	58	36	98	100	49	29	66	69	77
Offshore																
16	ND	160	190	370	ND	58	94	120	ND	76	96	12	ND	48	50	81
27	ND	62	220	360	ND	26	110	170	ND	36	140	190	ND	52	82	260
Cannery																
25	510	720	1,500	830	320	400	990	670	380	520	1,300	590	370	360	690	740
Outer Cove																
23	170	520	1,200	430	91	260	790	210	110	320	950	270	120	200	510	270
Tongass Narrows																
22	60	86	200	240	28	48	100	120	35	50	110	150	28	38	58	150

TABLE 7-29. (cont.)

Station	Benzo[k]-fluoranthene				Benzo[a]-pyrene				Indeno [1,2,3-cd]-pyrene				Dibenz[a,h]-anthracene			
	1994	1995	1996	1997	1994	1995	1996	1997	1994	1995	1996	1997	1994	1995	1996	1997
Upstream																
18	13	48	10 *U*	10 *U*	15	48	10 *U*	10 *U*	13	32	1	10 *U*	10 *U*	10 *U*	10 *U*	10 *U*
Facility																
5	ND	66	61	20 *U*	ND	62	65	32	ND	32	36	20 *U*	ND	10 *U*	100 *U*	20 *U*
4	120	240	170	180	130	260	170	190	81	160	140	20 *U*	18 *U*	50	39	20 *U*
3	130	240	150	68	140	220	220	240	89	110	110	120	24	26	22	20 *U*
2	31	100 *U*	52		37	100 *U*	56	80	31	100 *U*	40	40	19 *U*	100 *U*	100 *U*	20 *U*
Downstream																
13	25 *U*	62	48	49	25 *U*	70	46	63	28	54	33	34	25 *U*	16	100 *U*	20 *U*
11	22	40	51	26	25	48	67	45	25	38	51	24	22 *U*	12	100 *U*	20 *U*
Offshore																
16	ND	32	36	31	ND	30	40	42	ND	22	25	23	ND	10 *U*	6	20 *U*
27	ND	38	54	86	ND	34	60	140	ND	24	46	72	ND	10 *U*	100 *U*	20 *U*
Cannery																
25	270	240	530	250	360	260	750	390	230	200	520	230	51	64	73	20 *U*
Outer Cove																
23	81	170	440	87	86	180	620	170	57	130	350	120	15	30	49	22
Tongass Narrows																
22	24	38	72	49	29	38	63	100	18	28	37	75	10 *U*	10 *U*	100 *U*	14

TABLE 7-29. (cont.)

Station	Benzo[ghi]-perylene				Carcinogenic PAH Relative Potency Concentration			
	1994	1995	1996	1997	1994	1995	1996	1997
Upstream								
18	13	32	1	10 *U*	26	72	11	12
Facility								
5	ND	10 *U*	19	41	ND	96	145	53
4	69	130	90	64	194	421	284	322
3	91	96	79	61	220	338	325	381
2	35	100 *U*	19	91	60	135	130	129
Downstream								
13	48	48	30	50	34	108	114	102
11	22 *U*	32	31	20 *U*	45	75	146	71
Offshore								
16	ND	26	16	20 *U*	ND	48	63	75
27	ND	38	30	64	ND	50	134	201
Cannery								
25	200	170	290	160	506	423	1,050	567
Outer Cove								
23	52	120	250	85	129	271	839	253
Tongass Narrows								
22	19	28	32	63	42	55	133	149

Note: All concentrations reported as µg/kg dry weight.

Data shown represent NPDES stations only. Data for 1994 and 1995 are from ENSR (1994, 1995b). Data for 1996 and 1997 are a subset of the data from the present investigation for corresponding NPDES stations. Complete data from the present investigation can be found in Appendix A.

ND - no data
NPDES - National Pollutant Discharge Elimination System
PAH - polycyclic aromatic hydrocarbon
U - undetected at concentration listed

- **AVS:** The concentrations of AVS at Stations 16 and 22 (680–17,000 mg/kg) were considerably greater than the respective historical values (62–6,200 mg/kg)
- **Total Sulfide:** The concentrations of total sulfide at Stations 16, 22, and 27 (560–16,000 mg/kg) were considerably greater than the respective historical values (20–1,800 mg/kg)
- **BOD:** The BOD levels at Stations 2, 3, 4, 22, 23, 25, and 27 (3,500–64,000 mg/kg) were considerably greater than historical values (130–17,000 mg/kg)
- **Phenol:** The concentration of phenol at Station 5 (909 μg/kg) was considerably greater than the respective historical value (106 μg/kg)
- **4-Methylphenol:** The concentrations of 4-methylphenol at Stations 3, 5, 16, and 25 (1,200–16,000 μg/kg) were considerably greater than historical values (430–2,100 μg/kg)
- **Benzoic Acid:** The concentrations of benzoic acid at Station 4 (870–1,600 μg/kg) were considerably greater than historical values (26–480 μg/kg).

With the exception of the concentrations identified above, the results of the comparison of chemical concentrations among samples collected in 1994, 1995, 1996, and 1997 at the subset of 12 NPDES stations suggest that the top 2 cm of the sediment column is generally representative of the top 10 cm and that large subsurface increases in chemical concentrations are not found throughout most of Ward Cove to a depth of 10 cm.

7.1.3.2 Sediment Toxicity

The comparisons of sediment toxicity results among years at the 12 NPDES stations in Ward Cove are presented in Table 7-30 and Figure 7-12. The toxicity tests evaluated in all years were the amphipod test based on *Rhepoxynius abronius* and the echinoderm test based on the sand dollar *Dendraster excentricus* in 1996 and 1997 and the purple sea urchin *Strongylocentrotus purpuratus* in 1994 and 1995. The following major patterns were observed:

- Values of mean survival of amphipods and mean normal survival of echinoderm embryos in 1995, 1996, and 1997 generally were greater than the corresponding values observed during 1994, suggesting that the 1994 results are not a valid indication of sediment toxicity throughout Ward Cove. Because oxygenation of the echinoderm test chambers was improved in 1995, 1996, and 1997 (relative to 1994), that factor may have been partly responsible for the lower values of survival found in 1994. Because the toxicity laboratory changed the dissolved oxygen meters between Day 2 and Day 3 during the 1994

TABLE 7-30. SEDIMENT TOXICITY RESULTS IN WARD COVE BETWEEN 1994 AND 1997[a]

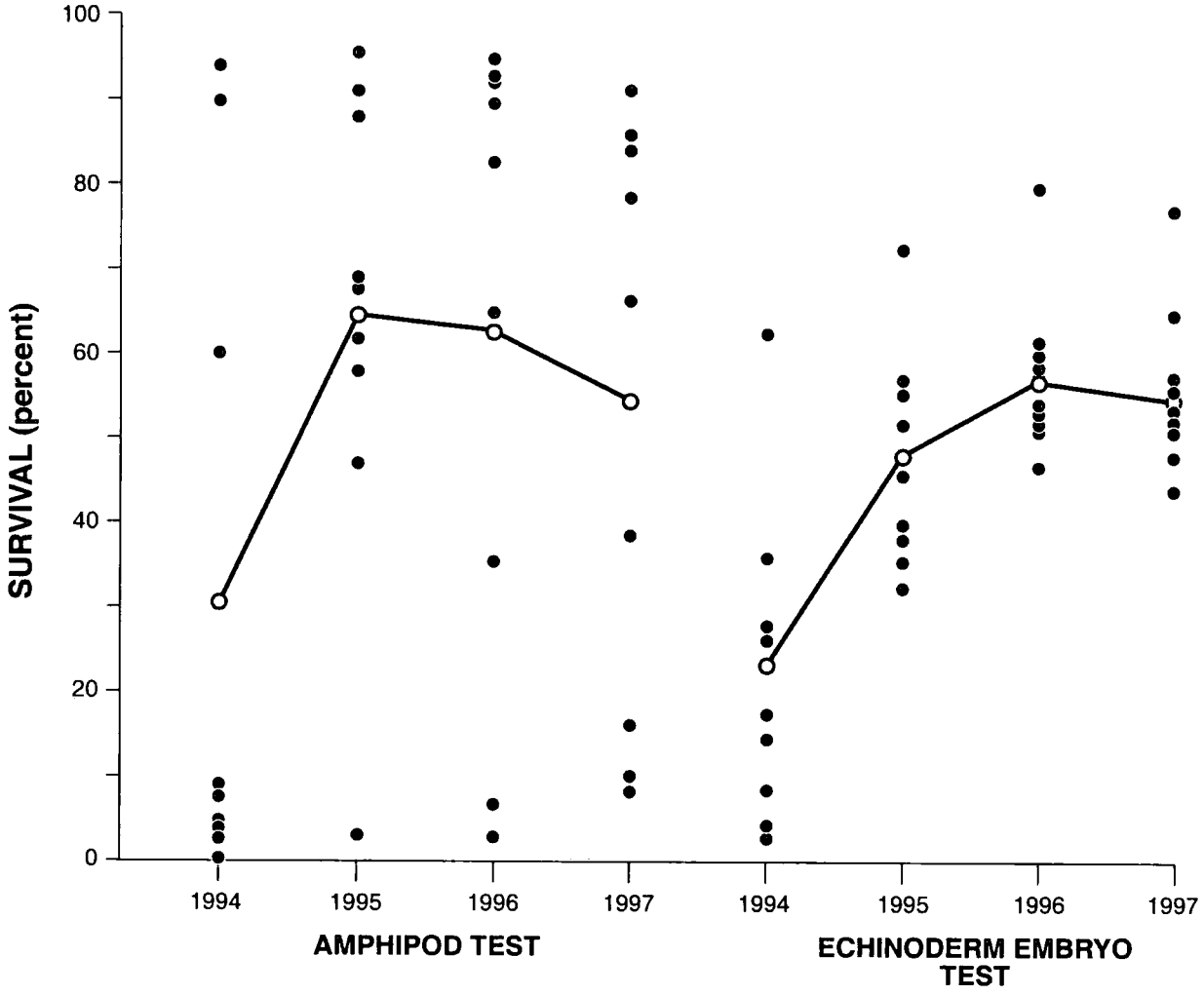
	Amphipod Survival[b] (percent)				Echinoderm Embryo Normal Survival[c] (percent)			
Station	1994	1995	1996	1997	1994	1995	1996	1997
Upstream								
18	60	88	95	90	15	57	58	50
Facility								
5	ND	75	25	39	ND	34	48	53
4	0	47	64	38	9	36	56	56
3	9	62	90	65	3	33	51	53
2	2	3	7	9	18	38	55	43
Downstream								
13	8	68	36	15	36	48	52	48
11	5	69	94	83	28	56	47	55
Offshore								
16	ND	92	30	89	ND	30	52	32
27	ND	86	85	75	ND	41	72	38
Cannery								
25	2	58	3	10	27	40	58	56
Outer Cove								
23	90	91	94	79	5	52	59	63
Tongass Narrows								
22	94	96	84	84	63	73	80	78

Note: ND - no data

[a] Results for 1994 and 1995 are based on the top 2 cm of the sediment column, whereas results for 1996 and 1997 are based on the top 10 cm of the sediment column.

[b] The test species was *Rhepoxynius abronius*.

[c] The test species for 1994 and 1995 was *Strongylocentrotus purpuratus;* the test species for 1996 and 1997 was *Dendraster excentricus.*



LEGEND

- ● Value for each of the nine stations evaluated in all three years
- ○ Mean for all nine stations

Note:
The test species for the amphipod test was *Rhepoxynius abronius*. The test species for the echinoderm embryo test was *Strongylocentrotus purpuratus* in 1994 and 1995 and *Dendraster excentricus* in 1996 and 1997

Figure 7-12. Comparison of sediment toxicity results in Ward Cove between 1994 and 1997.

echinoderm tests, it is not known how low the dissolved oxygen concentrations dropped. However, by Day 2, all concentrations for the test sediments had declined to below 6 mg/L.

- In general, the lowest values of survival for both toxicity tests were consistently found directly offshore from the KPC facility on the northern shoreline of Ward Cove and offshore from the cannery on the southern shoreline of the Cove.

- Mean amphipod survival throughout Ward Cove was similar between 1995, 1996, and 1997 (Figure 7-12), indicating that there were no major Cove-wide differences in toxicity between the top 2 cm of sediment (1995 data) and the top 10 cm of sediment (1996 and 1997 data). However, survival at several individual stations (Stations 5, 13, and 25) was considerably lower for the top 10 cm, relative to the top 2 cm.

- Mean normal survival of echinoderm embryos throughout Ward Cove was greater in 1996 and 1997 than in 1995 (Figure 7-12), indicating that there was no Cove-wide increase in sediment toxicity for the top 10 cm of sediment relative to the top 2 cm. In addition, normal survival in 1996 and 1997 at most stations was greater than the station-specific values found in 1995. The comparison of echinoderm embryo results among years is somewhat confounded by the use of different test species during each year. As stated in the technical studies work plan (PTI 1996), the sand dollar (*Dendraster excentricus)* was substituted for the purple sea urchin (*Strongylocentrotus purpuratus*) in 1996 and 1997 because of the concern that the fine-grained nature of sediments in parts of Ward Cove may have influenced the toxicity tests based on the sea urchin in 1994 and 1995.

The patterns described above for the subset of 12 NPDES stations indicate that, in general, there is no substantial Cove-wide increase in sediment toxicity for subsurface sediments to a depth of 10 cm, relative to the toxicity observed for the top 2 cm. However, subsurface sediments in several localized areas of the Cove may be more toxic than surface sediments based on the results of the amphipod test.

7.1.4 Results of the Specialized Toxicity Tests

In this section, results of the four kinds of specialized toxicity tests conducted during Phase 2 in 1997 are described. As noted in Section 2.3, these procedures were conducted primarily to evaluate the potential roles of ammonia and sulfide in causing sediment toxicity at eight representative stations from three subareas of Ward Cove. The four procedures included:

- **Sediment purging procedure:** applied to all eight stations using the amphipod *Rhepoxynius abronius* as the test species
- **Sediment *Ulva* procedure:** applied to all eight stations using *R. abronius* as the test species
- **Porewater *Ulva* procedure:** applied to all eight stations using *R. abronius* as the test species and to three stations (one from each subarea) using the echinoderm *Dendraster excentricus* as the test species
- **Porewater aeration procedure:** applied to all eight stations using *R. abronius* as the test species.

The results of the specialized toxicity tests are described below.

7.1.4.1 Sediment Purging Procedure

The sediment purging procedures were based on the methods specified by U.S. EPA (1994e), in which each sediment sample is purged until porewater ammonia concentrations are lower than species-specific no-effect concentrations. For *Rhepoxynius abronius*, the no-effect concentration was identified as 30 mg/L (U.S. EPA 1994e). Toxicity testing is then initiated immediately after porewater ammonia concentrations are reduced to the target levels.

As a result of a misunderstanding by the analytical laboratory, purging of all sediment samples was conducted for 10 days, despite the fact that porewater ammonia concentrations after the first day of purging in all samples ranged from 4.0 to 16 mg/L (Table 7-31), which is considerably less than the no-effect concentration of 30 mg/L for *Rhepoxynius abronius*. The laboratory continued purging all samples because elevated porewater sulfide concentrations persisted in all samples. Despite the departure from the protocols specified by U.S. EPA (1994e), the resulting information is considered useful because it provides relevant information on the effects of porewater ammonia and sulfide on amphipod toxicity at the concentrations present after purging was completed.

Because ammonia concentrations after purging and immediately prior to test initiation (i.e., Day 9) ranged from 0.6 to 6.0 mg/L (Table 7-31), all concentrations were considerably lower than the no-effect concentration of 30 mg/L for *Rhepoxynius abronius* and the 96-hour LC50 of 79 mg/L identified for *R. abronius* by Kohn et al. (1994). It therefore is unlikely that ammonia was responsible for any observed amphipod toxicity following initiation of testing on Day 10. However, the use of the 96-hour LC50 for ammonia to evaluate potential toxicity to *R. abronius* in the present study should be qualified by the fact that the exposure period of 10 days was 2.5 times longer than the exposure period used by Kohn et al. (1994). It therefore is possible that the test organisms responded to ammonia concentrations lower than the 96-hour LC50 as a result of the longer exposure period.

TABLE 7-31. RESULTS OF SEDIMENT PURGING TESTS USING *Rhepoxynius abronius*[a]

Station	Porewater Ammonia (mg/L)				Porewater Sulfide (mg/L)				Amphipod Survival (percent)	
	Day 2	Day 5	Day 9	Day 17	Day 2	Day 5	Day 9	Day 17	Unpurged	Purged
Subarea 1										
12	14	4.5	5.5	4.5	36	14	23	2.5 *U*	14 (11.9)	55 (23.2)
13	10	6.5	2.0	2.0	39	36	14	2.8	15 (22.6)	49 (25.3)
44	16	6.0	6.0	6.0	35	30	23	3.0	1 (2.2)	25 (29.2)
Subarea 2										
16	4.0	2.0	1.0	0.5 *U*	3.8	2.5 *U*	2.5 *U*	2.5 *U*	89 (4.2)	86 (11.9)
17	8.5	2.5	2.0	0.5 *U*	21	11	6.3	2.5 *U*	43 (39.9)	72 (23.9)
35	6.2	4.0	1.0	1.0	26	17	14	5.5	75 (17.0)	39 (29.5)
Subarea 3										
7	5.5	2.5	0.5	1.0	18	7.5	2.5 *U*	2.5 *U*	58 (15.7)	87 (16.4)
34	9.5	2.5	2.0	2.0	39	18	10	3.8	39 (10.3)	66 (30.3)

Note: Standard deviations listed in parentheses.
U - undetected at the concentration listed

[a] Sediment was loaded into test chamber on Day 0, purging began on Day 1, test initiation with amphipods began on Day 10, and test termination occurred on Day 20.

By contrast with ammonia, concentrations of sulfide in pore water immediately prior to test initiation in six of the eight sediment samples (6.3–23 mg/L; Table 7-31) were considerably greater than the 48-hour LC50 of 1.6 mg/L identified for *R. abronius* by Knezovich et al. (1996). Sulfide was therefore a potential contributor to any amphipod toxicity observed after test initiation for those samples. Similar to that for ammonia, the 48-hour LC50 value for sulfide should be qualified for several reasons. First, a longer exposure period was used in the present study, which would tend to result in the organisms responding to lower sulfide concentrations than the LC50. Second, because sulfide toxicity is sensitive to the pH of the pore water (i.e., it increases with decreasing pH), sulfide toxicity in the present study may have been influenced by the wider range of pH values (i.e., 8.2±0.3) than those tested by Knezovich et al. (1996) (i.e., 8.0±0.1). Finally, because toxicity testing was conducted under static conditions in the present study and under flow-through conditions by Knezovich and coworkers (in which sulfide concentrations were maintained at constant levels), sulfide toxicity could be less in the present study because it would have gradually declined over the 10-day exposure period (Knezovich 1998, pers. comm.).

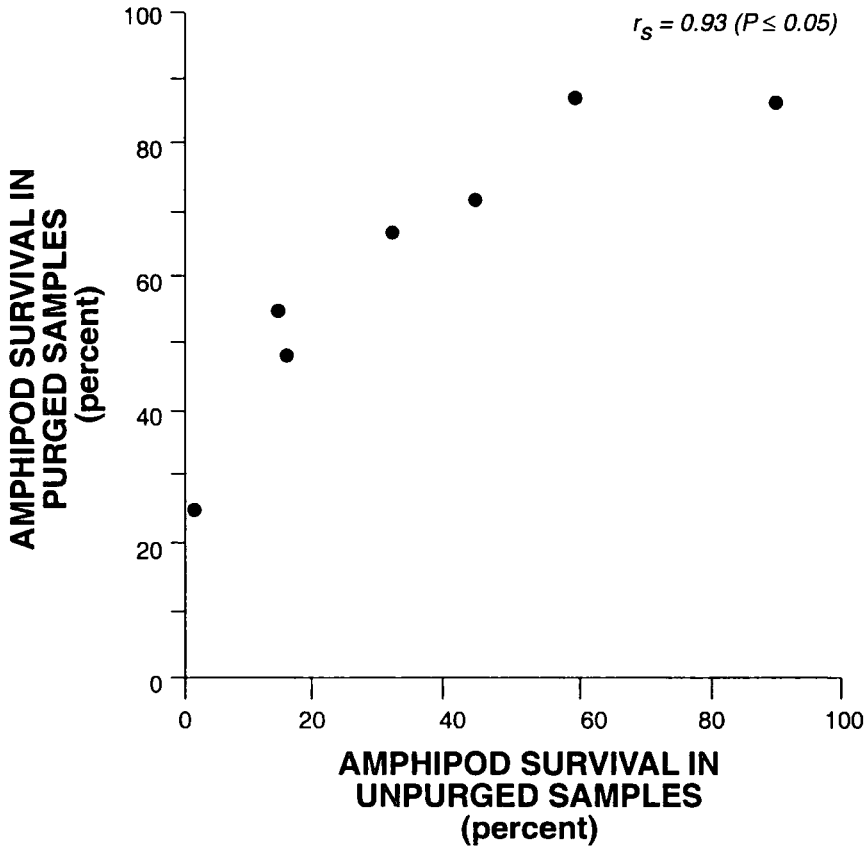
The detailed results of the sediment purging procedures for the eight stations from three subareas of Ward Cove are presented in Table 7-31. In general, purging reduced sediment toxicity by a similar degree in most samples, so that the rank order of toxicity was significantly correlated (r_s = 0.93; $P \leq 0.05$; Spearman rank correlation coefficient) between purged and unpurged samples (Figure 7-13).

For six of the eight sediment stations (Stations 7, 12, 13, 17, 34, and 44), amphipod survival increased considerably following sediment purging. The increase in survival at the six stations ranged from 24 percent (Station 44) to 41 percent (Station 1) and averaged 31 percent.

By contrast with results for the six stations described above, amphipod survival at Station 16 was very high initially (89 percent) and declined by only 3 percent after purging. However, given the degree of variability in survival for the two treatments (standard deviations = 4 and 12 percent), it is unlikely that the 3 percent decrease in survival after purging is meaningful, and survival can be considered to have stayed relatively constant following the purging treatment. This result would be expected for a sample that is initially nontoxic.

The purging results for Station 35 were difficult to interpret because amphipod survival declined from 75 percent in unpurged samples to 39 percent following purging. Because there is no apparent reason why toxicity should increase following purging, the results for this station are considered anomalous and are not discussed further.

Porewater concentrations of both ammonia and sulfide declined substantially following purging in all six of the samples in which amphipod survival increased as the result of purging. Ammonia concentrations in samples after 1 day of purging (Day 2) ranged from 5.5 to 16 mg/L and declined by an average of 75 percent to a range of 0.6–6.0 mg/L by



r_S = *Spearman rank correlation coefficient*

Figure 7-13. Comparison of amphipod (*Rhepoxynius abronius*) survival in unpurged and purged sediment samples.

the end of the purging period (Day 9). Ammonia concentrations then remained relatively constant until the end of the 10-day test period (Day 17). Because all of the ammonia concentrations in the purged samples were considerably lower than the 96-hour LC50 of 79 mg/L for *Rhepoxynius abronius* (Kohn et al. 1994), it is questionable whether ammonia was a major cause of the observed toxicity, regardless of an exposure period (10 days) that was 2.5 times greater than one on which the LC50 was determined.

Sulfide concentrations in the six samples after 1 day of purging (Day 2) ranged from 18 to 39 mg/L and declined by an average of 62 percent to a range of 2.5 *U* to 23 mg/L by the end of the purging period (Day 9). Sulfide concentrations then continued to decline by an average of 67 percent to a range of 2.5 *U* to 3.8 mg/L by the end of the 10-day test period (Day 17). Because most of the sulfide concentrations measured in both unpurged and purged samples exceeded the 48-hour LC50 of 1.4 mg/L for *Rhepoxynius abronius* (Knezovich et al. 1996), sulfide may have played a role in causing the observed toxicity in most samples. Because the detection limit for sulfide (i.e., 2.5 mg/L) was greater than the 48-hour LC50, it is possible that sulfide concentrations exceeded the LC50 in all samples and therefore potentially contributed to toxicity in all samples.

The potential roles of ammonia and sulfide in causing the observed amphipod toxicity in the sediment purging procedure were evaluated further by comparing concentrations of each of these variables with amphipod survival using the Spearman rank correlation coefficient (r_s) (Figure 7-14). Initial concentrations were used for the correlations because much of the toxicity found during the 10-day exposure period could have occurred in the first few days of exposure, especially for volatile substances such as sulfide (Knezovich et al. 1996).

Despite the fact that ammonia concentrations were low relative to the 96-hour LC50 for *Rhepoxynius abronius*, significant ($P \leq 0.05$) negative correlations were found between amphipod survival and porewater ammonia concentrations for both unpurged and purged samples (Figure 7-14). For sulfide, only the correlation with amphipod survival in purged sediment was significant ($P \leq 0.05$; Figure 7-14). However, the negative relationship between those two variables for unpurged samples was relatively strong ($r_s = -0.65$; $P \leq 0.07$). Because amphipod survival was strongly related to concentrations of both ammonia and sulfide in unpurged and purged samples, the correlation analysis cannot conclusively identify the relative contribution of each substance to the observed toxicity. However, as discussed above, it is unlikely that ammonia contributed to any observed amphipod toxicity because porewater concentrations in all samples were considerably less than the no-effect concentration and 96-hour LC50 for *R. abronius*.

In summary, sediment purging resulted in a considerable increase in amphipod survival for most samples. Although both ammonia and sulfide are implicated as potential causative factors for the observed toxicity, only sulfide was present in pore water at concentrations high enough to have potentially played a role in causing the observed toxicity. Because purging did not remove all toxicity in some samples, it is possible that chemicals other than sulfide also contributed to the observed toxicity.

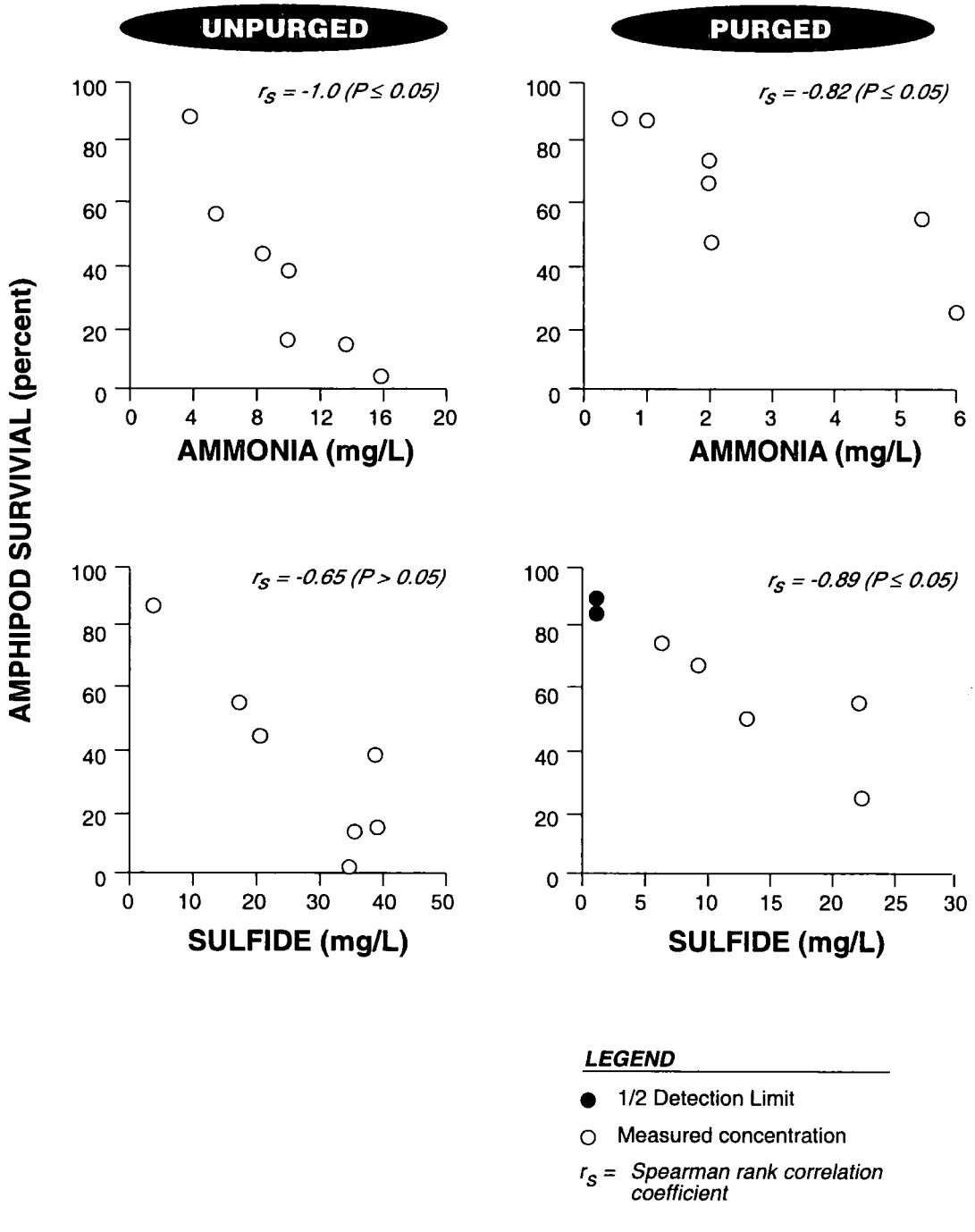


Figure 7-14. Comparison of amphipod (*Rhepoxynius abronius*) survival and ammonia and sulfide concentrations in unpurged and purged sediment samples.

7.1.4.2 Sediment *Ulva* Procedure

The detailed results of the sediment *Ulva* procedure are presented in Table 7-32. Because amphipod survival was very high (90–100 percent) in all eight of the untreated samples, the subsequent treatment with *Ulva* had little meaning with respect to reducing the toxicity of the test sediments. The reason for the observed lack of toxicity is unknown. This procedure can therefore not be used to directly assess the roles of ammonia and sulfide in causing toxicity in the test sediments.

Although the sediment *Ulva* procedure cannot be used to directly evaluate the roles of ammonia and sulfide in causing sediment toxicity, it is noteworthy that the porewater ammonia concentrations for Stations 12, 13, and 44 (7.5–12 mg/L) were not toxic to the amphipods (i.e., survival = 90–100 percent). Although this finding would be expected because these values are considerably lower than the 96-hour LC50 of 79 mg/L for *Rhepoxynius abronius* (Kohn et al. 1994), these values are similar to most of the higher ammonia concentrations found in the unpurged samples used for the sediment purging procedure (Table 7-31). This further supports the suggestion made in the previous section that ammonia did not appear to be a major cause of toxicity in the unpurged and purged samples.

Because none of the sulfide concentrations measured in the sediment *Ulva* procedure were substantially greater than the 48-hour LC50 of 1.4 mg/L for *Rhepoxynius abronius* (Knezovich et al. 1996), the lack of observed toxicity in all of the samples subjected to this procedure is consistent with the suggestion made in the previous section that sulfide may have been a major cause of toxicity in the unpurged and purged samples. Furthermore, because sulfide is very unstable, the primary reason that the sediment *Ulva* procedure did not work in the present study may have been that sulfide in the pore water of the test sediments was oxidized during sample handling and test setup. This potential experimental artifact would be more likely to occur for the *Ulva* procedure than the purging procedure, because the *Ulva* procedure uses a much smaller amount of test sediment (20 g) than the purging procedure (200 g).

7.1.4.3 Porewater *Ulva* Procedure

The detailed results of the porewater *Ulva* procedure using *Rhepoxynius abronius* are presented in Table 7-33. As shown in Figure 7-15, the correlation between ammonia and sulfide concentrations in pore water was significant (r_s = 0.92; $P \leq 0.05$; Spearman rank correlation coefficient) for the eight sediment samples. This strong covariance between the two variables makes it difficult to determine the independent effects of each variable in causing toxicity.

The treatment of pore water with *Ulva* considerably reduced the toxicity of all samples except Station 16, which was nontoxic to begin with (Figure 7-16). The initial toxic units of all samples except Station 16 ranged from 3.0 to 10 and declined to a range of less than 1.0 to 2.7 after the *Ulva* treatment. Although three of those samples became

TABLE 7-32. RESULTS OF SEDIMENT *Ulva* TESTS USING *Rhepoxynius abronius*

	Ammonia (mg/L)				Sulfide (mg/L)				Amphipod Survival (percent)	
	Untreated		*Ulva* Treated		Untreated		*Ulva* Treated			*Ulva*
Station	Day 0	Day 2	Day 0	Day 2	Day 0	Day 2	Day 0	Day 2	Untreated	Treated
Subarea 1										
12	9.5	10	0.5	2.0	1.9	0.5 *U*	2.5 *U*	0.5 *U*	90	96
13	7.5	8.0	0.5	0.5 *U*	3.1	0.5 *U*	2.5 *U*	0.5 *U*	100	100
44	12	12	0.5	0.5	5.3	0.5 *U*	2.5 *U*	0.5 *U*	100	100
Subarea 2										
16	2.0	1.5	0.5 *U*	0.5 *U*	2.5 *U*	0.5 *U*	2.5 *U*	0.5 *U*	100	100
17	4.0	3.0	0.5	0.5 *U*	2.5 *U*	0.5 *U*	2.5 *U*	0.5 *U*	100	96
35	2.5	2.5	0.5 *U*	0.5 *U*	0.6 *U*	0.5 *U*	2.5 *U*	0.5 *U*	90	100
Subarea 3										
7	3.0	3.0	0.5 *U*	0.5 *U*	2.5 *U*	0.5 *U*	2.5 *U*	0.5 *U*	100	100
34	4.0	3.0	0.5 *U*	0.5 *U*	2.5 *U*	0.5 *U*	2.5 *U*	0.5 *U*	100	96

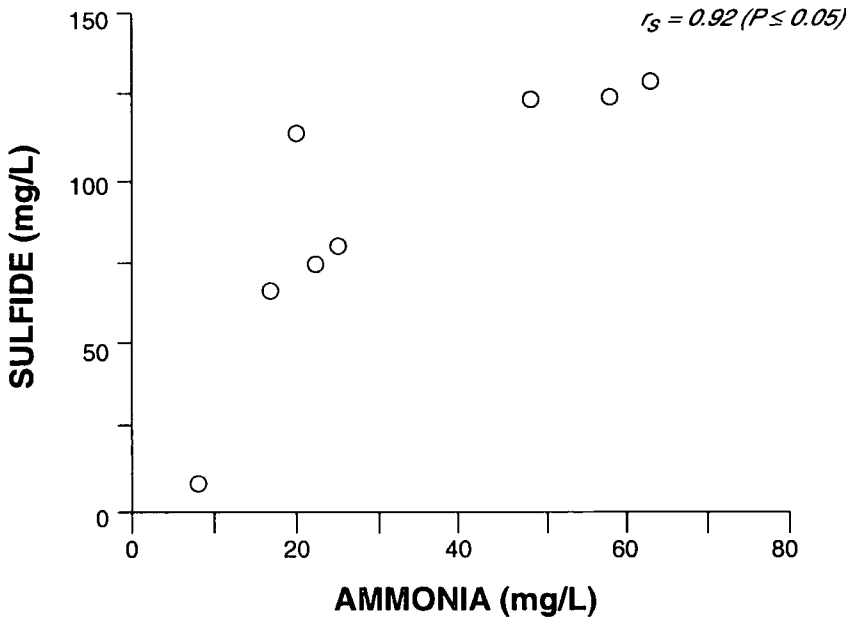
Note: *U* - undetected at the concentration listed

TABLE 7-33. RESULTS OF POREWATER TESTS USING *Rhepoxynius abronius*

	Ammonia (mg/L)			Sulfide (mg/L)			LC50 (percent pore water)			Toxic Units[a]		
Station	B	A	U	B	A	U	B	A	U	B	A	U
Subarea 1												
12	58	58	33	125	11	60	10	>100	37	10	<1.0	2.7
13	48	43	27	125	7.5	65	11	>100	51	9.3	<1.0	2.0
44	63	60	37	130	11	58	17	40	40	6.0	2.5	2.5
Subarea 2												
16	7.5	7.5	1.7 *U*	10 *U*	2.5 *U*	5.0 *U*	>100	>100	>100	<1.0	<1.0	<1.0
17	25	20	1.7	80	5.0	15	11	>100	>100	9.4	<1.0	<1.0
35	23	20	6.7	75	2.5	30	20	>100	100	5.1	<1.0	<1.0
Subarea 3												
7	18	18	3.3	65	5.0	23	33	>100	>100	3.0	<1.0	<1.0
34	20	23	3.3	115	6.3	50	12	>100	46	8.7	<1.0	2.2

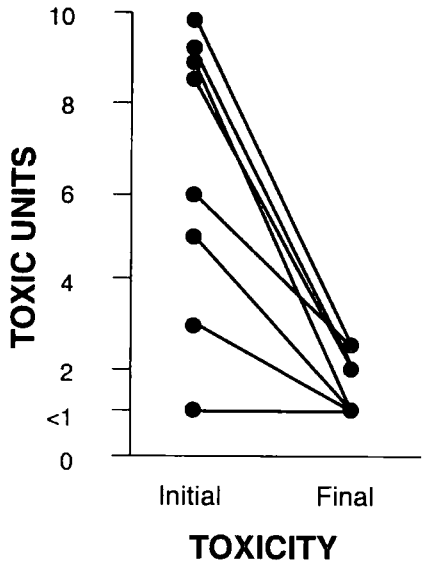
Note: B - baseline conditions
A - results for aeration procedure
U - results for *Ulva* procedure
LC50 - concentration lethal to 50 percent of the test population
U - undetected at concentration listed

[a] Toxic units = 100/LC50.



r_S = *Spearman rank correlation coefficient*

Figure 7-15. Comparison of concentrations of ammonia and sulfide in porewater samples.



Note: Toxic unit = 100/LC50

Figure 7-16. Comparison of initial and final results for sulfide, ammonia, and amphipod (*Rhepoxynius abronius*) survival for the porewater *Ulva* procedure.

nontoxic after the treatment (i.e., toxic units <1.0), toxicity remained in four of the samples (toxic units = 2.2 to 2.7).

Initial porewater ammonia concentrations in all samples but Station 16 ranged from 18 to 63 mg/L and declined an average of 65 percent to a range of 1.7 to 37 mg/L following the *Ulva* treatment. Initial porewater sulfide concentrations in all samples but Station 16 ranged from 65 to 130 mg/L and declined an average of 60 percent to a range of 15 to 65 mg/L following the treatment. All of the final ammonia concentrations were less than half the 96-hour LC50 of 79 mg/L for *Rhepoxynius abronius* (Kohn et al. 1994), whereas all of the final sulfide concentrations were considerably greater than the 48-hour LC50 of 1.6 mg/L for *R. abronius* (Knezovich et al. 1996). Final sulfide concentrations in the four samples having final toxic units of 2.2 to 2.7 were 50 to 65 mg/L. Final sulfide concentrations in all other samples were 30 mg/L or less. These results suggest that sulfide, rather than ammonia, may be responsible for most of the observed toxicity of the samples.

The detailed results of the porewater *Ulva* procedure using *Dendraster excentricus* are presented in Table 7-34. In general, results were similar to those described above for *Rhepoxynius abronius*. Concentrations of both ammonia and sulfide declined after the *Ulva* treatment, as did the toxicity of all three samples. However, despite the decline in ammonia and sulfide concentrations, all three samples remained toxic after the *Ulva* treatment, with toxic units ranging from 6.4 to 60. The relatively high level of residual toxicity may have been due to the fact that *D. excentricus* is very sensitive to sulfide toxicity. For example, Knezovich et al. (1996) determined an EC50 of 0.19 mg/L for larval abnormality for *Strongylocentrotus purpuratus* (an echinoderm similar to *D. excentricus*). That EC50 is approximately 8 times lower than the LC50 of 1.6 mg/L for *Rhepoxynius abronius* and 13 times lower than the detection limit of 2.5 mg/L in the present study.

7.1.4.4 Porewater Aeration Procedure

The detailed results of the porewater aeration procedure using *Rhepoxynius abronius* are presented in Table 7-33. Porewater samples were vigorously aerated for 1 hour prior to test initiation. Porewater aeration considerably reduced the toxicity of all samples except Station 16, which was nontoxic to begin with (Figure 7-17). The initial toxic units of all samples except Station 16 ranged from 3.0 to 10 and declined to a range of less than 1.0 to 2.5 after the aeration treatment. Six of those samples became nontoxic after the treatment (i.e., toxic units <1.0), whereas toxicity remained in one sample (toxic unit = 2.5).

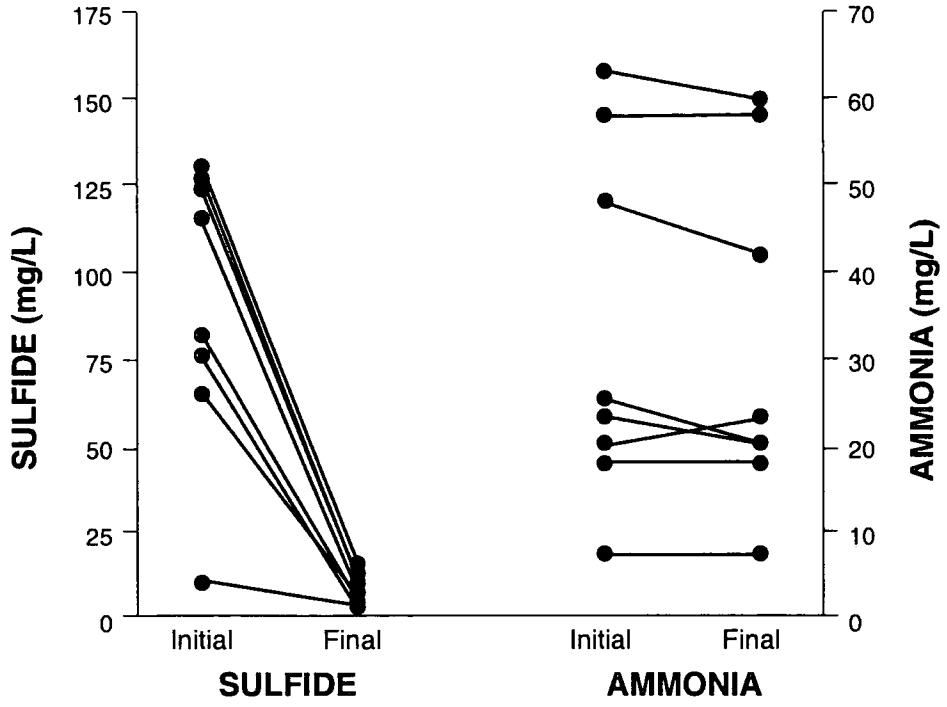
Initial porewater ammonia concentrations in all samples but Station 16 ranged from 18 to 63 mg/L and remained nearly constant following the aeration treatment. By contrast, initial porewater sulfide concentrations in all samples but Station 16 ranged from 65 to 130 mg/L and declined an average of 94 percent to a range of 2.5 to 11 mg/L following the treatment. Because the decline in sample toxicity largely corresponded to the decline in sulfide concentrations, and because ammonia concentrations remained relatively constant, it is likely that sulfide, rather than ammonia, was the major cause of the observed

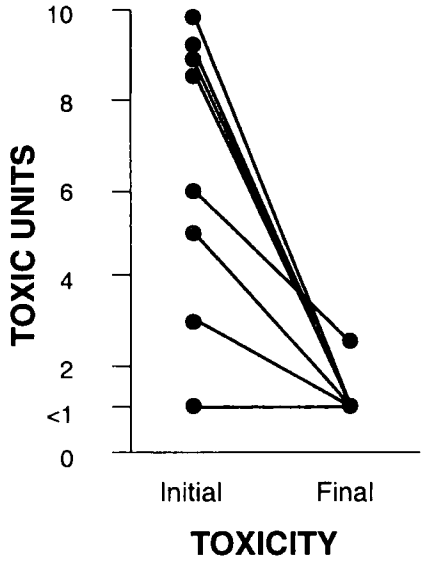
TABLE 7-34. RESULTS OF POREWATER TESTS USING *Dendraster excentricus*

Station	Ammonia (mg/L)			Sulfide (mg/L)			EC50 (percent pore water)			Toxic Units[a]		
	B	A	U	B	A	U	B	A	U	B	A	U
Subarea 1												
12	22	17	8.0	56	2.5 *U*	2.5 *U*	0.24	3.8	4.4	420	27	23
13	14	12	4.0	43	2.5 *U*	2.5 *U*	0.40	4.5	16	250	22	6.4
44	22	20	16	44	2.5 *U*	18	0.34	2.9	1.7	290	35	60

Note: B - baseline conditions
A - results for aeration procedure
U - results for *Ulva* procedure
EC50 - concentration effective in 50 percent of the test population
U - undetected at concentration listed

[a] Toxic units = 100/EC50





Note: Toxic unit = 100/LC50

Figure 7-17. Comparison of initial and final results for sulfide, ammonia, and amphipod (*Rhepoxynius abronius*) survival for the porewater aeration procedure.

toxicity. However, ammonia concentrations from stations in Subarea 1 (i.e., Stations 12, 13, and 44) may have been sufficiently elevated to have contributed to the observed toxicity.

The suggestion that sulfide might be the major cause of the observed toxicity is supported further by the fact that none of the final sulfide concentrations in the aerated test chambers were considerably greater than the 48-hour LC50 of 1.6 mg/L for *R. abronius* (Knezovich et al. 1996). As discussed earlier, the fact that the final sulfide concentrations exceeded the 96-hour LC50 identified by Knezovich et al. (1996) does not necessarily mean that toxicity would be expected. Difference in porewater pH and exposure conditions between the two studies could have resulted in amphipods being less sensitive to sulfide toxicity than would be predicted based on the results of Knezovich et al. (1996).

The detailed results of the porewater aeration procedure using *Dendraster excentricus* are presented in Table 7-34. In general, results were similar to those described above for *Rhepoxynius abronius*. Concentrations of ammonia remained relatively constant following aeration, whereas sulfide concentrations declined to below the detection limit. However, despite the decline in sulfide concentrations, all three samples remained toxic after aeration, with toxic units ranging from 6.4 to 60.

7.1.4.5 Consideration of Selected Chemicals as Toxic Agents

Although the primary chemicals evaluated during the aeration procedure (as well as the other specialized toxicity tests) were ammonia and sulfide, it is possible that other chemicals such as 4-methylphenol and other components of wood leachate may have been responsible for the observed toxicity. However, only sulfide has sufficient volatility and oxidizes rapidly enough to account for the change in toxicity observed for Ward Cove sediments after the aeration procedure, which was conducted at an approximate rate of 4 L/minute for 1 hour (Caldwell 1998, pers. comm.). Each of these chemicals is discussed in detail below.

Sulfide—The toxicity of pulp mill and wood waste has been shown by many researchers to be due primarily to the release of hydrogen sulfide (Leach and Thakore 1973; Waldichuk 1988; Sedell et al. 1991). Hydrogen sulfide rapidly volatilizes independent of pH (Morel and Hering 1993). In addition, hydrogen sulfide is unstable in the presence of oxygen and rapidly oxidizes to sulfate. Oxidation rates are faster at higher pH values (Zhang and Millero 1994). The instability of hydrogen sulfide in the presence of oxygen was demonstrated by the aeration procedure conducted during the present study, in which hydrogen sulfide concentrations decreased substantially following sample aeration for 1 hour.

4-Methylphenol—Although 4-methylphenol is present in Ward Cove sediment, it is unlikely to be removed significantly from pore water by aeration. The aerobic

half-life for biodegradation of 4-methylphenol ranges from 1 to 43 hours in the presence of acclimated bacteria (Howard et al. 1991). Given that the Ward Cove sediments were largely anaerobic before the aeration procedure began, the bacteria responsible for aerobic degradation of 4-methylphenol would not be expected to be present. Therefore, the degradation rate of 4-methylphenol during the aeration procedure is probably closer to the maximum measured degradation period of 43 hours, and degradation during the 1-hour aeration procedure was probably minimal.

In addition to its relatively slow rate of biodegradation, 4-methylphenol is not highly volatile. The amount of 4-methylphenol that could be expected to volatilize into the air stream passing through the sediment during the present study was calculated to be only 0.25 percent of the total amount of 4-methylphenol present in the sediment. Therefore, it is unlikely that the observed reduction in porewater toxicity following aeration was related to 4-methylphenol.

Other Constituents of Wood Waste—Although sulfide is the primary toxic compound found in wood waste, leachates from wood and bark are also known to be toxic to aquatic organisms (Buchanan et al. 1976). The primary toxic compounds in wood leachate have been found to be the conjugate bases of resin acids, also known as resin acid soaps (Leach and Thakore 1973) These acids make up about 0.25 percent of softwood bark (Fengel and Wegener 1989). Leach and Thakore (1973) found that the resin acid soap isopimarate accounted for 55 percent of observed toxicity in kraft mill effluent, with abietate and dehydroabietate contributing most of the rest of the toxicity. The remaining toxicity was contributed by fatty acid conjugate bases such as oleate, linoleate, and linolenate (the acids themselves were not toxic, only the conjugate bases). The calcium and magnesium salts of these compounds are relatively insoluble (Loudon 1984), which may explain why wood leachate is less toxic in seawater than in fresh water (Pease 1974). These compounds are highly recalcitrant (Wilson et al. 1996) and are not likely to be degraded quickly. Furthermore, these compounds would be fully deprotonated at the neutral pH values present in marine sediments (Mead et al. 1986), and because ionic compounds have considerably lower volatilities than non-ionic compounds (Huheey 1983), they would not be expected to volatilize quickly. It therefore is unlikely that the observed reduction in porewater toxicity following aeration was related to these other constituents of wood waste.

7.1.5 Summary

The major results of the sediment toxicity assessment for Ward Cove sediments in 1996 and 1997 can be summarized as follows:

- Sediment toxicity and sediment chemical concentrations were evaluated at 44 stations throughout Ward Cove and at 2 stations in Moser Bay, a nearby reference area.

- Sediment toxicity was found in only two of the four toxicity tests used to evaluate Ward Cove sediments: the amphipod test based on *Rhepoxynius abronius* and the echinoderm embryo test based on *Dendraster excentricus* (i.e., the normal survival endpoint). The response range for the *R. abronius* test was very broad, ranging from 0 to 96 percent survival. By contrast, the response range for the echinoderm embryo test was narrower, with most values ranging from 30 to 80 percent normal survival.

- Sediment toxicity was not found in 1996 in either the amphipod test based on *Leptocheirus plumulosus* or the juvenile polychaete test based on *Neanthes* sp. Survival at all stations was very high for the *L. plumulosus* test, ranging from 89 to 100 percent. Individual growth rate at all stations for the juvenile polychaete was also high (0.51–0.74 mg/day), relative to mean individual growth rate in the reference area (0.60 mg/day).

- Sediment toxicity was not found at most stations based on the normality endpoint of the echinoderm embryo test. Normality ranged from 74 to 98 percent in 1996 and from 90 to 99 percent in 1997 and was ≥90 percent (i.e., the minimum allowable value for acceptable negative controls) in only 14 of 61 cases for the two years.

- Most stations at which sediment toxicity was found were located offshore from the KPC facility and downcurrent from the facility along the northern shoreline of Ward Cove.

- The results of the four sediment toxicity tests were used to develop site-specific sediment quality values (WCSQVs) for major CoPCs, including TOC, ammonia, BOD, COD, and 4-methylphenol. The site-specific sediment quality values were developed using the AET approach. The $WCSQV_{(1)}$ for each CoPC was based on the lowest AET for the four toxicity tests, whereas the $WCSQV_{(2)}$ was represented by the second lowest AET for the four tests.

- Stations at which CoPCs exceeded their respective site-specific sediment quality values generally were located offshore from the KPC facility and downcurrent from the facility along the northern shoreline of Ward Cove. Exceedances of site-specific sediment quality values were also found offshore from the fish cannery on the southern shoreline of the Cove.

- Most exceedances of $WCSQV_{(2)}$ values at the 44 stations sampled in Ward Cove were found for ammonia (13 stations) and 4-methylphenol (18 stations). By contrast, exceedances of $WCSQV_{(2)}$ values for TOC, BOD, and COD were found at only 6 or fewer stations, depending on the CoPC.

- *Rhepoxynius abronius* survival exhibited significant ($P \leq 0.05$) and strong ($r_s > -0.75$) negative correlations with sediment concentrations of ammonia and 4-methylphenol. By contrast, normal survival of echinoderm embryos did not correlate strongly with any CoPCs. *R. abronius* survival also exhibited significant ($P \leq 0.05$) and strong ($r_s > -0.75$) negative correlations with concentrations of ammonia and sulfide in the pore water of the toxicity test chambers at the end of the 10-day test period.

- Comparisons with historical sediment data collected in 1994 and 1995 in Ward Cove showed that both chemical concentrations and sediment toxicity results for the top 2 cm of sediment (1994 and 1995) were similar to the values found for the top 10 cm of sediment (1996 and 1997).

- Results of four specialized toxicity tests applied to sediments from eight representative stations in Ward Cove suggest that sulfide, rather than ammonia, may be a major cause of the observed sediment toxicity. Because both CoPCs covaried, it was difficult to determine their independent contributions to toxicity. However, sulfide appeared to be the major cause of toxicity because porewater concentrations in most samples substantially exceeded the 48-hour LC50 for *Rhepoxynius abronius*, and because simple aeration of pore water (and the resulting oxidation of sulfide) eliminated toxicity in all but one sample. By contrast, ammonia concentrations generally were lower than the 96-hour LC50 for *R. abronius*, and toxicity did not respond as strongly to reductions in ammonia concentrations as it did to reductions in sulfide concentrations.

- The CoCs for sediment toxicity are ammonia, sulfide, and 4-methylphenol.

In general, the results of the specialized toxicity tests were consistent with the results of the four kinds of sediment toxicity tests used to characterize sediments throughout Ward Cove (i.e., the 10-day tests based on the amphipods *Rhepoxynius abronius* and *Leptocheirus plumulosus*, the 20-day juvenile polychaete test based on *Neanthes* sp., and the 96-hour echinoderm embryo test based on *Dendraster excentricus*). The implication based on the specialized tests that sulfide was largely responsible for the observed toxicity is consistent with the patterns of toxicity found for the four sediment toxicity tests.

The unusual pattern of two tests exhibiting toxic responses (i.e., the amphipod test based on *Rhepoxynius abronius* and the echinoderm embryo test based on normal survival) and two tests showing no toxic responses (i.e., the amphipod test based on *Leptocheirus plumulosus* and the juvenile polychaete test) is consistent with sulfide being the primary toxicant, given the different life histories of the test species. Because *L. plumulosus* and *Neanthes* live in tubes, they have an enhanced ability to isolate themselves from the ambient sediment by controlling the diffusion rate of porewater solutes into the tube

environment (Aller 1982). In addition, by aerating the water in their tubes, organisms can effectively isolate themselves from oxidizable porewater constituents such as sulfide. By controlling the microenvironments within their tubes, many tubicolous organisms can inhabit sediments that are toxic to free-burrowing organisms such as *R. abronius*. This ability partly accounts for the fact that the first organisms to colonize many disturbed sediments are generally small, opportunistic, tube-dwelling polychaetes, followed by tube-dwelling amphipods (Rhoads and Boyer 1982).

7.2 FOOD-WEB ASSESSMENT

The objective of the food-web assessment is to determine whether chemicals in the sediments of Ward Cove pose a potential risk of adverse effects to key ecological receptors in the food web of the Cove. To be conservative, the assessment focuses on the birds and mammals found at the top of the site-specific food web, because they are considered to be at greatest risk from bioaccumulation in the Cove food web. Risks were predicted using the maximum chemical concentrations found in Cove sediments during 1996 and 1997 and food-web models based on conservative, yet realistic, assumptions. Risks were also predicted using the maximum sediment chemical concentrations found in 1994, 1995, or the present investigation (i.e., 1996 or 1997; Appendix G).

Food-web exposure models were used to evaluate potential ecological risk to two mammal species and two sea bird species resulting from exposure to chemicals in Ward Cove. Mammals evaluated were the harbor seal and river otter. Sea birds evaluated were the marbled murrelet and pelagic cormorant. These species were selected primarily because they are upper trophic level species whose habitat-use characteristics suggest they have the highest potential for exposure to bioaccumulative chemicals in Ward Cove.

In selecting the representative receptor species, consideration was given to the kinds of organisms commonly found near Ward Cove, as well as the presence of protected (i.e., threatened or endangered) species. ENSR (1995d) reviewed and summarized the available information on the organisms found in Tongass Narrows near Ward Cove. Although there are no protected fish species in the vicinity, there are several protected species of birds and mammals that may spend some amount of time near the Cove.

For protected birds, one endangered species (i.e., American peregrine falcon [*Falco peregrinus anatum*]) could occur in Tongass Narrows as a transient, primarily during seasonal migration (USFWS 1998). However, the American peregrine falcon would not be expected to be at risk of exposure to CoPCs from Ward Cove because they rarely are found in the vicinity of the Cove and fish is not a major dietary component (Terres 1996).

For protected mammals, one endangered species (humpback whale [*Megaptera novaengliae*]) and one threatened species (Stellar sea lion [*Eumetopia jubatus*]) could occur in Tongass Narrows near Ward Cove (NMFS 1998). Although humpback whales occur periodically in the vicinity of Tongass Narrows, the area is not considered an important habitat for that species (ENSR 1995d). For example, in nine surveys conducted between 1991 and 1993 by the National Marine Fisheries Service, no

humpback whales were seen in the vicinity of Tongass Narrows (Dahlheim 1995, pers. comm.).

Although Tongass Narrows is included in the range of the Stellar sea lion, there are no rookeries near Ward Cove. The closest rookery is located on Forrester Island, approximately 100 miles southwest of the Cove (ENSR 1995d). A haul-out area has been observed on Grindall Island (approximately 15 miles from the Cove), where more than 200 Stellar sea lions have been observed on several occasions (McAllister 1994, pers. comm.). Additional documented haul-out areas in the vicinity of Ward Cove include West Rock and Duke Island, both of which are located approximately 30 miles from the Cove. Based on the overall range of the Stellar sea lion and the proximity of several haul-out areas to the Ward Cove, it is possible that individual sea lions visit the Cove periodically and are potentially at risk from exposure to CoPCs in the Cove. However, because the nearest rookery to the Cove is approximately 100 miles away, it is unlikely that CoPCs from the Cove pose a risk to this species during the breeding period. The harbor seal was selected for the food-web assessment because it is representative of the protected Stellar sea lion, but has a higher potential for being exposed to CoPCs from the Cove. Harbor seals have a higher potential for exposure because they are more abundant in Tongass Narrows than Stellar sea lions, they are found in Tongass Narrows throughout the year (NOAA 1988), and they do not make extensive migrations (Bigg 1969).

Food-web exposure models estimate exposure to chemicals expressed as a total daily dose for each ecological receptor (Pastorok et al. 1996). Because toxicity reference values (TRVs) are typically reported as the threshold daily dose to an individual, estimation of a site-specific rate of chemical intake (IR_h) allows direct comparison of exposure estimates to toxicity benchmark values. Exposure assumptions are based on chemical characteristics and natural history information of each ecological receptor compiled from the literature or obtained through discussions with expert researchers. Specific exposure assumptions and references used in developing the food-web exposure models for each ecological receptor are provided in Section 7.2.2. Species-specific model variables and assumptions are described below.

The general structure of the exposure model is as follows:

$$IR_{ingestion} = \sum_h (T_h)(IR_h) = \left[\sum_h (T_h)\right]\left[\frac{\sum_i (C_{ih} \times M_i \times A_i \times FR)}{BW}\right]$$

where:

$IR_{ingestion}$ = species-specific total rate of chemical intake by ingestion (mg/kg-day wet weight)

T_h = fraction of year that species spends in habitat h

IR_h = species-specific rate of chemical intake by ingestion in habitat h (mg/kg-day wet weight)

C_{ih} = concentration of the chemical in medium i of habitat h (mg/kg dry weight)

M_i = rate of ingestion of medium i (kg dry weight/day)

A_i = relative gastrointestinal absorption efficiency for the chemical in medium i (proportion)

FR = proportion of Ward Cove site area relative to receptor home range

BW = body weight of receptor species (kg).

For all food-web exposure models, T_h and A_i were conservatively considered to equal 1.0.

Site-specific data on sediment chemical concentrations collected in the Phase 1 and 2 investigations were incorporated into the model to estimate chemical uptake. Estimates of chemical uptake were derived using the mean and maximum sediment concentrations for 1996 and 1997 data combined. Chemical concentrations in prey species were estimated using site-specific sediment chemical concentrations and BSAFs obtained from the scientific literature (see Section 4.4.1 and Tables 4-5 and 4-6).

7.2.1 Chemicals of Potential Concern for Ecological Risk

From the standpoint of bioaccumulation, the primary CoPCs in Ward Cove have been identified as total mercury and PCDDs/Fs (U.S. EPA 1994a; ENSR 1995a, EVS 1996). However, to be conservative, several additional chemicals were added to this list because they were found at elevated concentrations (relative to reference conditions) throughout relatively large areas of the Cove. These additional chemicals are arsenic, cadmium, zinc, and PAHs. Although other chemicals were found at elevated concentrations in Cove sediments (i.e., phenol, 4-methylphenol, benzoic acid, and pulp mill compounds), they were not considered in the food-web assessment because their distribution was highly localized, they have rarely been addressed in food-web assessments in other studies, and there is little information in the literature regarding their bioaccumulation potential.

Mammalian and avian TRVs are available for arsenic, cadmium, mercury (total and methylmercury), zinc, and 2,3,7,8-TCDD; a mammalian TRV also is available for benzo[a]pyrene. Because TRVs do not exist for most of the PCDD/F congeners, sediment concentrations of both PCDD/F congeners were converted to 2,3,7,8-TCDD TECs using the toxicity equivalence factor (TEF) methodology proposed by U.S. EPA (1989c). TEFs in the EPA methodology are derived from studies on mammalian species. TEFs are also available for birds, but not for all PCDD/F congeners; therefore, exposure models use mammalian-derived TEFs to calculate TECs for both mammals and birds. Available data indicate that TEFs for birds are within the same range as values for mammals (Kennedy et al. 1996). Therefore, using mammalian TEFs for avian receptors probably does not constitute a major uncertainty in the exposure models. Risk to wildlife receptors was modeled using TRVs for 2,3,7,8-TCDD. A similar approach was used to evaluate risk of PAHs to mammals; relative potency factors were used to convert sediment concentrations of the

carcinogenic PAHs to benzo[a]pyrene equivalents. Exposure to PAHs is discussed qualitatively for birds, because no TRVs are available to compare with exposure estimates.

7.2.2 Exposure Assessment

In the exposure assessment, estimates were made of daily intake of chemicals by each receptor as a result of exposure through the food web. Incidental sediment ingestion is included in the food-web models, with each ecological receptor (i.e., harbor seal, river otter, marbled murrelet, and pelagic cormorant) assumed to ingest sediment while foraging at 2 percent of the daily food ingestion rate. The primary reference for sediment ingestion by wildlife species (Beyer et al. 1994) does not include data on marine species. The estimate of 2 percent incidental sediment ingestion is likely conservative given that the majority of the prey species consumed by each of the ecological receptors are pelagic, and no sediment ingestion is expected when feeding on pelagic species. Many of the remaining prey items are taken from rocky bottoms where incidental sediment ingestion is expected to be negligible.

Dietary intake rates are compared to TRVs in the risk characterization step to estimate potential risks. The intake rate is determined in part by life history traits of receptor species that influence their exposure to chemicals. Life history traits of the receptors relevant to the exposure assessment are described in the following sections.

7.2.2.1 Life History Characteristics of Mammalian Receptors

Harbor Seal—Harbor seals are top predators in the coastal marine environment of Alaska (Frost et al. 1996). Harbor seals forage primarily in shallow, near-shore waters making short, relatively shallow dives. More than half of the dives occur in water less than 50 m in depth and only 2 percent occur in water deeper than 150 m (Swain et al. 1996). Suryan (1995) reports that uneven, shallow seafloors with tide rips are characteristic harbor seal foraging areas. About 85 percent of the foraging occurs within 32 km (20 miles) of haul-out sites, with harbor seals typically using multiple haul-out sites within their home range (Frost et al. 1996). Time spent at haul-out sites is thought to be affected by several factors, including weather, seasonal variation in food, reproduction (i.e., pupping), and human disturbance (Pitcher and McAllister 1981; Wilson 1993).

The diet of harbor seals is diverse, comprising various types of fish (i.e., pelagic, demersal, anadromous, catadromous), cephalopods, and crustaceans (Ronald et al. 1982). The diet of harbor seals in southeastern Alaska is assumed to consist of fish (70 percent), squid and octopus (22 percent), crab (4 percent), and shrimp (4 percent). The estimate of diet composition was qualitatively developed from data reported in several documents, including Ronald et al. (1982) and five studies reported in U.S. EPA (1993b). No BSAF values were found in the literature for any of the CoPCs in squid; therefore, for the food-web exposure model, the diet of harbor seal was assumed to consist of fish (84 percent), crabs (8 percent), and shrimp (8 percent). Exclusion of squid from the model represents

an uncertainty in the calculation of hazard quotients. A total daily food ingestion rate of 1.9 kg/day (dry weight) was estimated using allometric scaling (Nagy 1987) and a mean body weight of 56.7 kg for female harbor seals in the Gulf of Alaska (Silva and Downing 1995). Body weights of females were used because the TRV is based on female exposure.

Harbor seals center their activity around haul-out sites, which may include gravel or sand beaches, intertidal reefs, rocky shorelines, mud bars, and floating glacial ice (Frost et al. 1996). Haul-out sites are located in areas away from human or other disturbance factors and at sites where seals have immediate access to deep water. Haul-out sites also are the focus of the reproductive effort with mating and the birth of pups generally occurring at the haul-out sites (Frost et al. 1996). For this ecological risk assessment, a harbor seal haul-out site is conservatively assumed to occur within Ward Cove, although no haul-out sites are known from this area.

In general, harbor seals are sedentary, spending as much as 20 percent of their time on land at haul-out sites (Ronald et al. 1982, Swain et al. 1996). It is estimated that most harbor seal activity occurs within 50 km (30 miles) of haul-out sites (Lewis 1996, pers. comm.), although some individuals may routinely travel 100–150 km every 7–10 days (Frost 1996, pers. comm.) and round-trips of 300 km have been documented (Swain et al. 1996). For purposes of the food-web modeling, foraging by harbor seals is conservatively estimated to occur within 5 km of the haul-out sites (Stewart et al. 1989; Suryan 1995). A 5-km radius defines a foraging territory of 7,800 ha. For the food-web exposure model, it is estimated that half of this area (3,900 ha) consists of unsuitable terrestrial habitats and that foraging is equally distributed across the remainder of the foraging territory. Thus, Ward Cove (111 ha) conservatively represents about 3 percent of the foraging range of a harbor seal.

River Otter—The river otter belongs to the family Mustelidae (weasel) and is closely related to the sea otter (*Enhydra lutris*). Although primarily a freshwater species, river otters have been documented from estuarine habitats of coastal Washington, marine habitats of the San Juan Archipelago and Strait of Juan de Fuca (Toweill and Tabor 1982), British Columbia (Stenson et al. 1984), and southeastern Alaska (Larsen 1984). Adult river otters in North America weigh between 5 and 14 kg (Harris 1968), with the largest individuals occurring in southeastern Alaska (Toweill and Tabor 1982). River otters also exhibit sexual dimorphism with females being smaller than males.

In southeastern Alaska and British Columbia, the diet of river otters foraging in marine habitats consists primarily of fish, with minor quantities of crabs, clams, gastropods, shrimp, limpets, birds, mammals, and other marine invertebrates (Toweill and Tabor 1982; Larsen 1983, 1984; Stenson et al. 1984). The most common fish groups identified from stomach contents and scat analyses include cottids (sculpins), scorpaenids (rockfish), embiotocids (surfperches), and hexagrammids (greenlings and lingcod), fish typically found in tidal and subtidal zones, often in rocky substrates. As originally proposed by Ryder (1955), river otters prey on fish in direct proportion to their abundance and

inverse proportion to their swimming ability. For the food-web exposure model, the diet is estimated to consist of fish (83 percent), crabs (10 percent), bivalves (3.5 percent) and gastropods (3.5 percent). Although birds (i.e., cormorants and grebes) may constitute up to 6 percent of the diet (Stenson et al. 1984), no BSAF values are available to estimate bird tissue concentrations from sediments. Exclusion of birds from the model represents an uncertainty in the calculation of hazard quotients. A total daily food ingestion rate of 0.422 kg/day (dry weight) was estimated using allometric scaling (Nagy 1987) and derived from a mean body weight of 9.1 kg reported for both sexes from Waterton Lakes, Alberta, Canada (Silva and Downing 1995).

Home range size for river otters in southeastern Alaska has been estimated at 900–2,500 ha with population densities of 0.43–0.58 individuals per kilometer of shoreline (Larsen 1983). Males tend to be solitary outside the breeding season, maintaining larger territories than females, with mean daily movements of 9–16 km (Erlinge 1967). Female river otters usually occur with a family group of two or three pups, often associated with one to three subadults or nonbreeding adults (Melquist and Hornocker 1979). According to these data, Ward Cove would represent about 12 percent of the smallest reported home range for river otters and less than 5 percent of the largest home range (Larsen 1983; U.S. EPA 1993b). However, river otters in the vicinity of the KPC site are estimated to obtain up to 25 percent of their diet from habitats within Ward Cove (Larsen 1996, pers. comm.). The remainder of the river otter diet is likely to be collected from various freshwater habitats (25 percent) and from coastal areas outside of Ward Cove (50 percent).

River otters typically breed in late winter or early spring over a period of 3–5 months (Toweill and Tabor 1982). Adult river otters do not reach sexual maturity until 2–3 years of age, although they may not breed until 5–7 years of age (Toweill and Tabor 1982). Breeding may occur each year (Tabor and Wight 1977, as cited by Toweill and Tabor 1982) although researchers in several parts of North America report river otters breeding only in alternate years or at 2- to 3-year intervals (Liers 1951; Lauhachinda 1978; Mowbray et al. 1979, as cited by Toweill and Tabor 1982). Litters typically are two to four pups (Hooper and Ostenson 1949, as cited by Toweill and Tabor 1982).

7.2.2.2 Life History Characteristics of Avian Receptors

Marbled Murrelet—The marbled murrelet belongs to the family Alcidae and is closely related to both puffins and auklets. The species occurs from central California to the Kenai Peninsula, Alaska, and is known to breed throughout this range during summer months. Marbled murrelets forage primarily in shallow coastal waters, inlets, bays, sounds, and saltwater passages (Marshall 1990). The species often gathers in small flocks near the mouths of rivers, although in Prince William Sound, marbled murrelets are more commonly observed in exposed waters and are relatively uncommon in bays, fjords, and passes (Marshall 1990; Kuletz et al. 1995).

Marbled murrelets are solitary, opportunistic feeders, with diet selection influenced by site-specific conditions that regulate the types and quantity of available prey (Sealy 1975b; Burkett 1995). The diet consists primarily of invertebrates and small marine fish, although marbled murrelets may also forage in freshwater habitats (Carter and Sealy 1986). Common invertebrate prey items include krill (euphasiids), shrimp (mysids), and amphipods (Burkett 1995). Small fish include schooling species such as the sand lance (*Ammodytes hexapterus*), anchovy (*Engraulis encrasicholus*), Pacific herring (*Clupea harengus*), capelin (*Mallotus villosus*), and seaperch (*Cymatogaster aggregata*) (Marshall 1990; Burkett 1995). Fish constitute a larger percentage of the diet than invertebrates during the summer reproductive season, at which time sand lance and seaperch are the most commonly eaten prey (Sealy 1975b). For the food-web exposure models, the diet of the marbled murrelet is estimated to consist of sand lance and Pacific herring (70 percent), krill and shrimp (20 percent), and seaperch (10 percent) (Sealy 1975b; Burkett 1995). A total daily food ingestion rate of 0.022 kg/day (dry weight) was estimated using allometric scaling (Nagy 1987) from a mean body weight (sex unspecified) of 0.22 kg (Dunning 1993).

Data from Alaska indicate a typical foraging range of 20–30 km from the nest site for marbled murrelets (Kuletz et al. 1995). Based on radiotelemetry data, a minimum home range of 119 km^2 (11,900 ha) has been calculated for six nesting marbled murrelets in Alaska (Kuletz et al. 1995). Foraging usually occurs within 1 km of the shore in water less than 115 m in depth (Sealy 1975b; Kuletz et al. 1995). However, in some areas of Alaska, foraging occurs in mid-channel areas near deep-water sills where upwelling promotes productivity and concentrates prey (Burrell 1987; Hunt 1995). If it is conservatively assumed that only 10 percent of the home range represents potential foraging areas and all potential areas receive equal use, then Ward Cove (111 ha) represents less than 9 percent of the foraging area of a marbled murrelet. For purposes of the food-web model, it is assumed that 10 percent of the diet of a marbled murrelet is obtained from habitats in Ward Cove.

Nesting occurs primarily inland, although marbled murrelets are also known to nest on the ground and on cliffs along coastal areas (Kuletz et al. 1995). When nesting inland, marbled murrelets construct nests on the ground, on tree branches, or in tree cavities within areas of old growth forest. Sexual maturity is not reached until the year after hatching, and as much as 15 percent of the population each year may be nonreproductive individuals (Sealy 1975a). Usually, only one egg is laid, a characteristic of most alcids. During winter, marbled murrelets overwinter in the same general areas used during the breeding season, although the most northerly populations move southward (Marshall 1990).

Pelagic Cormorant—Pelagic cormorants occur from southern California to the Bering Sea and are abundant in Alaska (Ainley et al. 1981; Nysewander 1986). Pelagic cormorants in Alaska forage primarily from inshore areas, usually within 3 km of the shore (Ainley et al. 1981; Nysewander 1986). The most common prey items include sand lance, shrimp, gunnels (*Pholis laeta*), and polychaetes (*Nereis* sp.). Pelagic cormorants

typically feed on solitary, nonschooling fish in rocky substrates or sea bottoms covered with kelp. Sanger (1983) reported the diet of adult pelagic cormorants to consist entirely of fish, primarily sand lance, cottids, capelins, and walleye pollock, with sand lance predominant in the diet of nestlings. Therefore, for the food-web exposure model, the diet is assumed to consist entirely of fish. A total daily food ingestion rate of 0.093 kg/day (dry weight) was estimated using allometric scaling (Nagy 1987) from a mean female body weight of 1.7 kg (Dunning 1993). Although 45 percent of the diet is bottom fish, foraging typically occurs in rocky-bottomed substrates or in kelp beds, so the potential for ingesting sediment may be low.

No data were found in the literature regarding the size of foraging territories for either pelagic or double-crested cormorants (*Phalacrocorax auritus*). Although most foraging by pelagic cormorants occurs within 3 km of shore (Ainley et al. 1981; Nysewander 1986), the linear distance along the coastline that would be used is not known. For purposes of the food-web exposure model, it is assumed that the foraging territory of a pelagic cormorant is similar to that of the marbled murrelet (1,190 ha) and that Ward Cove represents about 10 percent of the foraging territory.

Pelagic cormorants breed from Forrester Island in southeastern Alaska to Cape Thompson in the Chukchi Sea and throughout the Aleutian Islands (Nysewander 1986). The total breeding population in Alaska is estimated at between 41,000 and 90,000 birds (Nysewander 1986; Sowls et al. 1978). Breeding colonies of the pelagic cormorant are small, usually having fewer than 100 pairs, and are located in precipitous locations on the shoulders and ledges of cliffs (Nysewander 1986). Nest site fidelity is low with nest sites and whole colonies often changing locations annually (Nysewander 1986). Breeding begins in late May to early June.

7.2.3 Toxicity Assessment

Daily dietary CoPC exposures estimated for receptor species in the exposure assessment are compared with TRVs that pose no risk of adverse effects in test species. TRVs were obtained from studies in the literature in which a chronic no-observed-adverse-effect level (NOAEL) was observed or estimated on the basis of a relevant ecological endpoint (i.e., reproduction, mortality). The following sections discuss the derivation of TRVs for each CoPC.

7.2.3.1 Derivation of Toxicity Reference Values (Mammalian Receptors)

Arsenic—An unbounded chronic lowest-observed-adverse-effect level (LOAEL) of 1.26 mg/kg-day was derived for arsenic from a multi-generation study of mice (Schroeder and Mitchener 1971). One dose level of arsenic (as arsenite, As^{3+}) was administered orally in water and food, and effects on reproduction were monitored over three generations. Mice survived well through the third generation. No maternal deaths, dead litters, or runts were recorded during the study. There were eight young deaths and

one failure to breed, but this response was not significantly greater than the response of the control group. The only observed effect was a decrease in litter size to 75 percent, 93 percent, and 77 percent of control group litter size for the F_1, F_2, and F_3 generations, respectively. Because the extent of the decrease in litter size was not severe, the LOAEL was adjusted to a NOAEL of 0.126 mg/kg-day by the application of an uncertainty factor of 10 (U.S. EPA 1994b).

Cadmium—A bounded NOAEL of 1 mg/kg-day in the diet was derived for potential effects of cadmium (soluble salt) on rats in a 6-week study (Sutou et al. 1980). Three dose levels of cadmium were administered by oral gavage, and the study endpoint was reproductive success. No adverse effects were observed at the 1 mg/kg-day dose, but fetal implantations and fetal survivorship were reduced by 28 and 50 percent, respectively, and fetal resorptions increased by 400 percent in a 10 mg/kg-day dose group. Because the study considered oral exposure during reproduction, the 1 mg/kg-day dose was considered to be a chronic NOAEL.

Methylmercury—In a multi-generation toxicity study with laboratory rats, a bounded chronic NOAEL of 0.032 mg/kg-day was derived for methylmercury chloride (Verschuuren et al. 1976). Three doses levels of methylmercury chloride were administered orally in the diet over three generations, with reproduction (pup viability) as the study endpoint. A dose of 0.016 mg/kg-day significantly reduced pup viability, while no effect was observed at 0.032 mg/kg-day.

Zinc—A chronic NOAEL of 160 mg/kg-day was derived from a dietary study of zinc (zinc oxide) evaluating reproductive endpoints (fetal resorption and fetal growth rates) in laboratory rats by Schlicker and Cox (1968). The zinc oxide was administered orally in the diet, and two dose levels were evaluated. Rats exposed to 320 mg/kg-day zinc in the diet displayed increased rates of fetal resorption and reduced fetal growth rates. Because no effects were observed at the 160 mg/kg-day dose rate and the exposure occurred during gestation (a critical life stage), this dose was considered a chronic NOAEL.

Benzo[a]pyrene—An unbounded chronic LOAEL value of 10 mg/kg-day was derived for benzo[a]pyrene from a reproductive study on mice (Mackenzie and Angevine 1981). Three dose levels were administered by oral intubation, and pregnancy rates, percentage of viable litters, pup weights, and fertility were monitored. Although the study was of short duration (7 to 16 days), it evaluated effects during a critical life stage. Total sterility was observed in 97 percent of the offspring in 40 and 100 mg/kg-day dose groups, and fertility was impaired among offspring in the 10 mg/kg-day group. Because all dose levels produced some measure of reproductive effects, the lowest dose tested was a chronic LOAEL of 10 mg/kg-day, to which a LOAEL-to-NOAEL uncertainty factor of 10 was applied to yield a NOAEL of 1 mg/kg-day.

2,3,7,8-TCDD—A chronic NOAEL of 1×10^{-6} mg/kg-day was derived for 2,3,7,8-TCDD from a multi-generation study of rats in which doses were administered orally in their diet (Murray et al. 1979). Fertility and neonatal survival were significantly reduced among rats receiving 1×10^{-4} mg/kg-day or 1×10^{-5} mg/kg-day 2,3,7,8-TCDD. Because no significant differences were observed at the 1×10^{-6} mg/kg-day dose and the study considered exposure through three generations, including critical life stages (reproduction), this dose was considered to be a chronic NOAEL.

7.2.3.2 Derivation of Toxicity Reference Values (Avian Receptors)

Arsenic—A bounded chronic NOAEL of 10 mg/kg-day was derived for arsenic from a study using mallard ducks (*Anas platyrhynchos*) (Stanley et al. 1994). Three dose levels of arsenic (as sodium arsenate) were administered in the diet for 16 weeks, and effects on reproductive performance were assessed. An increase in the days between pairing and production of the first egg, a decrease in egg weights, and decreases in duckling growth were observed at an arsenic intake rate of 40 mg/kg-day, while no adverse effects were seen at 10 mg/kg-day.

Cadmium—A bounded chronic NOAEL of 0.71 mg/kg-day was derived for cadmium from a study using white Leghorn chickens (Leach et al. 1979). Three dose levels of cadmium (as cadmium sulfate) were administered in the diet for 48 weeks, and effects on egg production and eggshell thickness were assessed. Significant decreases in egg production and eggshell thickness were observed at a cadmium intake rate of 2.82 mg/kg-day, while no adverse effects were seen at 0.71 mg/kg-day.

Methylmercury—An unbounded LOAEL of 0.064 mg/kg-day was derived for potential effects of methylmercury dicyandiamide on mallards in a multi-generation study evaluating reproductive endpoints (egg production and number of ducklings hatched) by Heinz (1979). A NOAEL of 0.032 mg/kg-day was developed by applying a LOAEL-to-NOAEL uncertainty factor of 2, as suggested by U.S. EPA (1993b). An uncertainty factor of 2 rather than 10 was used because the LOAEL appeared to be near the threshold for dietary effects.

Zinc— A bounded chronic NOAEL of 98.8 mg/kg-day was derived for zinc from a study using broiler chickens (Johnson et al. 1962). Seven dose levels of zinc (as zinc oxide) were administered in the diet of growing broiler chickens for 10 weeks, with mortality and growth monitored throughout the study. A reduction in live weight, but no effect on mortality, was observed at a zinc intake rate of 131 mg/kg-day, while no adverse effects were seen at 98.8 mg/kg-day.

2,3,7,8-TCDD—A bounded chronic NOAEL of 1.4×10^{-5} mg/kg-day was derived for 2,3,7,8-TCDD from a study of ring-necked pheasants (*Phasianus colchicus*) in which doses of 2,3,7,8-TCDD were administered intraperitoneally (Nosek et al. 1992). The intraperitoneal exposure route used in the study is comparable to oral routes of exposure (U.S. EPA 1993b). Egg production and hatchability were significantly reduced among birds receiving 1.4×10^{-4} mg/kg-day, but no significant differences were observed at 1.4×10^{-5} mg/kg-day or 1.4×10^{-6} mg/kg-day. Because the study considered exposure through a critical life stage (reproduction), the 1.4×10^{-5} mg/kg-day dose was considered a chronic NOAEL.

7.2.4 Risk Characterization

In this section, the results of the exposure and effects assessments are combined to estimate the risks to avian and mammalian receptors posed by CoPCs in prey species tissue. Risks are presented as hazard quotient values, which are calculated for each CoPC by dividing the total daily dose by the appropriate TRV. Hazard quotients less than 1.0 indicate that a CoPC is unlikely to cause adverse ecological effects, given the conservative assumptions used in the food-web exposure model. A hazard quotient greater than 1.0 indicates that the exposure of the modeled individual has exceeded the TRV. If the rate of exposure exceeds the TRV, then there is a potential that some fraction of the population may experience an adverse health effect as a direct result of the presence of the CoPC. However, the hazard quotient must be considered with regard to the uncertainty associated with the parameters evaluated as part of the model. The major uncertainties associated with the risk estimation are also described in this section.

7.2.4.1 Risk Estimation

Food-web exposure models indicate that harbor seals are not at risk of adverse effects from exposure to any CoPC at Ward Cove (Table 7-35). For river otters, a risk of adverse effects may exist from exposure to PCDDs/Fs, because the hazard quotient exceeds 1.0 based on the maximum sediment concentration, although not when based on the mean sediment concentration (Table 7-36). Based on the maximum sediment concentration, a risk of adverse effects may exist from exposure to cadmium at Ward Cove for marbled murrelets (Table 7-37). Food-web models indicate that pelagic cormorants are not at risk of adverse effects from exposure to any CoPC at Ward Cove (Table 7-38).

EPA guidelines for screening-level ecological risk assessments (U.S. EPA 1997a) suggest that if multiple contaminants of concern exist at a site, it might be appropriate to sum hazard quotients for receptors to derive a hazard index. This approach assumes that effects of simultaneous exposure to a mixture of CoPCs are strictly additive. In addition, this approach is most properly applied only to compounds that induce the same effect by the same mechanism of action. This limitation is recognized in Superfund risk

TABLE 7-35. FOOD-WEB EXPOSURE MODEL CALCULATIONS FOR ASSESSMENT OF HYPOTHETICAL RISK TO HARBOR SEALS AT WARD COVE

HARBOR SEAL (*Phoca vitulina*)

Body weight (kg)	56.7	Sediment ingestion rate (kg/day)	0.038
Food ingestion rate (kg/day)	1.898	Area use factor	0.03

Food Item	Percent of diet
Fish	84
Crabs	8
Shrimp	8

Chemical (max. concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Crab BSAF	Crab Concentration (mg/kg)	Crab Exposure (mg/kg-day)	Shrimp BSAF	Shrimp Concentration (mg/kg)	Shrimp Exposure (mg/kg-day)	Total Food Exposure (mg/kg-day)	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Arsenic	39	7.8×10^{-4}	0.12	0.47	3.9×10^{-4}	0.022	0.086	6.9×10^{-6}	NA			4.0×10^{-4}	0.0012	0.13	0.009
Cadmium	7.3	1.5×10^{-4}	2.0	14.6	0.012	3.0	21.9	0.0018	44	321	0.026	0.040	0.040	1.0	0.04
Mercury	0.7	1.4×10^{-5}	0.38	0.27	2.2×10^{-4}	0.13	0.091	7.3×10^{-6}	1.0	0.70	5.6×10^{-5}	2.9×10^{-4}	3.0×10^{-4}	0.032	0.009
Zinc	396[a]	0.0080	5.0	1,980	1.7	3.2	1,267	0.10	0.16	63	0.0051	1.8	1.8	160	0.011
2,3,7,8-TCDD	4.6×10^{-5}	9.2×10^{-10}	1.0	1.9×10^{-4}	1.6×10^{-7}	1.0	2.6×10^{-5}	2.1×10^{-9}	0.70	2.5×10^{-5}	2.0×10^{-9}	1.7×10^{-7}	1.7×10^{-7}	1.0×10^{-6}	0.17
Benzo[a]pyrene	0.41[b]	8.2×10^{-6}	NA			0.63	0.14	1.1×10^{-5}	NA			1.1×10^{-5}	1.9×10^{-5}	1.0	1.9×10^{-5}

Chemical (mean concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Crab BSAF	Crab Concentration (mg/kg)	Crab Exposure (mg/kg-day)	Shrimp BSAF	Shrimp Concentration (mg/kg)	Shrimp Exposure (mg/kg-day)	Total Food Exposure (mg/kg-day)	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Arsenic	22	4.4×10^{-4}	0.12	0.26	2.2×10^{-4}	0.022	0.048	3.9×10^{-6}	NA			2.3×10^{-4}	6.7×10^{-4}	0.13	0.005
Cadmium	3.5	7.0×10^{-5}	2.0	7.0	0.0059	3.0	11	8.4×10^{-4}	44	154	0.012	0.019	0.019	1.0	0.02
Mercury	0.10	2.0×10^{-6}	0.38	0.038	3.2×10^{-5}	0.13	0.013	1.0×10^{-6}	1.0	0.10	8.0×10^{-6}	4.1×10^{-5}	4.3×10^{-5}	0.032	0.001
Zinc	190	0.0038	5.0	950	0.80	3.2	610	0.049	0.16	30.4	0.0024	0.85	0.86	160	0.005
2,3,7,8-TCDD	1.7×10^{-5}	3.4×10^{-10}	1.0	7.2×10^{-5}	6.1×10^{-8}	1.0	9.5×10^{-6}	7.7×10^{-10}	0.70	9.2×10^{-6}	7.4×10^{-10}	6.2×10^{-8}	6.3×10^{-8}	1.0×10^{-6}	0.06
Benzo[a]pyrene	0.16	3.2×10^{-6}	NA			0.63	0.054	4.4×10^{-6}	NA			4.4×10^{-6}	7.6×10^{-6}	1.0	7.6×10^{-6}

Note: All concentrations expressed as dry weight values.

BSAF - biota-sediment accumulation factor
NA - not available
TCDD - tetrachlorodibenzo-*p*-dioxin
TRV - toxicity reference value

[a] Excludes a value of 530 mg/kg detected in 1997 at Station 25 near the cannery.

[b] Excludes higher values detected near the cannery and state airplane ramp at Stations 23 and 25.

TABLE 7-36. FOOD-WEB EXPOSURE MODEL CALCULATIONS FOR ASSESSMENT OF HYPOTHETICAL RISK TO RIVER OTTERS AT WARD COVE

RIVER OTTER (*Lutra canadensis*)

Body weight (kg)	9.1	Sediment ingestion rate (kg/day)	0.0084
Food ingestion rate (kg/day)	0.422	Area use factor	0.25

Food Item	Percent of diet
Fish	83
Crabs	10
Bivalves	3.5
Gastropods	3.5

Chemical (max. concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Crab BSAF	Crab Concentration (mg/kg)	Crab Exposure (mg/kg-day)	Bivalve BSAF
Arsenic	39	0.0090	0.12	0.47	0.0045	0.022	0.086	9.9×10^{-5}	0.71
Cadmium	7.3	0.0017	2.0	14.6	0.14	3.0	21.9	0.025	7.5
Mercury	0.7	1.6×10^{-4}	0.38	0.27	0.0026	0.13	0.091	1.1×10^{-4}	4.5
Zinc	396[a]	0.092	5.0	1,980	19.1	3.2	1,267	1.5	7.3
2,3,7,8-TCDD	4.6×10^{-5}	1.1×10^{-8}	1.0	1.9×10^{-4}	1.9×10^{-6}	1.0	2.6×10^{-5}	3.0×10^{-8}	0.90
Benzo[a]pyrene	0.41[b]	9.5×10^{-5}	NA			0.63	0.14	1.6×10^{-4}	0.63

Chemical (mean concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Crab BSAF	Crab Concentration (mg/kg)	Crab Exposure (mg/kg-day)	Bivalve BSAF
Arsenic	22	0.0051	0.12	0.26	0.0025	0.022	0.048	5.6×10^{-5}	0.71
Cadmium	3.5	8.1×10^{-4}	2.0	7.0	0.067	3.0	11	0.012	7.5
Mercury	0.10	2.3×10^{-5}	0.38	0.038	3.7×10^{-4}	0.13	0.013	1.5×10^{-5}	4.5
Zinc	190	0.044	5.0	950	9.1	3.2	610	0.70	7.3
2,3,7,8-TCDD	1.7×10^{-5}	3.9×10^{-9}	1.0	7.2×10^{-5}	6.9×10^{-7}	1.0	9.5×10^{-6}	1.1×10^{-8}	0.90
Benzo[a]pyrene	0.16	3.7×10^{-5}	NA			0.63	0.054	6.3×10^{-5}	0.63

TABLE 7-36. FOOD-WEB EXPOSURE MODEL CALCULATIONS FOR ASSESSMENT OF HYPOTHETICAL RISK TO RIVER OTTERS AT WARD COVE

RIVER OTTER (*Lutra canadensis*)

Body weight (kg)	9.1	Sediment ingestion rate (kg/day)	0.0084
Food ingestion rate (kg/day)	0.422	Area use factor	0.25

Food Item	Percent of diet
Fish	83
Crabs	10
Bivalves	3.5
Gastropods	3.5

Chemical (max. concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Crab BSAF	Crab Concentration (mg/kg)	Crab Exposure (mg/kg-day)	Bivalve BSAF
Arsenic	39	0.0090	0.12	0.47	0.0045	0.022	0.086	9.9×10^{-5}	0.71
Cadmium	7.3	0.0017	2.0	14.6	0.14	3.0	21.9	0.025	7.5
Mercury	0.7	1.6×10^{-4}	0.38	0.27	0.0026	0.13	0.091	1.1×10^{-4}	4.5
Zinc	396[a]	0.092	5.0	1,980	19.1	3.2	1,267	1.5	7.3
2,3,7,8-TCDD	4.6×10^{-5}	1.1×10^{-8}	1.0	1.9×10^{-4}	1.9×10^{-6}	1.0	2.6×10^{-5}	3.0×10^{-8}	0.90
Benzo[a]pyrene	0.41[b]	9.5×10^{-5}	NA			0.63	0.14	1.6×10^{-4}	0.63

Chemical (mean concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Crab BSAF	Crab Concentration (mg/kg)	Crab Exposure (mg/kg-day)	Bivalve BSAF
Arsenic	22	0.0051	0.12	0.26	0.0025	0.022	0.048	5.6×10^{-5}	0.71
Cadmium	3.5	8.1×10^{-4}	2.0	7.0	0.067	3.0	11	0.012	7.5
Mercury	0.10	2.3×10^{-5}	0.38	0.038	3.7×10^{-4}	0.13	0.013	1.5×10^{-5}	4.5
Zinc	190	0.044	5.0	950	9.1	3.2	610	0.70	7.3
2,3,7,8-TCDD	1.7×10^{-5}	3.9×10^{-9}	1.0	7.2×10^{-5}	6.9×10^{-7}	1.0	9.5×10^{-6}	1.1×10^{-8}	0.90
Benzo[a]pyrene	0.16	3.7×10^{-5}	NA			0.63	0.054	6.3×10^{-5}	0.63

TABLE 7-37. FOOD-WEB EXPOSURE MODEL CALCULATIONS FOR ASSESSMENT OF HYPOTHETICAL RISK TO MARBLED MURRELETS AT WARD COVE

MARBLED MURRELET (*Brachyrampus marmoratus*)

Body weight (kg)	0.22	Sediment ingestion rate (kg/day)	0.0004
Food ingestion rate (kg/day)	0.022	Area use factor	0.1

Food Item	Percent of diet
Fish	80
Shrimp	20

Chemical (max. concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Shrimp BSAF	Shrimp Concentration (mg/kg)	Shrimp Exposure (mg/kg-day)	Total Food Exposure (mg/kg-day)	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Arsenic	39	0.0078	0.12	0.47	0.0038	NA			0.0038	0.012	10	0.0012
Cadmium	7.3	0.0015	2.0	14.6	0.12	44	321	0.64	0.76	0.76	0.71	1.07
Mercury	0.7	1.4×10^{-4}	0.38	0.27	0.0021	1.0	0.70	0.0014	0.0035	0.0037	0.032	0.11
Zinc	396[a]	0.079	5.0	1,980	15.9	0.16	63	0.13	16.0	16.1	98.8	0.16
2,3,7,8-TCDD	4.6×10^{-5}	9.2×10^{-9}	1.0	1.9×10^{-4}	1.6×10^{-6}	0.70	2.5×10^{-5}	5.0×10^{-8}	1.6×10^{-6}	1.6×10^{-6}	1.4×10^{-5}	0.12

Chemical (mean concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Shrimp BSAF	Shrimp Concentration (mg/kg)	Shrimp Exposure (mg/kg-day)	Total Food Exposure (mg/kg-day)	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Arsenic	22	0.0044	0.12	0.26	0.0021	NA			0.0021	0.007	10	6.5×10^{-4}
Cadmium	3.5	7.0×10^{-4}	2.0	7.00	0.056	44	154	0.31	0.37	0.37	0.71	0.52
Mercury	0.10	2.0×10^{-5}	0.38	0.04	3.0×10^{-4}	1.0	0.10	2.0×10^{-4}	5.1×10^{-4}	5.3×10^{-4}	0.032	0.016
Zinc	190	0.038	5.0	950	7.6	0.16	30.4	0.061	7.7	7.7	98.8	0.078
2,3,7,8-TCDD	1.7×10^{-5}	3.4×10^{-9}	1.0	7.2×10^{-5}	5.8×10^{-7}	0.70	9.2×10^{-6}	1.8×10^{-8}	6.0×10^{-7}	6.0×10^{-7}	1.4×10^{-5}	0.043

Note: All concentrations expressed as dry weight values.
BSAF - biota-sediment accumulation factor
NA - not available
TCDD - tetrachlorodibenzo-*p*-dioxin
TRV - toxicity reference value

[a] Excludes a value of 530 mg/kg detected in 1997 at Station 25 near the cannery.

TABLE 7-38. FOOD-WEB EXPOSURE MODEL CALCULATIONS FOR ASSESSMENT OF HYPOTHETICAL RISK TO PELAGIC CORMORANTS AT WARD COVE

PELAGIC CORMORANT (*Phalacrocorax pelagicus*)

Body weight (kg)	1.7	Sediment ingestion rate (kg/day)	0.0019
Food ingestion rate (kg/day)	0.093	Area use factor	0.1

Food Item	Percent of diet
Fish	100

Chemical (max. concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Total Food Exposure (mg/kg-day)	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Arsenic	39	0.0043	0.12	0.47	0.0026	0.0026	0.0068	10	6.8×10^{-4}
Cadmium	7.3	8.0×10^{-4}	2.0	14.6	0.080	0.080	0.081	0.71	0.11
Mercury	0.7	7.7×10^{-5}	0.38	0.27	0.0015	0.0015	0.0015	0.032	0.048
Zinc	396[a]	0.043	5.0	1,980	10.8	10.8	10.9	98.8	0.11
2,3,7,8-TCDD	4.6×10^{-5}	5.0×10^{-9}	1.0	1.9×10^{-4}	1.1×10^{-6}	1.1×10^{-6}	1.1×10^{-6}	1.4×10^{-5}	0.077

Chemical (mean concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Total Food Exposure (mg/kg-day)	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Arsenic	22	0.0024	0.12	0.26	0.0014	0.0014	0.0039	10	3.9×10^{-4}
Cadmium	3.5	3.8×10^{-4}	2.0	7.0	0.038	0.038	0.039	0.71	0.055
Mercury	0.10	1.1×10^{-5}	0.38	0.038	2.1×10^{-4}	0.0002	2.2×10^{-4}	0.032	0.007
Zinc	190	0.021	5.0	950	5.2	5.2	5.2	98.8	0.053
2,3,7,8-TCDD	1.7×10^{-5}	1.9×10^{-9}	1.0	7.2×10^{-5}	3.9×10^{-7}	3.9×10^{-7}	4.0×10^{-7}	1.4×10^{-5}	0.028

Note: All concentrations expressed as dry weight values.
BSAF - biota-sediment accumulation factor
TCDD - tetrachlorodibenzo-*p*-dioxin
TRV - toxicity reference value

[a] Excludes a value of 530 mg/kg detected in 1997 at Station 25 near the cannery.

assessment guidelines for human health evaluation (U.S. EPA 1989e), where it is noted that application of the hazard index equation to compounds that do not induce the same effect or act by the same mechanism could overestimate the potential for effects. This situation is likely for the ecological risk assessment at Ward Cove, because the mechanisms of toxicity and sites of action vary among the metals, PCDDs/Fs, and PAHs that constitute the CoPCs at the site.

Evaluation of the hazard quotients presented in Tables 7-35 to 7-38 indicates only two instances where the hazard index, if calculated, would be greater than 1.0: river otters and marbled murrelets when the hazard index is based on maximum sediment CoPC concentrations (the hazard index for murrelets based on mean concentrations is 1.0). In both cases, the substance responsible for causing the hazard index to exceed 1.0 is a CoPC that has a hazard quotient greater than 1.0 (2,3,7,8-TCDD for river otters and cadmium for marbled murrelets). Thus, the hazard index conveys no information regarding risk beyond that presented by the hazard quotients, which constitute a more appropriate tool for evaluating risk to ecological receptors at the Cove.

Avian risk of adverse effects from exposure to PAHs could not be estimated quantitatively. However, avian and mammalian exposure to PAHs is strongly influenced by the varying abilities of aquatic organisms (their food source) to metabolize PAHs. Aquatic organisms that readily metabolize PAHs do not accumulate PAHs in tissue and, thus, do not pose a potential source of exposure for avian species. Bivalves have a poorly developed capability for metabolizing and eliminating PAHs and may show some bioaccumulation. Fish and crustaceans, the major food sources of the ecological receptors, are typically much more efficient at metabolizing PAHs than bivalves and show bioaccumulation only in heavily polluted areas (Albers 1995). Concentrations of PAHs in sediments at Ward Cove are very low, with no individual PAH having a maximum concentration greater than 2 mg/kg dry weight. Furthermore, trophic level increases in accumulation of PAHs have not been observed in aquatic ecosystems (Albers 1995), which suggests that exposure of birds to PAHs through the food web is likely to be minimal.

Risk estimates can also be calculated for exposure of wildlife receptors to metals and PCDDs/Fs at the reference area based on concentrations in sediment samples collected at Moser Bay as part of the Phase 1 and 2 investigations (Tables A1-2 and A1-4 in Appendix A1). Exposure parameters are the same as in the models used to estimate risk at Ward Cove, except that the area use factor is set at one minus the Ward Cove area use factor, to reflect the fact that time not spent foraging in Ward Cove is spent foraging in other areas. Results for mammalian receptors show that no risk exists for harbor seals (Table 7-39) or river otters (Table 7-40). A risk of adverse effects resulting from exposure to cadmium exists for murrelets (Table 7-41) based on maximum or mean sediment concentrations, but no risk exists for cormorants (Table 7-42). A risk of adverse effects resulting from exposure to cadmium also exists for murrelets at Ward Cove.

TABLE 7-39. FOOD-WEB EXPOSURE MODEL CALCULATIONS FOR ASSESSMENT OF HYPOTHETICAL RISK TO HARBOR SEALS AT MOSER BAY

HARBOR SEAL (*Phoca vitulina*)

Body weight (kg)	56.7	Sediment ingestion rate (kg/day)	0.038
Food ingestion rate (kg/day)	1.898	Area use factor	0.97

Food Item	Percent of diet
Fish	84
Crabs	8
Shrimp	8

Chemical (max. concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Crab BSAF	Crab Concentration (mg/kg)	Crab Exposure (mg/kg-day)	Shrimp BSAF	Shrimp Concentration (mg/kg)	Shrimp Exposure (mg/kg-day)	Total Food Exposure (mg/kg-day)	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Arsenic	12	0.0078	0.1	0.14	0.0039	0.02	0.026	6.9×10^{-5}	NA			0.0040	0.012	0.13	0.094
Cadmium	1.5	9.7×10^{-4}	2.0	3.0	0.082	3.0	4.5	0.012	44	66.0	0.17	0.27	0.27	1.0	0.27
Mercury	0.10	6.5×10^{-5}	0.38	0.038	0.0010	0.13	0.013	3.4×10^{-5}	1.0	0.10	2.6×10^{-4}	0.0013	0.0014	0.032	0.044
Zinc	90.3	0.059	5.0	452	12.3	3.2	289	0.75	0.16	14.4	0.038	13.1	13.2	160	0.082
2,3,7,8-TCDD	1.6×10^{-6}	1.0×10^{-9}	1.0	6.5×10^{-6}	1.8×10^{-7}	1.0	8.6×10^{-7}	2.2×10^{-9}	0.70	8.7×10^{-7}	2.2×10^{-9}	1.8×10^{-7}	1.8×10^{-7}	1.0×10^{-6}	0.18

Chemical (mean concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Crab BSAF	Crab Concentration (mg/kg)	Crab Exposure (mg/kg-day)	Shrimp BSAF	Shrimp Concentration (mg/kg)	Shrimp Exposure (mg/kg-day)	Total Food Exposure (mg/kg-day)	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Arsenic	8.6	0.0056	0.12	0.10	0.0028	0.02	0.019	4.9×10^{-5}	NA			0.0029	0.0085	0.13	0.067
Cadmium	0.9	5.7×10^{-4}	2.0	1.8	0.048	3.0	2.6	0.0069	44	39	0.10	0.16	0.16	1.0	0.16
Mercury	0.08	5.2×10^{-5}	0.38	0.030	8.3×10^{-4}	0.13	0.010	2.7×10^{-5}	1.0	0.08	2.1×10^{-4}	0.0011	0.0011	0.032	0.035
Zinc	78.2	0.051	5.0	391	10.7	3.2	250	0.65	0.16	13	0.033	11.3	11.4	160	0.071
2,3,7,8-TCDD	1.4×10^{-6}	8.8×10^{-10}	1.0	5.5×10^{-6}	1.5×10^{-7}	1.0	7.3×10^{-7}	1.9×10^{-9}	0.70	7.3×10^{-7}	1.9×10^{-9}	1.5×10^{-7}	1.5×10^{-7}	1.0×10^{-6}	0.15

Note: All concentrations expressed as dry weight values.

BSAF - biota-sediment accumulation factor
TCDD - tetrachlorodibenzo-*p*-dioxin
TRV - toxicity reference value

TABLE 7-40. FOOD-WEB EXPOSURE MODEL CALCULATIONS FOR ASSESSMENT OF HYPOTHETICAL RISK TO RIVER OTTERS AT MOSER BAY

RIVER OTTER (*Lutra canadensis*)

Body weight (kg)	9.1	Sediment ingestion rate (kg/day)	0.0084
Food ingestion rate (kg/day)	0.422	Area use factor	0.75

Food Item	Percent of diet
Fish	83
Crabs	10
Bivalves	3.5
Gastropods	3.5

Chemical (max. concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Crab BSAF	Crab Concentration (mg/kg)	Crab Exposure (mg/kg-day)	Bivalve BSAF
Arsenic	12	0.0083	0.1	0.14	0.0042	0.022	0.026	9.2×10^{-5}	0.71
Cadmium	1.5	0.0010	2.0	3.0	0.087	3.0	4.5	0.016	7.5
Mercury	0.10	7.0×10^{-5}	0.38	0.038	0.0011	0.13	0.013	4.5×10^{-5}	4.5
Zinc	90.3	0.063	5.0	452	13.0	3.2	289	1.0	7.3
2,3,7,8-TCDD	1.6×10^{-6}	1.1×10^{-9}	1.0	6.5×10^{-6}	1.9×10^{-7}	1.0	8.6×10^{-7}	3.0×10^{-9}	0.90

Chemical (mean concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Crab BSAF	Crab Concentration (mg/kg)	Crab Exposure (mg/kg-day)	Bivalve BSAF
Arsenic	8.6	0.0060	0.1	0.10	0.0030	0.022	0.019	6.6×10^{-5}	0.71
Cadmium	0.9	0.0006	2.0	1.8	0.051	3.0	2.6	0.0092	7.5
Mercury	0.08	5.6×10^{-5}	0.38	0.030	8.8×10^{-4}	0.13	0.010	3.6×10^{-5}	4.5
Zinc	78.2	0.054	5.0	391	11.3	3.2	250	0.87	7.3
2,3,7,8-TCDD	1.4×10^{-6}	9.4×10^{-10}	1.0	5.5×10^{-6}	1.6×10^{-7}	1.0	7.3×10^{-7}	2.5×10^{-9}	0.90

TABLE 7-40. (cont.)

RIVER OTTER (*Lutra canadensis*)

Body weight (kg)	9.1	Sediment ingestion rate (kg/day)	0.0084
Food ingestion rate (kg/day)	0.422	Area use factor	0.75

Food Item	Percent of diet
Fish	83
Crabs	10
Bivalves	3.5
Gastropods	3.5

Chemical (max. concentration)	Bivalve Concentration (mg/kg)	Bivalve Exposure (mg/kg-day)	Gastropod BSAF	Gastropod Concentration (mg/kg)	Gastropod Exposure (mg/kg-day)	Total Food Exposure (mg/kg-day)	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Arsenic	0.85	0.0010	0.71	0.85	0.0010	0.0063	0.015	0.13	0.12
Cadmium	11.3	0.014	39	58.5	0.71	0.19	0.19	1.0	0.19
Mercury	0.45	5.5×10^{-4}	2.0	0.20	2.4×10^{-4}	0.0019	0.0020	0.032	0.063
Zinc	659	0.80	5.0	452	0.55	15.4	15.5	160	0.10
2,3,7,8-TCDD	2.2×10^{-6}	2.7×10^{-9}	0.90	1.2×10^{-6}	1.5×10^{-9}	2.0×10^{-7}	2.0×10^{-7}	1.0×10^{-6}	0.20

Chemical (mean concentration)	Bivalve Concentration (mg/kg)	Bivalve Exposure (mg/kg-day)	Gastropod BSAF	Gastropod Concentration (mg/kg)	Gastropod Exposure (mg/kg-day)	Total Food Exposure (mg/kg-day)	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Arsenic	0.61	0.0007	0.71	0.61	7.4×10^{-4}	0.0045	0.011	0.13	0.08
Cadmium	6.6	0.008	39	34.3	0.042	0.11	0.11	1.0	0.11
Mercury	0.36	4.4×10^{-4}	2.0	0.16	1.9×10^{-4}	0.0015	0.0016	0.032	0.050
Zinc	571	0.69	5.0	391	0.48	13.3	13.4	160	0.084
2,3,7,8-TCDD	1.9×10^{-6}	2.3×10^{-9}	0.90	1.0×10^{-6}	1.2×10^{-9}	1.6×10^{-7}	1.7×10^{-7}	1.0×10^{-6}	0.17

Note: All concentrations expressed as dry weight values.

BSAF - biota-sediment accumulation factor
TCDD - tetrachlorodibenzo-*p*-dioxin
TRV - toxicity reference value

TABLE 7-41. FOOD-WEB EXPOSURE MODEL CALCULATIONS FOR ASSESSMENT OF HYPOTHETICAL RISK TO MARBLED MURRELETS AT MOSER BAY

MARBLED MURRELET (*Brachyrampus marmoratus*)

Body weight (kg)	0.22	Sediment ingestion rate (kg/day)	0.0004
Food ingestion rate (kg/day)	0.022	Area use factor	0.9

Food Item	Percent of diet
Fish	80
Shrimp	20

Chemical (max. concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Shrimp BSAF	Shrimp Concentration (mg/kg)	Shrimp Exposure (mg/kg-day)	Total Food Exposure (mg/kg-day)	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Arsenic	12.0	0.022	0.12	0.14	0.010	NA			0.010	0.032	10	0.0032
Cadmium	1.5	0.0027	2.0	3.0	0.22	44.0	66.0	1.2	1.41	1.41	0.71	2.0
Mercury	0.10	1.8×10^{-4}	0.38	0.038	0.0027	1.0	0.10	0.0018	0.0045	0.0047	0.032	0.15
Zinc	90.3	0.16	5.0	452	32.6	0.16	14.4	0.26	32.9	33.0	98.8	0.33
2,3,7,8-TCDD	1.6×10^{-6}	2.9×10^{-9}	1.0	6.5×10^{-6}	4.7×10^{-7}	0.70	8.7×10^{-7}	1.6×10^{-8}	4.9×10^{-7}	4.9×10^{-7}	1.4×10^{-5}	0.035

Chemical (mean concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Shrimp BSAF	Shrimp Concentration (mg/kg)	Shrimp Exposure (mg/kg-day)	Total Food Exposure (mg/kg-day)	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Arsenic	8.6	0.016	0.12	0.10	0.0075	NA			0.0075	0.023	10	0.002
Cadmium	0.9	0.0016	2.0	1.8	0.13	44.0	38.7	0.7	0.83	0.83	0.71	1.2
Mercury	0.080	1.4×10^{-4}	0.38	0.030	0.0022	1.0	0.080	0.0014	0.0036	0.0038	0.032	0.12
Zinc	78.2	0.14	5.0	391	28.2	0.16	12.5	0.23	28.5	28.6	98.8	0.29
2,3,7,8-TCDD	1.4×10^{-6}	2.4×10^{-9}	1.0	5.5×10^{-6}	4.0×10^{-7}	0.70	7.3×10^{-7}	1.3×10^{-8}	4.1×10^{-7}	4.1×10^{-7}	1.4×10^{-5}	0.029

Note: All concentrations expressed as dry weight values.

BSAF - biota-sediment accumulation factor
TCDD - tetrachlorodibenzo-*p*-dioxin
TRV - toxicity reference value

TABLE 7-42. FOOD-WEB EXPOSURE MODEL CALCULATIONS FOR ASSESSMENT OF HYPOTHETICAL RISK TO PELAGIC CORMORANTS AT MOSER BAY

PELAGIC CORMORANT (*Phalacrocorax pelagicus*)

Body weight (kg)	1.7	Sediment ingestion rate (kg/day)	0.0019
Food ingestion rate (kg/day)	0.093	Area use factor	0.9

Food Item	Percent of diet
Fish	100

Chemical (max. concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Total Food Exposure (mg/kg-day)	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Arsenic	12.0	0.012	0.12	0.14	0.0071	0.0071	0.019	10	0.0019
Cadmium	1.5	0.0015	2.0	3.0	0.15	0.15	0.15	0.71	0.21
Mercury	0.10	9.9×10^{-5}	0.38	0.038	0.0019	0.0019	0.0020	0.032	0.062
Zinc	90.3	0.089	5.0	452	22.2	22.2	22.3	98.8	0.23
2,3,7,8-TCDD	1.6×10^{-6}	1.6×10^{-9}	1.0	6.5×10^{-6}	3.2×10^{-7}	3.2×10^{-7}	3.2×10^{-7}	1.4×10^{-5}	0.023

Chemical (mean concentration)	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish BSAF	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Total Food Exposure (mg/kg-day)	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Arsenic	8.6	0.0085	0.12	0.10	0.0051	0.0051	0.014	10	0.0014
Cadmium	0.9	8.7×10^{-4}	2.0	1.8	0.087	0.087	0.088	0.71	0.12
Mercury	0.080	7.9×10^{-5}	0.38	0.030	0.0015	0.0015	0.0016	0.032	0.05
Zinc	78.2	0.077	5.0	391	19.3	19.3	19.34	98.8	0.20
2,3,7,8-TCDD	1.4×10^{-6}	1.3×10^{-9}	1.0	5.5×10^{-6}	2.7×10^{-7}	2.7×10^{-7}	2.7×10^{-7}	1.4×10^{-5}	0.019

Note: All concentrations expressed as dry weight values.

BSAF - biota-sediment accumulation factor
TCDD - tetrachlorodibenzo-*p*-dioxin
TRV - toxicity reference value

7.2.4.2 Sources of Uncertainty

Several uncertainties exist in the estimation of risks for this ecological evaluation, and the actual risks may be higher or lower than predicted risks. The major uncertainties include the following:

- Concentrations of CoPCs in prey species are incompletely characterized and values used in the models are derived using literature BSAF values that may not be representative of Ward Cove
- Literature BSAF values were not found for all combinations of CoPCs and prey species
- Relative gastrointestinal absorption of CoPCs is assumed to be 100 percent
- Diet composition of receptors is derived from studies in the literature and may not be representative of individuals foraging at Ward Cove
- Exclusion of birds from the diet of river otters may underestimate the risk for this receptor
- Actual proportion of food obtained from Ward Cove relative to other habitats within the receptors' home range is not known
- TRVs derived from studies with laboratory species may not reflect interspecies differences in toxicity
- Uncertainty factors used for adjusting chronic LOAEL values to NOAELs in deriving TRVs may not accurately reflect differences on toxicity
- Background risk to receptors is overestimated if CoPC concentrations measured at Moser Bay are higher than concentrations elsewhere within the foraging range of the receptor
- Several chemicals found at elevated concentrations in Cove sediments (i.e., phenol, 4-methylphenol, benzoic acid, and pulp mill compounds) were not evaluated for risk in the food-web assessment.

The most important sources of uncertainty in the risk estimates are elaborated below.

Use of Literature BSAF Values—A limited amount of historical bioaccumulation data has been collected for several prey species at Ward Cove, as summarized in Appendix D. In particular, PCDDs/Fs have been measured in several fish species, crabs, mussels, and clams. This information can be used to evaluate the accuracy of risk estimates predicted by BSAF extrapolations. Spannagel (1991) reported TCDD TECs of 0.05 ng/kg wet weight in rockfish fillets and 0.27 ng/kg wet weight in Dungeness crab

muscle samples collected at Ward Cove. For the food-web exposure model, it is assumed that whole body concentrations are 10-fold greater than the concentrations reported in sectioned tissue (i.e., 0.5 ng/kg in rockfish and 2.7 ng/kg in crabs). In this model, rockfish and Dungeness crabs are considered representative of fish and crab species consumed by receptors at Ward Cove. As part of NPDES monitoring, mean concentrations of PCDDs/Fs were measured in mussel and clam tissues (EVS 1996). Mean tissue TCDD TECs in mussels and clams collected from Ward Cove stations were 0.73 ng/kg wet weight and 0.12 ng/kg wet weight, respectively. The average TCDD TEC for these two species (0.43 ng/kg) is considered representative of concentrations in bivalves consumed by receptors at Ward Cove. These reported tissue concentrations (converted to dry weight values) can be used in food-web exposure models in place of concentrations estimated on the basis of a BSAF extrapolation (Table 7-43). For species with no historical bioaccumulation data (shrimp and gastropods), default BSAF extrapolations are used. Recalculation of estimated risk to receptors using available site tissue data and maximum sediment PCDD/F concentrations reveals that hazard quotients decrease approximately 30-fold for harbor seals, river otters, and marbled murrelets and approximately 70-fold for pelagic cormorants (Table 7-43). In all cases, the hazard quotient is substantially lower than 1.0. This recalculation, although based on only a limited amount of tissue data, suggests that risk of adverse effects to receptors resulting from exposure to CoPCs through the food web may be overestimated by almost 2 orders of magnitude when using BSAF extrapolations to estimate tissue chemical concentrations.

Concentrations of PCDDs/Fs, shown as TECs, have also been measured in aquatic species inhabiting areas near the APC pulp mill in Sitka, Alaska, as part of an ecological risk assessment conducted for that site (Foster Wheeler 1997). As described in Section 2.2.3.2 and Appendix D2, similarly low concentrations of PCDDs/Fs as seen in biota collected at Ward Cove were reported in rockfish fillets and mussels collected from West Sawmill Cove in the APC investigation. Results from West Sawmill Cove were judged to be most comparable with Ward Cove because of the similar ranges of PCDDs/Fs and TOC concentrations in sediments from both locations. The one rockfish fillet analyzed from West Sawmill Cove had a total concentration of PCDDs/Fs of 0.004 ng/kg wet weight, which is approximately an order of magnitude lower than the concentration reported by Crook (1995, pers. comm.) for rockfish fillets from Ward Cove. Four mussel samples collected from West Sawmill Cove had concentrations that ranged from 0.37 to 4.5 ng/kg wet weight, with a mean value of 2.2 ng/kg. This range of concentrations encompasses the mean concentration in mussels from Ward Cove (0.73 ng/kg). Dungeness crabs were not collected from West Sawmill Cove, but crabs collected at two other marine locations near the APC site had total PCDD/F concentrations of 0.04 and 0.36 ng/kg wet weight in muscle tissue. These concentrations are similar to levels measured in muscle tissue of Dungeness crabs collected from Ward Cove (0.27 ng/kg).

The finding of similarly low concentrations of PCDDs/Fs in tissue collected in the APC investigation as in tissue samples collected from Ward Cove provides further evidence that bioaccumulation of PCDDs/Fs is limited in aquatic biota. This finding also supports that conclusion that BSAF extrapolations overestimate tissue concentrations in prey species, which results in overestimation of risk to avian and mammalian receptors.

TABLE 7-43. FOOD-WEB EXPOSURE MODEL CALCULATIONS FOR ASSESSMENT OF HYPOTHETICAL RISK OF PCDDs/Fs TO WILDLIFE RECEPTORS AT WARD COVE BASED ON MEASURED TISSUE CoPC CONCENTRATIONS

Receptor Species	Sediment Concentration (mg/kg)	Sediment Exposure (mg/kg-day)	Fish Concentration (mg/kg)	Fish Exposure (mg/kg-day)	Crab Concentration (mg/kg)	Crab Exposure (mg/kg-day)	Shrimp Concentration (mg/kg)	Shrimp Exposure (mg/kg-day)
Harbor seal	4.6×10^{-5}	9.2×10^{-10}	2.0×10^{-6}	1.7×10^{-9}	1.0×10^{-5}	8.3×10^{-10}	2.5×10^{-5}	2.0×10^{-9}
River otter	4.6×10^{-5}	1.1×10^{-8}	2.0×10^{-6}	1.9×10^{-8}	1.0×10^{-5}	1.2×10^{-8}		NA
Marbled murrelet	4.6×10^{-5}	9.2×10^{-9}	2.0×10^{-6}	1.6×10^{-8}		NA	2.5×10^{-5}	5.0×10^{-8}
Pelagic cormorant	4.6×10^{-5}	5.0×10^{-9}	2.0×10^{-6}	1.1×10^{-8}		NA		NA

TABLE 7-43. (cont.)

Receptor Species	Bivalve Concentration (mg/kg)	Bivalve Exposure (mg/kg-day)	Gastropod Concentration (mg/kg)	Gastropod Exposure (mg/kg-day)	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	Hazard Quotient
Harbor seal		NA		NA	5.4×10^{-9}	1.0×10^{-6}	0.0054
River otter	2.6×10^{-6}	1.1×10^{-9}	3.5×10^{-5}	1.4×10^{-8}	5.7×10^{-8}	1.0×10^{-6}	0.057
Marbled murrelet		NA		NA	7.5×10^{-8}	1.4×10^{-5}	0.0053
Pelagic cormorant		NA		NA	1.6×10^{-8}	1.4×10^{-5}	0.0011

Note: All concentrations expressed as dry weight values.
CoPC - chemical of potential concern
NA - not applicable
PCDD/F - polychlorinated dibenzo-*p*-dioxin and polychlorinated dibenzofuran
TRV - toxicity reference value

Historical data are available at Ward Cove on bioaccumulation of mercury for mussels and clams, as summarized in Appendix D. Because bivalves constitute only a minor component of the diet of one species (3.5 percent of the diet for river otters), it is not possible to recalculate estimated risk to receptors based on measured tissue data. However, the historical data can be used to evaluate the accuracy of body burden estimates derived by using a BSAF approach. Total mercury data for sediment from the five Ward Cove monitoring stations (EVS 1996) were used to predict body burdens in mussels and clams based on the BSAF equation presented in Table 4-6 of this report. Predicted concentrations range from 2- to 8-fold higher than concentrations actually measured in mussels and clams (Table 7-44), which indicates that prey body burdens, and hence risk to wildlife receptors, are overestimated when using the BSAF approach. In addition, the bioaccumulation monitoring study (EVS 1996) found that concentrations of methylmercury, the toxicologically important form of mercury, were very low in mussels and clams, with mean values of 0.0005 and 0.0007 mg/kg wet weight, respectively. Thus, the risk to receptors that consume these prey items is less than reported in this risk assessment, because the TRVs are based on intake of methylmercury, whereas the majority of ingested mercury is in a less toxic, nonmethylated form. Based on these limited bivalve data, it is likely that similar overestimates are made for other species when using a BSAF approach to estimate body burdens and with the assumption that all mercury exists in a methylated form.

Interspecies Differences in Toxicity—TRVs were not available for the wildlife species evaluated in the risk assessment, and values derived for other species were used instead. This approach increases uncertainty because the magnitude and direction (more or less sensitive) of differences among the species in sensitivity to the toxic effects of the CoPCs are not known. For most CoPCs, more than one TRV is available, and the various TRVs reflect the potential interspecies differences in sensitivity. For example, in an earlier risk assessment for Ward Cove, ENSR (1996a) derived an avian TRV for cadmium from a 90-day exposure study for mallard ducks (White and Finley 1978). That study evaluated effects on reproduction and determined a NOAEL of 1.45 mg/kg-day. This report used a 48-week study on chickens by Leach et al. (1979) to derive the TRV. Leach et al. (1979) examined the same endpoint (egg production) as White and Finley (1978) and produced a NOAEL of 0.71 mg/kg-day. Because Leach et al. (1979) used a longer exposure period than White and Finley (1978), this study was considered to provide a more realistic evaluation of chronic exposure.

To account for differences in toxicity to CoPCs among species, ENSR (1996a) applied uncertainty factors based on the taxonomic divergence between test species and the wildlife receptors evaluated in the food-web models. In that risk assessment, an uncertainty factor of 4 was used for cormorants (birds) and mammals, and a factor of 8 was used for marbled murrelets. The uncertainty factors were determined as follows: a factor of 2 was applied to extrapolate between different families of the same order, a separate factor of 2 was applied to extrapolate between different orders of the same class, and a third factor of 2 was applied for the marbled murrelet because it was listed as a candidate

TABLE 7-44. COMPARISON OF PREDICTED AND MEASURED MERCURY CONCENTRATIONS IN MUSSELS AND CLAMS FROM WARD COVE

Station	Sediment Mercury Concentration (mg/kg, dry weight)[a]	Mussels Mercury Concentration (mg/kg, wet weight) Predicted[b]	Mussels Mercury Concentration (mg/kg, wet weight) Measured[a]	Mussels Predicted/ Measured	Clams Mercury Concentration (mg/kg, wet weight) Predicted[b]	Clams Mercury Concentration (mg/kg, wet weight) Measured[a]	Clams Predicted/ Measured
WC-01	0.08	0.067	0.013	5.1			
WC-02	0.13	0.10	0.013	8.1	0.10	0.017	6.1
WC-03	0.08	0.068	0.015	4.5	0.068	0.026	2.6
WC-04	0.11	0.091	0.015	5.9	0.091	0.011	8.2
WC-05	0.04	0.031	0.014	2.2	0.031	0.018	1.7

[a] Data from EVS (1996).

[b] Based on algorithm presented in Table 4-6 of this report.

species at that time (the bird is no longer a listed species for the Ward Cove area). This uncertainty factor approach is designed to ensure a conservative result. The magnitude of the interspecies uncertainty factor is proportional to the perceived uncertainty as gauged by the phylogenetic distance between the test and receptor species.

Interspecies uncertainty factors were not applied in this risk assessment. This approach is consistent with other ecological risk assessments that have been performed at sediment sites in Region 10.

EPA Region 10 guidelines for ecological risk assessment indicate that an uncertainty factor approach is useful when effects data are available, but knowledge on the underlying toxicity mechanisms is lacking (U.S. EPA 1997c). EPA Region 10 guidelines suggest that a 2-fold uncertainty factor may be applied to extrapolate between different families of the same order and a separate 2-fold uncertainty factor may be applied to extrapolate between different orders of the same class (U.S. EPA 1997c). In this risk assessment, the avian (or mammalian) species evaluated belong to different orders than the avian (or mammalian) species used to derive the TRV. Thus, if the risk assessment had used an uncertainty factor scaling approach as described by EPA Region 10 guidance (i.e., 2-fold factors for different family–same order and different order–same class), hazard quotients for receptors in Ward Cove would be 4-fold higher than reported in Tables 7-35 to 7-38, based on either maximum or mean CoPC concentrations in sediment. In all cases, however, the hazard quotients would be less than 10, and considering the uncertainty surrounding derivation of hazard quotients, risk to receptors is not likely to be significant.

Extrapolation from a LOAEL to NOAEL for Derivation of TRVs— NOAEL data were available for all chemicals, except in two cases. For mammalian receptors, the chronic LOAELs for arsenic and benzo[a]pyrene were adjusted to a NOAEL using an uncertainty factor of 10 in accordance with the guidance of U.S. EPA (1994b). This uncertainty factor of 10 is more conservative than the uncertainty factor of 5 that is recommended in EPA Region 10 guidelines (U.S. EPA 1997c) for extrapolating from a chronic LOAEL to a NOAEL. If a factor of 5, rather than 10, had been used in the ecological risk assessment, the reported hazard quotients for arsenic and benzo[a]pyrene for mammalian receptors could overestimate risk by a factor of 2.

Site Use Factors—The actual proportion of food that receptors obtain from Ward Cove relative to other habitats within their home range, a reflection of site use factors, is not known. However, the food-web evaluation is a baseline ecological risk assessment that uses realistic estimates of the proportion of time each species is expected to occur in Ward Cove. The risk assessment appropriately avoids the use of conservative default exposure assumptions that are characteristic of a screening-level risk assessment.

Information about site use by receptors was derived from the scientific literature or through discussions with knowledgeable Alaskan authorities. Whenever possible,

species-specific and location-specific (i.e., studies in Alaska) information was used. When a range of values was available, values that are realistic but that increase the extent of exposure were chosen. For example:

- To focus the presence of harbor seals at Ward Cove, it was assumed that a haul-out site was present in the Cove, although no such sites are known from this area. Additionally, we assumed that seals forage within 5 km of the haul-out site even though an ADFG official had indicated to Exponent in a telephone conversation that seals may forage up to 50 km away from the haul-out site.
- Ward Cove was assumed to represent 25 percent of the home range of river otters. However, based on a study of river otters in southeastern Alaska, the area of Ward Cove represents only about 12 percent of the smallest recorded home range size.
- Ward Cove was assumed to constitute 10 percent of the home range of marbled murrelets, although data from Alaska show that nesting birds have a foraging range up to 30 km from the nest site and a minimum home range of 11,900 ha, which is more than 100 times the area of Ward Cove.

The assumptions outlined above indicate that, although precise area use factors are not available, the values used in the food-web models are reasonable and not likely to underestimate risk for wildlife receptors using Ward Cove.

Background Risk—For marbled murrelets, risk of adverse effects resulting from exposure to background cadmium concentrations is inferred based on CoPC concentrations in sediment samples collected at Moser Bay. This may be attributable, in part, to concentrations in Moser Bay overestimating background concentrations elsewhere within the foraging range. This issue is particularly important because background areas are assumed to account for 90 percent of the foraging range of murrelets. Risk might also be inferred if BSAF extrapolations overestimate tissue chemical burdens as illustrated above for Ward Cove, or if the TRV used in the exposure model equation is not accurate.

Unevaluated CoPCs—Several chemicals found at elevated concentrations in Ward Cove sediment (i.e., phenol, 4-methylphenol, benzoic acid, and pulp mill compounds) were not evaluated for risk in the food-web assessment. The distribution of these compounds was highly localized within Ward Cove, and thus they are not likely to be of concern for the mammalian and avian receptors that have expansive foraging ranges both within the Cove and in surrounding areas. Little information exists in the literature regarding the bioaccumulation potential of these compounds, but they have rarely been addressed in food-web assessments in other studies, and they are not generally considered

compounds that may pose a risk via accumulation through the food web. Thus, although these CoPCs were not evaluated, their limited distribution and low likelihood of bioaccumulation suggest that they are unlikely to represent a significant risk for wildlife receptors in Ward Cove.

7.2.5 Supplemental Evaluation of TCDD Accumulation through Maternal-Egg Transfer in Fish

As an additional evaluation at the request of EPA and ADEC to determine if dioxin concentrations in Ward Cove sediments are protective of bioaccumulative effects to higher trophic-level organisms, potential effects of TCDD on early life stages of fish can be evaluated using a simple maternal-egg transfer model. Accumulation of TCDD in fish eggs largely reflects maternal transfer. Therefore, the concentration of TCDD in eggs was related to the concentration in their mothers, which in turn can be related to sediment concentrations using a BSAF approach. Because early life stages of fish (eggs and embryos) are generally more sensitive to the effects of TCDD than older individuals (Sijm and Opperhuizen 1996), this approach should also be protective of adult benthic and demersal fishes.

Studies cited in U.S. EPA (1993a) indicate that, on a wet weight basis, the TCDD concentration in lake trout eggs was about 30–40 percent of the maternal concentration. Walker et al. (1991) reported a NOAEL of 3.5×10^{-5} mg/kg wet weight TCDD TEC for mortality in lake trout fish eggs. Using a maternal-egg transfer ratio of 0.40 (40 percent), this no-effect tissue concentration in eggs corresponds to about 8.5×10^{-5} mg/kg wet weight TCDD TEC in the parent fish. Based on the lipid concentration of 0.102 used for the Ward Cove ecological risk assessment (Table 4-6), the corresponding maternal lipid-normalized TCDD TEC is 8.3×10^{-4} mg/kg. Dividing the lipid-normalized concentration by 1.04 (the BSAF value for fish that was used in the ecological risk assessment; Table 4-5) results in a TOC-normalized sediment TCDD TEC of 8.0×10^{-4} mg/kg, which would be protective of fishes. The maximum TOC-normalized TCDD TEC in Ward Cove sediments is 4.6×10^{-4} mg/kg, based on a maximum sediment dry weight concentration of 4.6×10^{-5} mg/kg and 10 percent TOC (Table 4-5). The maximum TOC-normalized concentration is below the calculated threshold criterion.

The predicted sediment quality value of 8.0×10^{-4} mg/kg for TCDD TEC is largely influenced by the conservative BSAF value used to predict the relationship between concentrations in sediment and in fish tissue. Steady-state BSAF values for TCDD for Lake Ontario fish species range from 0.03 to 0.20 (U.S. EPA 1993a). Substituting these values for the BSAF used in the Ward Cove risk assessment results in sediment threshold values between 4.2×10^{-3} and 2.8×10^{-2} mg/kg, which greatly exceed concentrations recorded in Ward Cove. These results indicate that concentrations of PCDDs/Fs in Ward Cove sediments should not be of concern for fish or other higher trophic-level organisms.

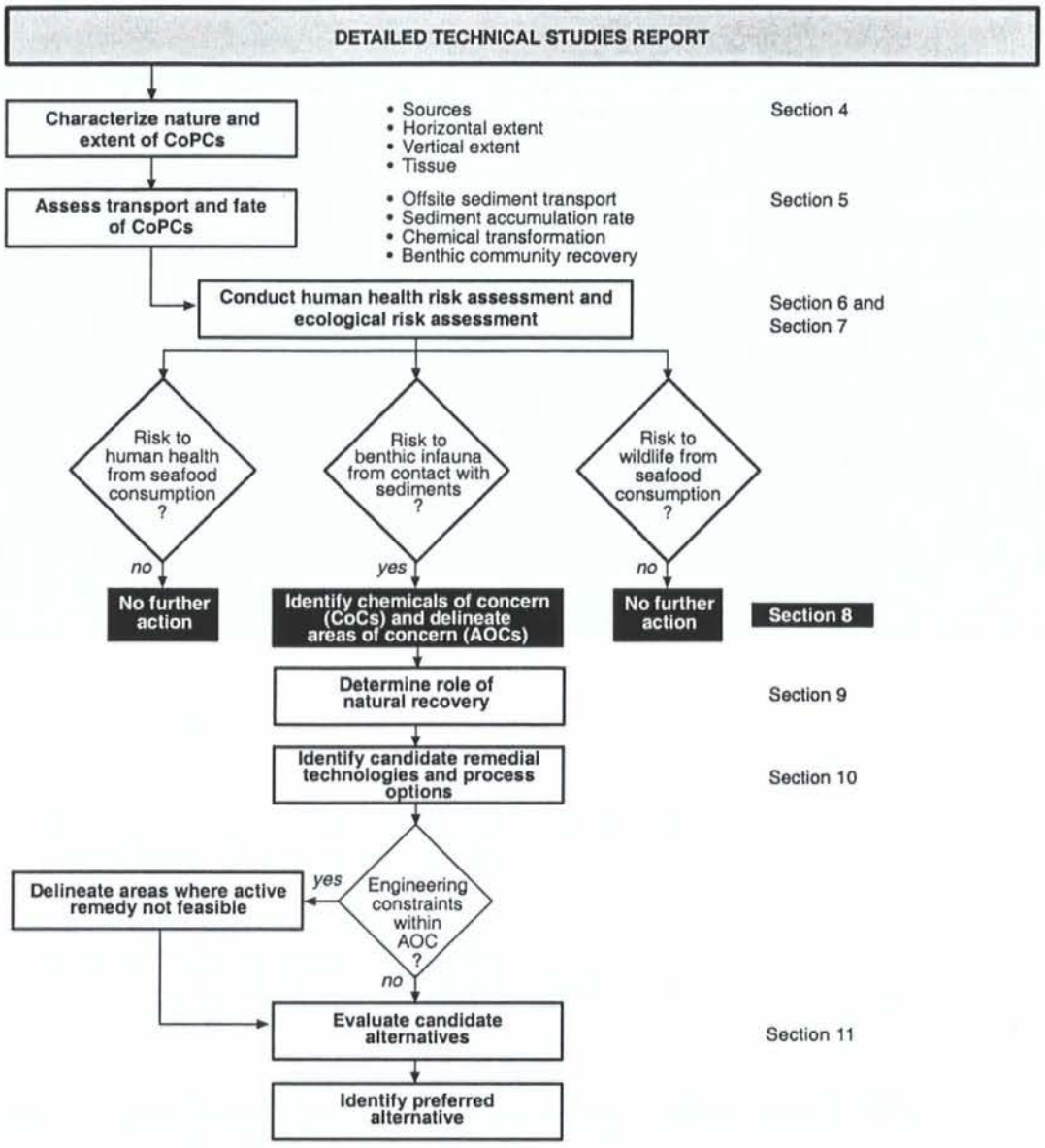
In addition to the BSAF used to calculate fish tissue concentrations, several other sources of uncertainty could affect the results of the fish egg TCDD accumulation model. First,

the maternal-egg TCDD transfer ratio is dependent on lipid levels in adult fish and eggs, and these values are not known for Ward Cove fishes. In addition, there are no site-specific data on TCDD concentrations in Ward Cove fishes and fish eggs that can be used as a check against the concentrations predicted by the transfer model. Second, the NOAEL used as the TRV is based on mortality for lake trout eggs and, because of possible interspecies differences, this value might not be accurate for Ward Cove fishes. However, U.S. EPA (1993a) also uses this value without incorporating any uncertainty factors in an assessment of effects of TCDD on aquatic life. Third, mammalian TEF values, not fish-specific TEF values, were used to calculate TCDD TECs in sediments, and thus the value might not represent the toxic potential of TCDD in sediments to fish. However, if fish-specific TEFs from the new World Health Organization TEF scheme (which has not yet been formally adopted by EPA) were used, the maximum TCDD TEC in sediment decreases slightly from 46 to 40 ng/kg. This change would increase, not decrease, the difference between the maximum TCDD concentration in Ward Cove and the calculated threshold criterion.

7.2.6 Conclusions

The use of maximum CoPC concentrations in food-web models and conservative, yet ecologically realistic, assumptions regarding life history characteristics of receptors provides a relatively high level of confidence in the results of the risk assessment. The models based on predicted prey tissue concentrations using a BSAF approach indicate a potential risk of adverse effects resulting from exposure of river otters to PCDDs/Fs and from exposure of marbled murrelets to cadmium. However, evaluation of some of the uncertainties associated with the assessment suggest that these risks may be overestimated in the modeling approach used for Ward Cove. Recalculations of hazard quotients using historical tissue data show that the models may overestimate risk from exposure to PCDDs/Fs from 30- to 70-fold, and bioaccumulation data for mercury suggest that risk from exposure to metals may be overestimated up to 10-fold. If true, these recalculations would result in hazard quotients substantially less than 1.0 for PCDDs/Fs and cadmium for mammalian and avian receptors. Exposure models, when evaluated in consideration of the identified uncertainties in the modeling approach, suggest that no risks of adverse effects result from exposure to CoPCs through the food web for mammalian and avian receptors at Ward Cove. In addition, a maternal-egg model used to evaluate potential effects of TCDD TECs on early life stages of fish indicates that concentrations of PCDDs/Fs in Ward Cove sediments should not be of concern for fish or other higher trophic-level organisms.

8. DELINEATION OF AREA OF CONCERN



The baseline human health risk assessment and the ecological evaluation culminate in the identification of the area of concern (AOC). The AOC represents that area and/or volume of sediment where active remedial action may be warranted. The AOC is subjected to an analysis of engineering feasibility (Section 10) and is the focus of the candidate remedial alternatives (Section 11). In this section, information from the baseline human heath risk assessment and the ecological evaluation is synthesized to delineate the sediment AOC in Ward Cove.

The chemicals present in the sediments of Ward Cove were evaluated to determine potential human health and ecological risk from direct exposure and exposure via the food web. The risk evaluations considered in detail three major types of exposure pathways:

- Human exposure to CoPCs through seafood consumption
- Wildlife (bird and mammal) exposure to CoPCs through seafood consumption
- Benthic organism exposure to CoPCs through direct contact.

Additional secondary exposure pathways (e.g., direct contact with sediments by humans) were evaluated as part of the sensitivity analyses.

The risks associated with the first two types of exposure were determined to fall within acceptable limits when considered in the context of the conservative modeling assumptions. The maximum estimated concentration of PCDDs/Fs (TEC) in seafood exceeded the risk-based concentration, which could qualify this group of chemicals as a CoC. However, measured concentrations in seafood, a more reliable indicator of risk, did not exceed risk-based concentrations; thus, PCDDs/Fs were not identified as a CoC. However, sediment toxicity was present in portions of the Cove at levels that warrant consideration for sediment remediation.

Superfund regulations require that remedial action objectives (RAOs) be established for a site (40 CFR 300.430(e)(2)(i)). RAOs provide a general description of what the cleanup will accomplish (e.g., protect the environment by reducing sediment toxicity levels, as appropriate). The RAOs are EPA's goals for addressing risk at the site. Thus, in Superfund, RAOs are established only for those pathways for which risk has been identified as exceeding acceptable levels. RAOs were identified for Ward Cove based on the results of the ecological evaluation and the sediment toxicity that appears to be related to non-persistent by-products from the decomposition of organic matter and wood debris. The RAOs for Ward Cove address the sediment toxicity for the benthic community. The RAOs for Ward Cove sediments are to:

- Reduce sediment toxicity
- Enhance recolonization of surface sediments to support a healthy benthic infaunal community with multiple taxonomic groups
- Provide a benthic macroinvertebrate community that constitutes an abundant food source to larger invertebrates and fishes.

In the following sections, the AOC is delineated on the basis of the sediment chemistry and sediment toxicity data presented in Section 7.1. The expanded site investigation data were not used to delineate the AOC because of problems associated with the accuracy of the station locations (U.S. EPA 1998f). As documented in Section 7.1, the most likely

causative agents of sediment toxicity in Ward Cove appear to be ammonia, sulfide, and 4-methylphenol, the CoCs for Ward Cove sediments. However, to be conservative, the delineation of the AOC is based on all Phase 2 CoPCs, except total sulfide (i.e., TOC, total ammonia, BOD, COD, 4-methylphenol). The delineation was not based on total sulfide because no sediment quality values were available for that chemical.

With respect to sulfide and sediment toxicity, the key variable is the concentration of hydrogen sulfide in pore water. As described previously, concentrations of hydrogen sulfide in pore water are not necessarily directly related to concentrations of total sulfide in bulk sediments, because other variables (e.g., pH) determine the fraction of total sulfide that is present as hydrogen sulfide in pore water. In the present study, concentrations of sulfide in pore water were determined only for the eight stations evaluated using the specialized toxicity tests. It therefore was not possible to determine the spatial extent of porewater sulfide concentrations in Ward Cove or to develop site-specific sediment quality values for that variable. Nevertheless, the high concentrations of total sulfide in bulk sediments found in parts of the Cove (Figure 4-6) indicate that it is likely that porewater concentrations of hydrogen sulfide are also elevated in those areas and are contributing to any observed sediment toxicity.

The delineation of the AOC relies on a weight-of-evidence approach for identifying stations at which unacceptable ecological risks are posed. Use of a weight-of-evidence approach is recommended by U.S. EPA (1996e, 1997b). Under this approach, the inclusion of a station as part of an AOC requires agreement among multiple chemical and/or biological indicators that unacceptable risk exists at that station. The requirement for multiple lines of evidence enhances confidence that an unacceptable risk is truly present at a station. By contrast, delineation of an AOC using results of single indicators can be biased by potential artifacts encountered with the indicators. For example, a significant toxicity response at a station in the absence of any chemicals that exceed sediment quality values may be the result of the test organisms being stressed during handling and testing or being sensitive to nonchemical factors (e.g., sediment grain size distribution). From another standpoint, the exceedance of a sediment quality value by a chemical in the absence of corroboration by a significant toxicity response could mean that the chemical is not sufficiently bioavailable to result in toxicity.

Although exceedances of SQS and $WCSQV_{(1)}$ values are discussed below, the AOC is delineated based on exceedances of MCUL and $WCSQV_{(2)}$ values because the latter values provide a greater degree of confidence that ecological risks are present. In this manner, it will be ensured that the evaluation of remedial options and any future remediation costs will be focused on those parts of Ward Cove that pose the greatest ecological risk.

8.1 EXCEEDANCES OF SQS AND $WCSQV_{(1)}$ VALUES

In Figure 8-1, the sediment concentrations of CoPCs and the sediment toxicity results that exceed their respective SQS or $WCSQV_{(1)}$ values are identified for each of the 44 stations sampled in Ward Cove during the Phase 1 investigation in 1996 or the Phase 2

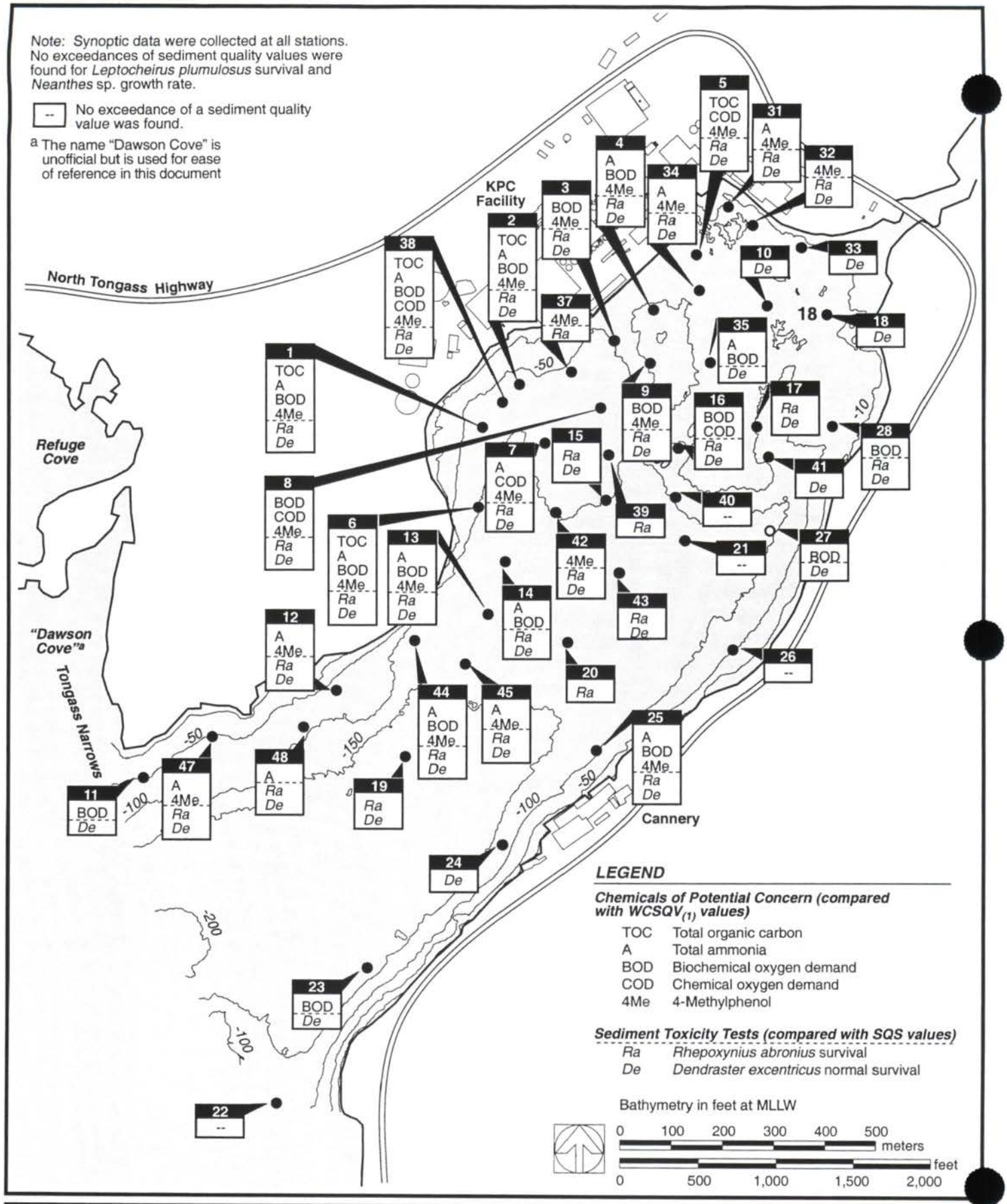


Figure 8-1. Distribution of exceedances of SQS and WCSQV$_{(1)}$ values in Ward Cove in 1996 and 1997.

investigation in 1997. Most exceedances of multiple chemical and/or biological indicators were confined to stations located offshore from the KPC facility and southwest of the facility along the northern shoreline of the Cove. Multiple exceedances were also found immediately offshore from the fish cannery on the southern shoreline of the Cove.

Outside of the two general areas described above, exceedances of SQS and $WCSQV_{(1)}$ values for single or multiple indicators were less common. For example, no exceedances were found at four stations (Stations 21, 22, 26, and 40), and only a single exceedance of an indicator was found at seven stations (Stations 10, 18, 20, 24, 33, 39, and 41). The echinoderm test (based on normal survival) was responsible for the single SQS exceedance at five of the seven stations at which a single toxicity test exceeded its SQS.

The echinoderm embryo test (based on normal survival) was responsible for singularly identifying the most stations as exceeding their respective SQS or $WCSQV_{(1)}$ values. This characteristic of the echinoderm test casts doubt on whether its singular exceedance at farfield stations is meaningful. The lack of corroboration of the echinoderm test exceedances by any of the sediment chemical exceedances or exceedances by any of the other three sediment toxicity tests implies that the echinoderm test may not be responding to CoPC concentrations at those stations. Several aspects of the echinoderm embryo test make it a less robust tool for determining the AOC for Ward Cove. First, there is an unquantified error component associated with the determination of the *percent normal survival endpoint*, as discussed earlier in this section. Second, higher variability has been observed for this test as compared to other tests used to characterize sediment toxicity. This higher variability prompted Washington State and the Puget Sound Dredged Material Management Program agencies to require that statistical comparisons using this test be conducted at a significance level of $P \leq 0.10$, whereas the results of all other toxicity tests used for regulatory purposes in Washington State are evaluated at $P \leq 0.05$ (Michelsen 1996). Moreover, at the national level, U.S. EPA (1998a) did not select the echinoderm (or any other larval test) for implementing its contaminated sediment management strategy.

As previously described, the delineation of the AOC in Ward Cove relies on a weight-of-evidence approach for identifying stations at which unacceptable ecological risks are posed. Given the concerns regarding the echinoderm test expressed at the national level and the higher variability and unquantified error associated with the percent normal survival endpoint, and given site-specific results showing the uncorroborated performance of the test (relative to other environmental indicators) at numerous stations in Ward Cove, this test will not be used to singularly identify potential sediment problems in Ward Cove.

It is possible that survival was substantially underestimated at numerous stations, thus artificially lowering the estimates of percent normal survival. This potential underestimate of survival is consistent with the fact that significant effects based on the normality (or abnormality) endpoint of the echinoderm embryo test were found at only six of the 44 stations evaluated in the Cove. All six of those stations were located immediately offshore from the KPC mill, where multiple environmental indicators identified potential

problems. Additional information on the performance of the echinoderm test and interpretation of test results is provided in U.S. EPA (1999a).

The distribution of exceedances of $WCSQV_{(1)}$ values in Ward Cove based on the CoPC concentrations found during NPDES monitoring in 1994 and 1995 (Tables 7-26 through 7-29) is presented in Figure 8-2. The pattern of exceedances found for the historical data (based on the top 2 cm of sediment) is similar to the pattern found for the data collected in the Phase 1 investigation in 1996 (based on the top 10 cm of sediment). Exceedances of $WCSQV_{(1)}$ were found offshore from the KPC facility, downcurrent from the facility along the northern shoreline of the Cove, and offshore from the fish cannery.

8.2 EXCEEDANCES OF MCUL AND $WCSQV_{(2)}$ VALUES

In Figure 8-3, the sediment concentrations of CoPCs and the sediment toxicity results that exceed their respective MCUL or $WCSQV_{(2)}$ values are identified for each of the 44 stations sampled in Ward Cove during the Phase 1 investigation in 1996 or the Phase 2 investigation in 1997. As described for the exceedances of SQS and $WCSQV_{(1)}$ values in the previous section, most exceedances of MCUL and $WCSQV_{(2)}$ values were located offshore or southwest of the KPC facility or immediately offshore from the fish cannery. In contrast to the spatial pattern of SQS and $WCSQV_{(1)}$ exceedances, many more farfield stations (19) exhibited no MCUL or $WCSQV_{(2)}$ exceedances (i.e., Stations 15, 20 to 24, 26, 28, 40, and 43) or only single MCUL exceedances (Stations 10, 11, 18, 19, 27, 33, 35, 39, and 41). In addition, the echinoderm embryo test (based on normal survival) accounted for the MCUL exceedance at seven of the nine farfield stations having only single exceedances.

The distribution of exceedances of $WCSQV_{(2)}$ values in Ward Cove based on the CoPC concentrations found during NPDES monitoring in 1994 and 1995 (Tables 7-26 through 7-29) is presented in Figure 8-4. In general, the pattern of exceedances found in the historical data is similar to the pattern found in the Phase 1 investigation in 1996. Exceedances of $WCSQV_{(2)}$ values were found offshore from the KPC facility, southwest of the facility along the northern shoreline of the Cove, and offshore from the fish cannery.

8.3 DELINEATION OF AREA OF CONCERN

The AOC identified for more detailed evaluation was delineated on the basis of the kinds of exceedances of sediment quality values found at individual stations (Sections 8.1 and 8.2). The AOC was delineated using a weight-of-evidence approach that requires multiple lines of evidence for identifying stations at which unacceptable ecological risks are posed. This approach is currently recommended by EPA for sediment quality assessments throughout the United States (U.S. EPA 1997a,b; 1998d,e). The underlying premise of the approach is that every kind of environmental indicator has limitations and, therefore, no one indicator can be relied on alone to provide conclusive evidence of sediment toxicity. As part of the delineation process for Ward Cove, stations were grouped

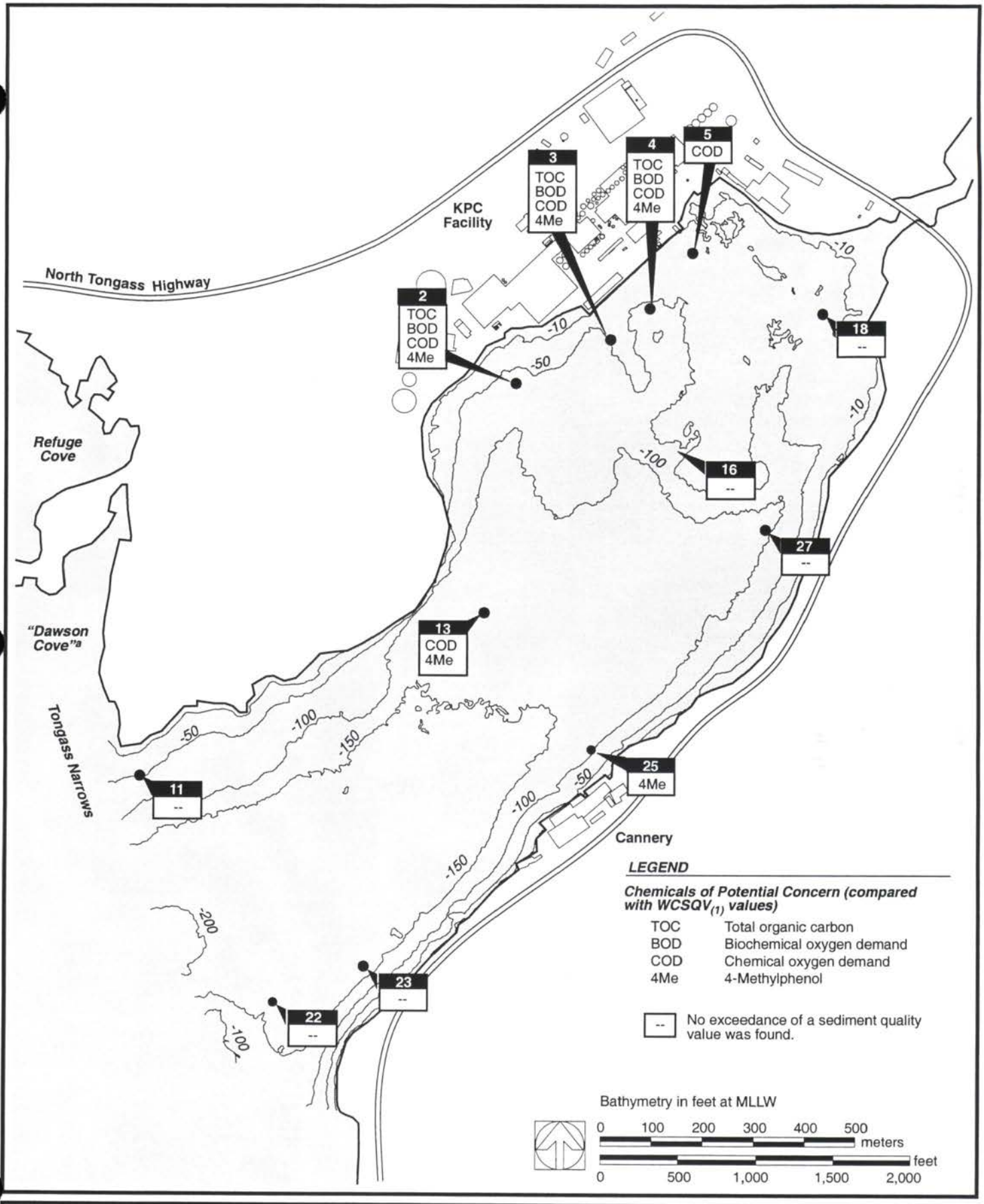


Figure 8-2. Distribution of exceedances of $WCSQV_{(1)}$ values in Ward Cove in 1994-1995.

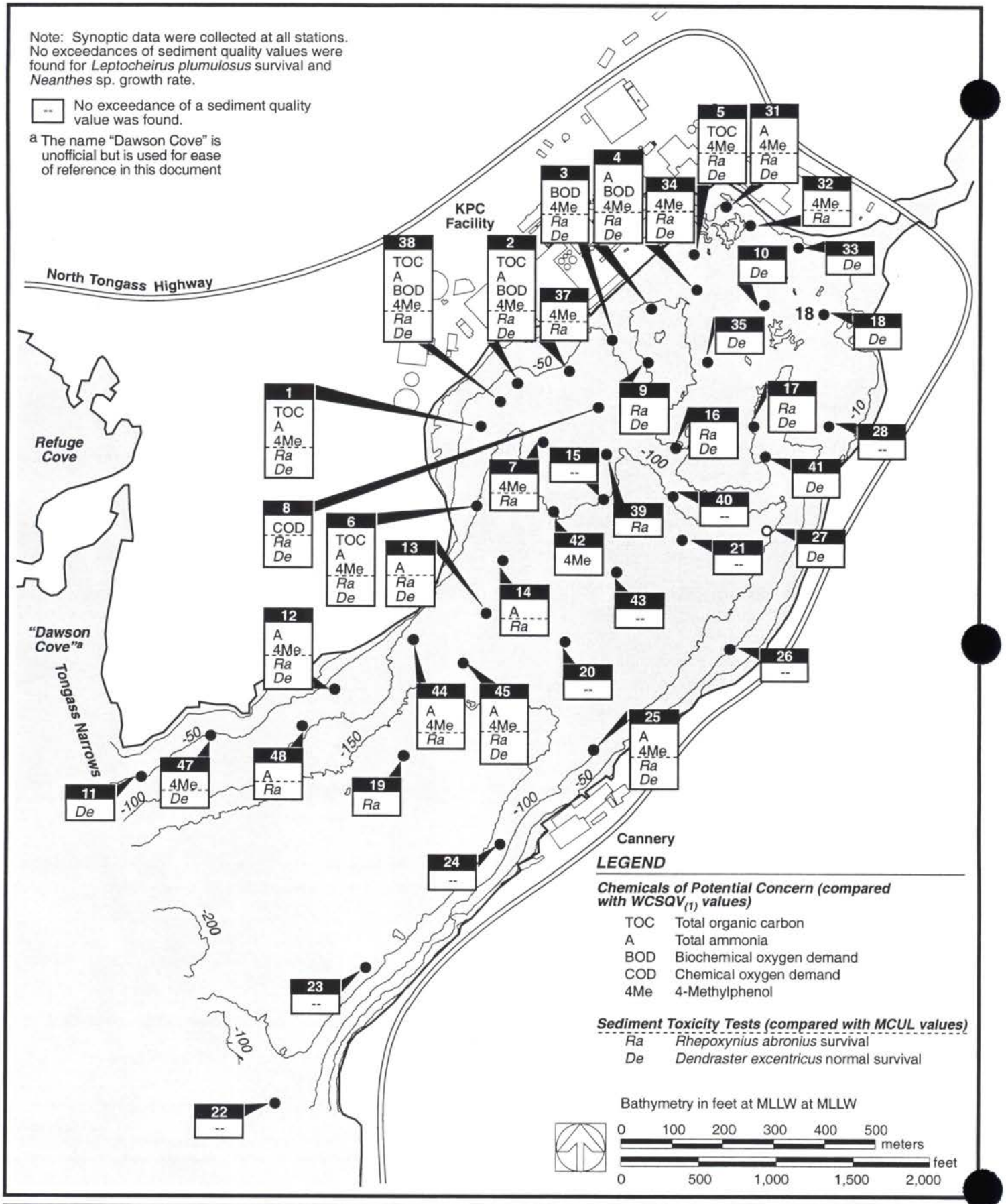


Figure 8-3. Distribution of exceedances of MCUL and $WCSQV_{(2)}$ values in Ward Cove in 1996 and 1997.

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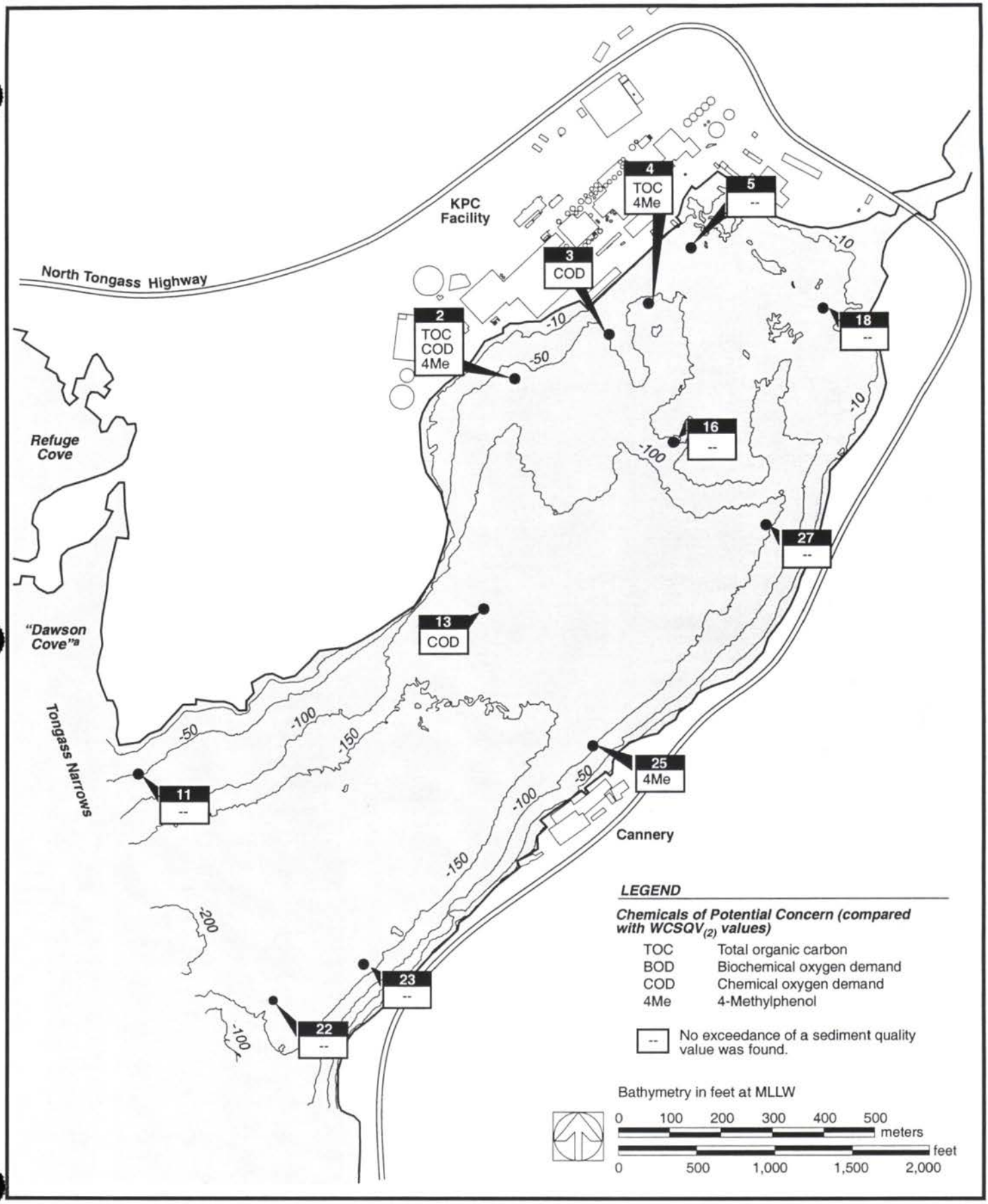


Figure 8-4. Distribution of exceedances of $WCSQV_{(2)}$ values in Ward Cove in 1994-1995.

into two categories based on whether they were considered an AOC station or whether they were not considered an AOC station. The criteria used to designate stations were as follows:

- **AOC Stations:** Stations considered part of the AOC were those that had one or both of the following attributes:

  - The MCUL values were exceeded for both toxicity tests (i.e., the amphipod test based on *Rhepoxynius abronius* and the echinoderm embryo test based on normal survival)

  - The MCUL value for one toxicity test was exceeded and the $WCSQV_{(2)}$ value for one or more CoPCs was exceeded

  Based on those criteria, 23 stations (Stations 1 to 9, 12 to 14, 16, 17, 31, 32, 34, 37, 38, 44, 45, 47 and 48) were designated as being part of an AOC located offshore and downcurrent from the KPC facility.

  Although Station 25 off the fish cannery met the criteria for being part of the AOC, the localized exceedances at this station were not considered to be related to the KPC facility. Station 25 was therefore not designated as part of the AOC that should be evaluated in greater detail.

- **Non-AOC Stations:**

  - No chemical or biological indicator exceeded its MCUL or $WCSQV_{(2)}$ value. Based on this criterion, 10 stations (15, 20 to 24, 26, 28, 40, and 43) were designated as not being part of the AOC.

  - A single exceedance of the MCUL for a toxicity test or CoPC was found, but no other exceedances of sediment quality values for any of the other chemical or biological indicators were found that would corroborate the results of the single MCUL exceedance. Based on this criterion, 10 farfield stations (10, 11, 18, 19, 27, 33, 35, 39, 41, and 42) were designated as not being part of the AOC. Nine of the 10 stations (i.e., all but Station 42) were based on single exceedances for a toxicity test.

Of the nine stations designated as not being part of the AOC based on a single exceedance for a toxicity test, the designations for seven of those stations (i.e., 10, 11, 18, 27, 33, 35, and 41) were based only on the results for the echinoderm embryo test. However, as discussed in Section 7, the endpoint based on normal survival for that test is affected by an error component, because the expected density of embryos in each test chamber is based on an estimated value. By contrast, the normality endpoint for that test is based on known numbers of surviving organisms and is therefore not affected by the same kind of error component as the endpoint based on normal survival. The normality endpoint was therefore used to corroborate the endpoint based on percent normal survival for the seven

stations identified above. Based on the statistical analyses presented in Tables 7-15 and 7-16, and summarized in Tables 7-9 and 7-10, the values of percent normality at the seven stations in 1996 and/or 1997 were as follows:

Station 10	97 percent
Station 11	95 and 96 percent
Station 18	94 and 97 percent
Station 27	95 and 95 percent
Station 33	95 percent
Station 35	97 percent
Station 41	97 percent

All of these values are very high (i.e., ≥94 percent), and none of them differed significantly ($P \leq 0.05$) from the mean values of 97 and 98 percent found in Moser Bay in 1996 and 1997, respectively. In addition, all of these values are greater than the minimum allowable value of 90 percent for acceptable negative controls (Fox and Littleton 1994).

Based on the results of the normality endpoint for the echinoderm embryo test described above, the significant results for the endpoint based on normal survival were not corroborated for any of the seven stations at which that endpoint was the only environmental indicator that identified a potential problem. This lack of corroboration supports the decision not to include those seven stations as part of the AOC.

In delineating the AOC, contour lines were drawn based on interpolations for the chemical or biological indicator that provided the most environmentally conservative estimate of the boundary of each AOC between adjacent stations. For chemicals, the contour represents the estimated position of the sediment quality value for the most conservative of the six CoPCs. For toxicity tests, the contour represents the estimated position of the screening value for the more conservative of the amphipod or echinoderm embryo tests.

Based on the criteria described above, one spatially contiguous AOC of approximately 87 acres was identified for more detailed evaluation (Figure 8-5). The AOC was defined by 23 stations. It was located offshore from the KPC facility and extended to the southwest along the northern shoreline of Ward Cove. Although sediment quality values were not available for dioxins and furans, the AOC included all stations with TCDD TECs greater than 0.22 μg/kg organic carbon.

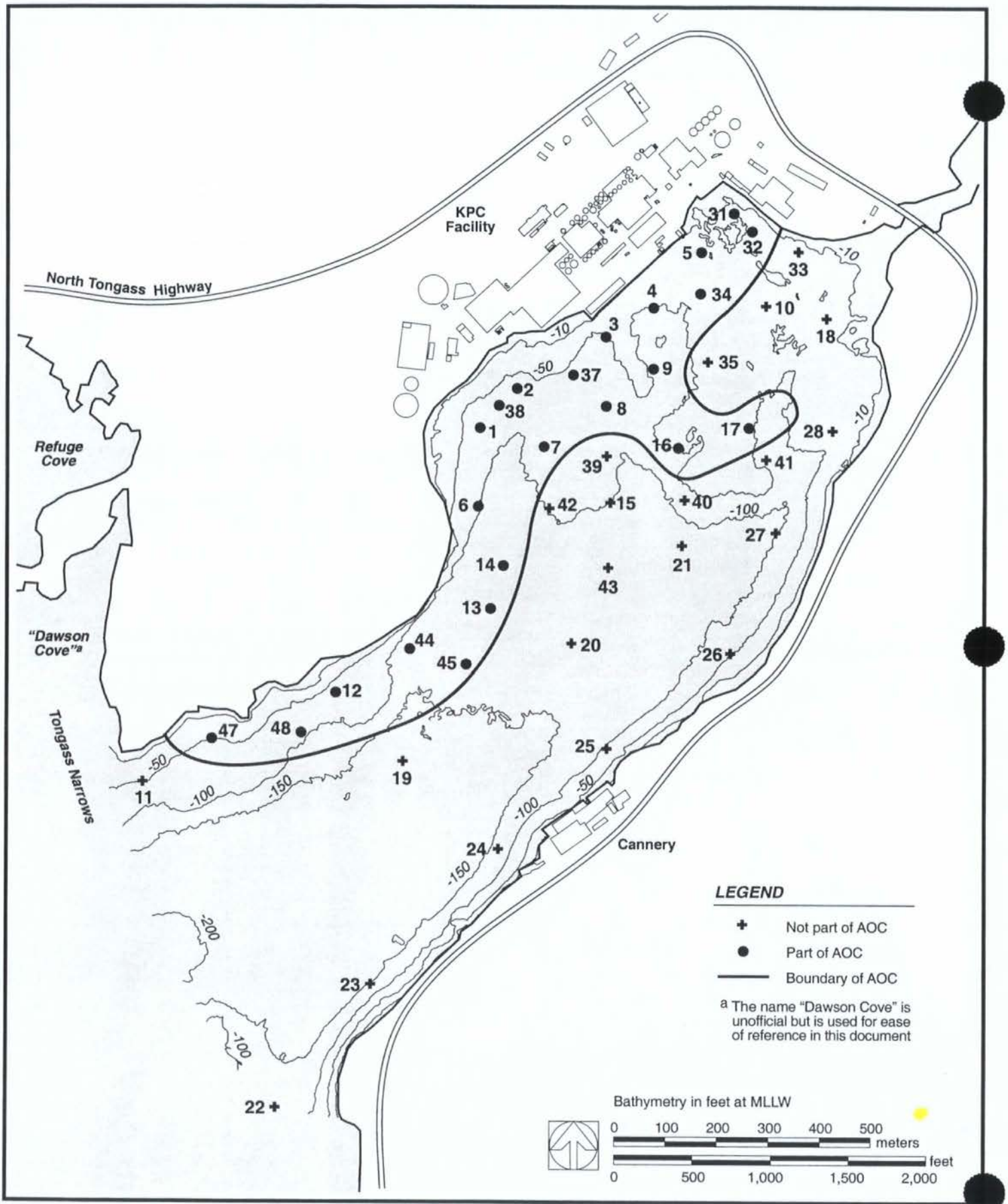
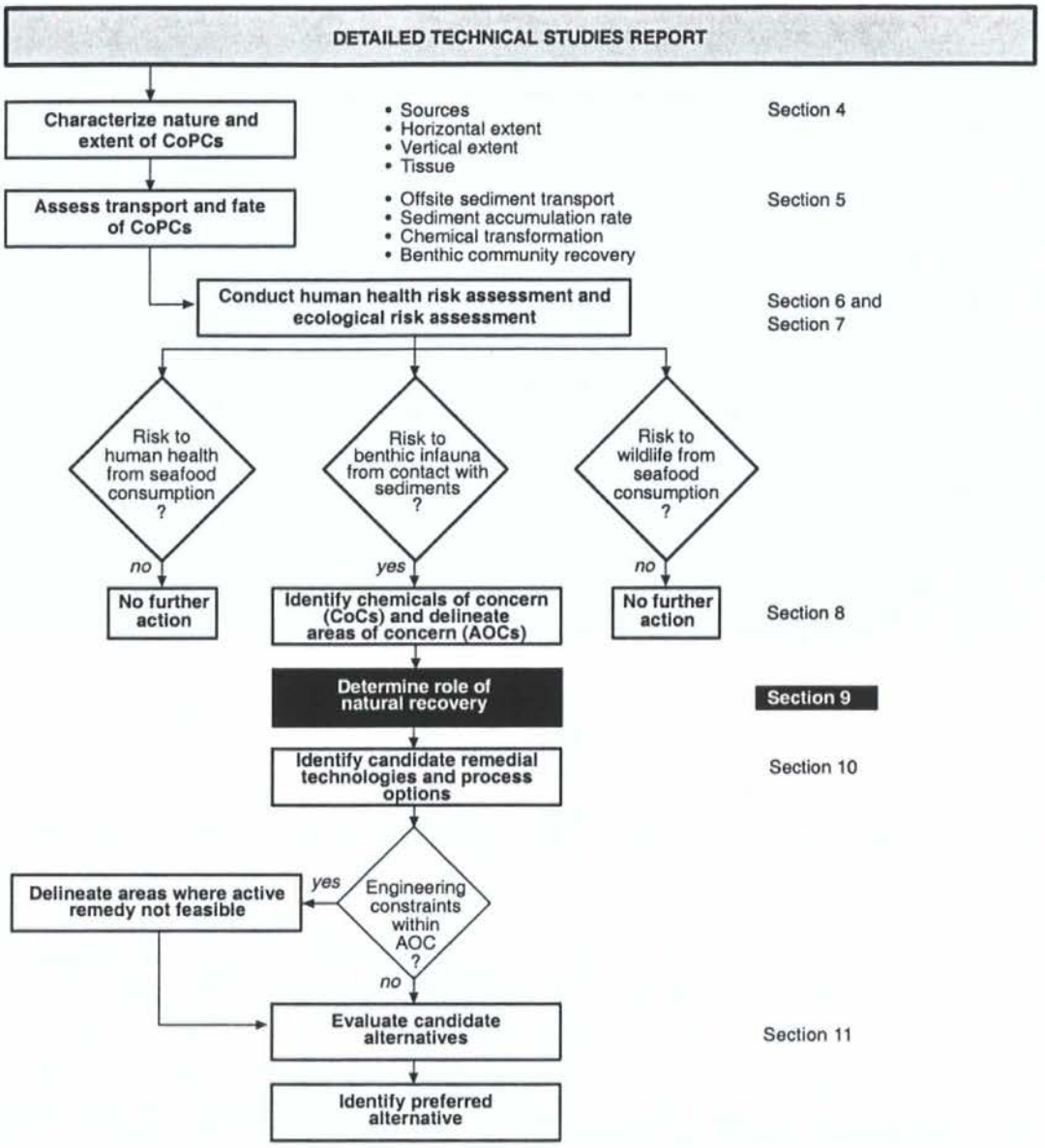


Figure 8-5. Delineation of area of concern for further evaluation.

9. NATURAL RECOVERY



Natural recovery is an integral part of EPA's contaminated sediment management strategy (U.S. EPA 1998a). As stated in U.S. EPA (1998a): "In certain circumstances, the best strategy may be to implement pollution prevention measures as well as point and non-point source controls to allow natural attenuation. Natural attenuation may include natural processes that can reduce or degrade the concentrations of contaminants in the environment including biodegradation, dispersion, dilution, sorption, volatilization, and chemical or biological stabilization, transformation or destruction of contaminants, and the deposition of clean sediments to diminish risks associated with the site." In this con-

text, natural recovery refers only to the biologically active zone of sediment (0–10 cm), not to the entire sediment column.

The factors to consider when determining whether natural recovery is appropriate for a site include the following (U.S. EPA 1998a):

- The specific chemicals present and their associated risk (Sections 4, 6, and 7)
- The designated uses impaired during recovery
- The size of the affected area (Section 8)
- The feasibility and costs of remediation (Section 11)
- Site hydrodynamics, including the extent of downstream transport (discussed in this section)
- The time required for natural recovery (discussed in this section).

Ward Cove is an ideal site for considering natural recovery for all or part of the AOC for several reasons. The source of pulp mill effluent was eliminated with shutdown of the mill in 1997. The CoCs in sediments are natural products of organic matter degradation and are not persistent as are chemicals such as metals and hydrophobic organic compounds. These CoCs are mobile (e.g., sulfide, ammonia) or biodegradable (e.g., 4-methylphenol). The results of the specialized toxicity tests further support the potential for natural recovery, because they indicate that sulfide appears to be the major cause of sediment toxicity in sediment samples from most areas of the Cove. In addition, sediment chemicals are within acceptable limits for human health and wildlife and of limited toxicity to the benthos. Furthermore, existing sediment and hydrodynamic modeling indicate that offsite sediment transport is not a concern.

Some of the physical, chemical, and biological processes occurring in Ward Cove may have effects on natural recovery that are difficult to quantify. These processes and their effects are discussed qualitatively in Section 9.1. Future chemical conditions in the sediment can be predicted using an appropriate quantitative model that incorporates sediment accumulation, chemical transformation, and processes such as diffusion and advection. Section 9.2 describes the two-tier development of models for this purpose and their application to Ward Cove. Techniques for quantitative predictions of recovery rates of a biological community are not available, but a qualitative assessment can be made based on comparison to other high-TOC environments; Section 9.3 discusses the prospects for biological recovery in Ward Cove. Conclusions are summarized in Section 9.4.

9.1 PROCESSES AFFECTING NATURAL RECOVERY IN WARD COVE

Numerous physical, chemical, and biological processes affect the rate at which sediment conditions, and the associated biological communities, will recover from enrichment with

excess organic material. Several of these processes have been discussed, and quantified, in Section 5. Other processes may be known or suspected to occur in Ward Cove, but their effects on natural recovery are difficult to quantify. Some of these potentially important processes are:

- Differential sediment deposition resulting from steep underwater slopes along the north shore of Ward Cove
- Differential sediment deposition resulting from variations in source material throughout Ward Cove
- Changing rates and depths of sediment mixing and irrigation resulting from sediment recolonization by benthic organisms.

The potential effects of these processes are discussed qualitatively in the following three subsections. Because they are not easily quantified, these processes have not been incorporated into the natural recovery models described in Section 9.2; however, their effects should be considered when interpreting the modeling results and the analysis of case studies of natural recovery (Section 9.3).

9.1.1 Underwater Slopes

The north shore of Ward Cove west of the KPC mill has slopes greater than 25 percent for a distance of 45–90 m away from the shore (Section 10.2.2.1; Figure 3-1). This condition may limit the accumulation of fine-grained, organic-rich sediment in this area. Surface sediment was difficult to collect by grab sampling in this area, indicating that surface sediment is sparse or absent on the steep slopes. Several of the samples originally targeted for the area of steep slopes were moved offshore to the foot of the slope. Because effluent solids may not have settled on this slope, sediment in this region of Ward Cove may have limited adverse effects attributable to organic enrichment. Lateral (down-slope) transport of naturally occurring particles may also result in more rapid sediment accumulation (and natural recovery) at the foot of this slope.

9.1.2 Sediment Deposition

Sediment accumulation rates may be greater near the head of Ward Cove than near the mouth for four reasons:

- Supply of coarse-grained material from Ward Creek
- The inward movement of the deep layer of bottom water
- The slower current velocities near the head of Ward Cove
- Flocculation and settling of dissolved minerals in Ward Creek as they enter salt water.

The present distribution of organic-rich sediment may be attributable in part to these processes (i.e., spatial variation in water velocities).

The location at which the sediment accumulation rate was measured (Station 49) is near the mouth of Ward Cove. The measured sediment accumulation rate at this location is therefore expected to underestimate the rate near the head of the Cove. The need for occasional navigation dredging near the Ketchikan sawmill barge ramp confirms that relatively rapid sediment accumulation takes place at the head of the Cove. Most of the material dredged from near the barge ramp is sand, indicating that Ward Creek is the major source. The rate of natural recovery by clean sediment deposition is therefore likely to be highest near the head of Ward Cove, in some of the areas most affected by organic enrichment.

9.1.3 Recolonization and Sediment Mixing

Some types of benthic infauna—specifically, some polychaetes and amphipods—can isolate themselves from problem sediments by constructing tubes in the sediment and irrigating those tubes with oxygenated overlying water. As well as fostering more rapid recolonization of the sediment by microhabitat modification, tube-dwelling infauna may also promote recovery of the bulk sediment in Ward Cove. The concentrations of ammonia, sulfide, and 4-methylphenol in the sediment depend on the rate of transport of dissolved substances between the sediment surface and deep sediment. All of these chemicals are oxidized or degraded in the presence of oxygen, which is introduced to subsurface sediment by infauna. The tubes of pioneer infaunal organisms increase the effective sediment surface area and decrease the diffusion distance, thereby increasing the loss rate of CoPCs (active pumping of water through the tubes may even increase the effective surface area out of proportion to the change in physical area). Recolonization of the sediment by infauna can therefore improve the sediment conditions needed for further recolonization. The positive feedback inherent in this process is likely to be an important influence on the overall rate of natural recovery.

9.2 CHEMICAL RECOVERY

Lateral transport, settling, degradation, and diffusion of CoPCs in Ward Cove have been modeled by ENSR to determine the long-term fate of these chemicals. Modeling was conducted in two phases: the first phase was a box model of the entire Cove that was used to develop overall calibration data, identify data gaps or potential discrepancies, and provide a general estimate of recovery potential; the second phase was a more complex 3-dimensional model that incorporated spatial variation of currents, water depths, and chemical concentrations. The 3-dimensional model is capable of conducting long-term simulation of fate and transport of CoPCs in the sediments and the water column by considering such processes as burial, degradation, sorption, sediment/water exchange, and tidal hydrodynamic flushing.

The box model was used to simulate the period from the opening of the mill (1954) to 20 years in the future (2017). Simulation of the period from mill opening to the present was used to calibrate the model (i.e., to determine certain unmeasured rates or conditions that are important contributors to current, and future, conditions). Simulation of the period from the present to 2017 was used to predict future conditions in Ward Cove. The 3-dimensional model was used only to simulate future conditions because the box model calibration results applied to the 3-dimensional model as well. Both models were used to predict future concentrations of TOC, 4-methylphenol, ammonia, and sulfide in the sediment. The sediment quality values used for modeling purposes are as follows:

- **TOC**—0.30 and 0.31 kg/kg
- **4-Methylphenol**—670 μg/kg
- **Ammonia**—88 and 99 mg/kg.

These values are generally lower than the site-specific sediment quality values developed in Section 7 and thus provide a protective indication (i.e., overestimate) of the natural recovery time frame. During review of the agency draft of this report, the development of a sediment quality value for sulfide was determined to be of limited value because sulfide was measured as total sulfide in sediment but the toxicity data inferred that dissolved sulfide was the causative agent.[8] For the purpose of the modeling presented here, a total sulfide value of 4,300 mg/kg was used as a target value to estimate natural recovery rates for sulfide. BOD and COD were not explicitly modeled, because each of these represents a rate rather than a constituent that is subject to mass balance constraints. The TOC concentration in the sediment is assumed to provide an indication of the potential BOD and COD.

The following sections of this report describe the box model and 3-dimensional model setup and results. A more complete discussion of the models is included as Appendix F.

9.2.1 Box Model Approach (Phase 1)

The box model of Ward Cove uses TOXI5, which is the toxics modeling component of the EPA model WASP5 (Ambrose et al. 1993). The primary focus of this analysis is simulation of sediment processes leading to reduction in concentrations of CoPCs. The box model supports efficient development of model parameters such as loading and reaction rates, and formulation of transport and fate mechanisms for the CoPCs. It also provides an estimate of the overall natural recovery period for each CoPC.

[8] There is no simple empirical or theoretical relationship between total sulfide in sediment and dissolved sulfide in pore water. It is likely that dissolved sulfide would be a small fraction of total sulfide.

9.2.1.1 Input Data

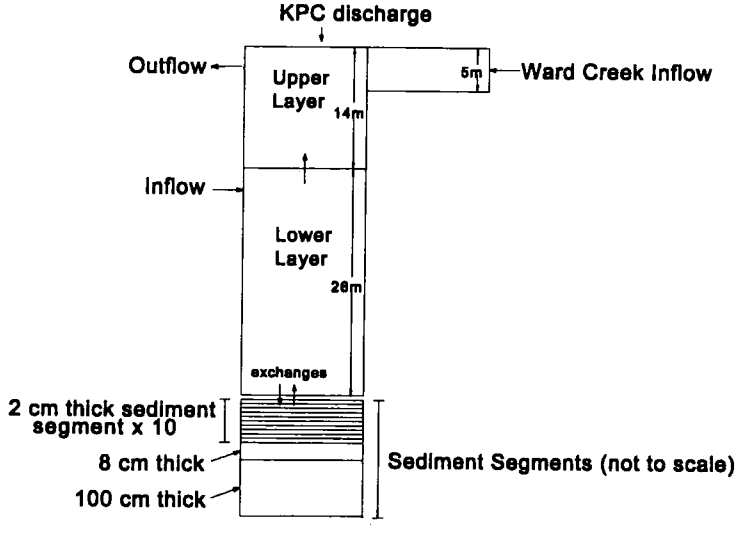
Historical data for Ward Cove are primarily conventional parameters such as TSS and BOD. More extensive data have been collected in 1996 and 1997 describing spatial distributions of CoPCs in the sediment of Ward Cove. Analysis of mill effluent was not initiated until 1989. The buildup of sediment concentrations occurred over a 43-year period, starting with the beginning of KPC operations in 1954.

Data for model input have been derived from several sources, including AWPCB (1957), FWQA (1970), Corps (1971), Higgins and Amoth (1995), Jones & Stokes and Kinnetic (1989), ENSR (1996b), Thibodeaux (1996), PTI (1997g), and NIH (1997). Data from summer 1997 sampling, reported in this document, were also used.

9.2.1.2 Model Setup

In plan view, the box model encompasses all of Ward Cove. The water column is modeled as two vertically distinct segments to represent the hydrodynamic layering typically found in fjords. Multiple sediment segments were used to improve the resolution of predicted chemical distributions.

Recent bathymetric data (summer 1997) were used to compute segment volumes. The projected area is approximately 998,800 m^2. The volume of the upper layer is 23,697,000 m^3, and the volume of the lower layer is 6,387,000 m^3. These numbers are based on upper and lower layer thicknesses of 14 and 26 m, respectively; water column layer thicknesses were based on current meter depths because the current meter data were used to compute flows for the two water layers. Sediment is represented as 12 layers, with 10 layers 2 cm thick underlain by an 8-cm thick layer and a 1-m thick layer. The following schematic illustrates the box model structure.



Flows for the box model were determined by analyzing 1997 current meter data. A flow plane was imposed across the width of Ward Cove, and velocities perpendicular (normal) to this plane were computed from the observed direction and magnitude data for Stations C and D. The velocity data at these stations were collected at a depth below the tidal range, so data for the surface layer were not available. An average normal velocity for the lower layer was computed for the observation period (33 days), which was multiplied by the layer's cross-section in the vertical plane to give inflow and outflow. This velocity was determined to be an inflow of 1.0 m^3/s for the lower hydrodynamic layer, which also moves vertically into the upper layer. For continuity, the upper layer outflow is the sum of Ward Creek, KPC effluent, and lower layer inflows. These flows provide a typical fjord circulation and is illustrated in the schematic above.

Chemical data for sediment CoPCs collected in 1997 were averaged because the box model encompasses the entire surface area of Ward Cove. Area-weighted averages were computed, because the sampling locations are not uniformly distributed. These values are listed in Table 9-1.

Effluent solids settling velocities have been previously measured (ENSR 1996b). A mass (volume)-weighted average settling rate of 0.0074 cm/s (6.4 m/day) was computed based on these data. It is assumed that this rate applies for the whole simulation period.

Effluent solids specific gravity has been measured at 1.27 (ENSR 1996b). This value is used for computation of dry weight solids density. Using an average sediment total solids content of 19.2 percent, and average TVS content of 40.7 percent (1997 data), and assuming a specific gravity of 2.65 for other solids, an overall particle density of 2,088 kg/m^3 and a bulk density of 1,111 kg/m^3 are obtained. These values give a dry weight density for sediment of 220 kg/m^3; this value is one of the model inputs. The TVS content in the sediment is assumed to have been derived entirely from effluent solids, using their corresponding specific gravity.

9.2.1.3 Additional Data Used for Modeling

A key calibration measurement is the sediment deposition rate. This rate was estimated using cesium and lead deposition data. The net sediment accumulation rate was estimated to be 0.35 cm/year from a site at the mouth of Ward Cove (Section 5.2).

Additional data were obtained from various historical reports and discharge permits; these data are shown in Table 9-2. The TSS data reflect pulp production at the mill and timing of effluent treatment. For example, primary treatment was installed in 1971, resulting in a large drop in TSS. Time series data of effluent constituents such as 4-methylphenol and ammonia concentrations were not available, so constant values were used over the entire simulation period. Some of the effluent concentrations, such as 4-methylphenol, were back calculated through the process of model calibration. This is discussed further below.

TABLE 9-1. AREA-WEIGHTED CONCENTRATIONS OF CoPCs FOR WARD COVE (SUMMER 1997 SAMPLING)

CoPC	Area-Weighted Concentration
4-Methylphenol	2,650 µg/kg
Total organic carbon	0.106 kg/kg
Ammonia	120 mg/kg
Sulfide	3,500 mg/kg

Note: CoPC - chemical of potential concern

TABLE 9-2. ADDITIONAL DATA USED IN THE TOXI5 BOX MODEL FOR WARD COVE

Data Item	Value	Source
Effluent TSS (mg/L)	265 (1955–1971)[a] 40 (1971–1980) 56 (1980–1988) 82 (1988–1996)	Higgins and Amoth (1995)
Effluent flow (mgd)	45 (1955–1971) 39 (1971–1980) 39 (1980–1988) 35 (1988–1996)	AWPCB (1957) FWQA (1970) Jones & Stokes and Kinnetic (1989)
Effluent organic content (percent)	31	ENSR (1996b)
4-Methylphenol effluent concentration (mg/L)	0.114–51[b]	1989 effluent scan
Ammonia effluent concentration (mg N/L)	1–2	Corps (1971)
4-Methylphenol diffusivity (cm^2/s)	1.8×10^{-5}	Thibodeaux (1996)
Ammonia diffusivity (cm^2/s)	0.87×10^{-5}	Thibodeaux (1996)

Note: TSS - total suspended solids

[a] Values in parentheses indicate the years over which the value was applied.

[b] Lower value is for 4-methylphenol while the larger is for total phenols.

9.2.2 Box Model Calibration

Calibration results indicate that sediment conditions are driven by solids deposition from the water column. Variation in solids loading from the KPC discharge (Table 9-2) produces large variations in sediment concentrations of the CoPCs. The box model was therefore first calibrated to match the observed sediment accumulation rate by adjusting the organic carbon decay rate. The box model was then calibrated to match the observed CoPC concentrations by adjusting production and loss rates.

9.2.2.1 Sediment Accumulation

The only station at which a sediment accumulation rate could be calculated was located near the mouth of Ward Cove (Section 5.2) and may not accurately reflect historical accumulation near the KPC discharge. As discussed previously (Section 5.2), this sediment accumulation rate is expected to be representative of much of Ward Cove following cessation of the mill discharge. Therefore, this accumulation rate is used for the box model.

TOXI5 in its original form was inadequate for modeling sediment accumulation in Ward Cove, because the model was not designed for decade-long simulations. Modifications were made to TOXI5 to eliminate the problems. A flow through, constant thickness sediment procedure was implemented, which fixes the upper sediment segment boundary to the surface, and holds the total sediment thickness to a constant value. This approach results in a velocity through the simulated segments proportional to the solids settling rate and water column concentration. Decay of organic solids was also added. Because conservation of mass is required, the resulting flux through the sediment is proportional to the sediment thickness, organic solids concentration, and decay rate. These modifications are discussed in detail in Appendix F.

Many small surface sediment layers were used to provide adequate resolution of the movement of a tracer that is tightly bound to sediment particles. The tracer simulates radioactive cesium generated by atmospheric nuclear tests in the 1950s and early 1960s. Model runs were made with two types of solids, one representing KPC effluent solids and another representing native material. Effluent suspended solids concentrations and flows used in all box model simulations are listed in Table 9-2.

Model simulations began in 1954 and continued through 1997 for calibration (43 years of simulation). A pulse of tracer was applied in 1963 for a 1-year period, and the tracer pulse was followed through the sediment up to 1997. Adjustments to the organic decay rate were made until the peak tracer concentration indicated a net accumulation rate of approximately 0.35 cm/year. An organic decay rate of 0.001/day was obtained using this method.

9.2.2.2 Total Organic Carbon

With the organic solids decay rate defined, the next important constituent to calibrate is TOC, because it affects CoPCs by binding the CoPC (e.g., 4-methylphenol) or serving as a surrogate for a constituent (BOD). The data used for TOC calibration include measured KPC effluent solids with an organic content of 31 percent, estimates of suspended solids loading (from Table 9-2), the calibrated organic decay rate (0.0009/day), and measured solids settling rate (6.4 m/day). The initial condition for the sediment at year 1954 uses native Ward Cove solids only, with their corresponding organic fraction. As effluent solids are added to the water column, they settle out and concentrate in the sediment. Because the two solids have differing organic fractions, the total sediment TOC changes. All TOC is assumed to be in the particulate form. The calibration parameter is the native solids organic fraction, which is altered until the predicted TOC concentration matches that found in 1997 (Table 9-1) after 43 simulation years. The native solids organic fraction was found to be 1 percent. This value is lower than the reference area (Moser Bay) value of 6 percent. This suggests that the effluent TOC source term is too high or the decay rate is too low. These effects are likely to extend the predicted recovery time for TOC. The peak TOC model results never exceed the sediment quality values established by PTI (1997g), because the model averages the entire surface area of Ward Cove and does not consider spatial variations.

9.2.2.3 4-Methylphenol

The sorption of 4-methylphenol to organic particulates is relatively low, with a log K_{ow} of 1.94 (NIH 1997). However, the solids concentration in sediments is very high, which allows substantial levels of 4-methylphenol to accumulate over time. Degradation of lignin compounds can also produce 4-methylphenol *in situ* (Hatcher 1988). The TOX15 model was modified to simulate *in situ* production of 4-methylphenol by first-order degradation of organic material (Appendix F). The *in situ* 4-methylphenol production rate was used as a calibration parameter. Calibration was conducted using a source term of 0.114 mg/L in the KPC effluent, based on the 1989 measurement. Exchange of 4-methylphenol between the sediment and water column is assumed to be governed by its diffusivity (Table 9-2) and an assumed sediment tortuosity of 1.41 (Thibodeaux 1996). Sorption of 4-methylphenol to solids was modeled using a log K_{ow} of 1.94. Aerobic degradation in the water column and anaerobic degradation in the sediment were assumed to occur at rates of 0.390 and 0.026/day, respectively (Howard et al. 1991). Other input values are the same as used and calibrated for TOC. A 4-methylphenol yield coefficient in the sediment (the amount of 4-methylphenol produced during decay of organic solids) of 2.1×10^{-5} g 4-methylphenol/g effluent solid is required for simulated results in 1997 to match the observed concentration in the top 10 cm of sediment (Table 9-1).

9.2.2.4 Ammonia

Ammonia was added to the wastewater during the waste treatment process to help break down wood fibers. The permit data indicate 1 to 2 mg/L may have been discharged by

KPC (Corps 1971). When the model is run with an effluent ammonia concentration of 1 mg/L, the simulated sediment ammonia concentration for 1997 is much lower than observed values. Decay of organic solids is assumed to be the source of some ammonia, because nitrogen would be present in the organic matrix of the solids. The calibration parameter used was the yield of ammonia from organic decay. As in the case of 4-methylphenol, the exchange of ammonia between the sediment and water column is also assumed to be governed by its diffusivity (Table 9-2) and an assumed tortuosity of 1.41 (Thibodeaux 1996). Other input values are the same as used and calibrated for TOC. For the current situation, ammonia is assumed not to sorb to solids but to occur only in the dissolved state. An ammonia production rate in the sediment of 0.0065 g NH_3-N/g solid is required for simulated results in 1997 to match the observed ammonia concentration in the top 10 cm of sediment (Table 9-1).

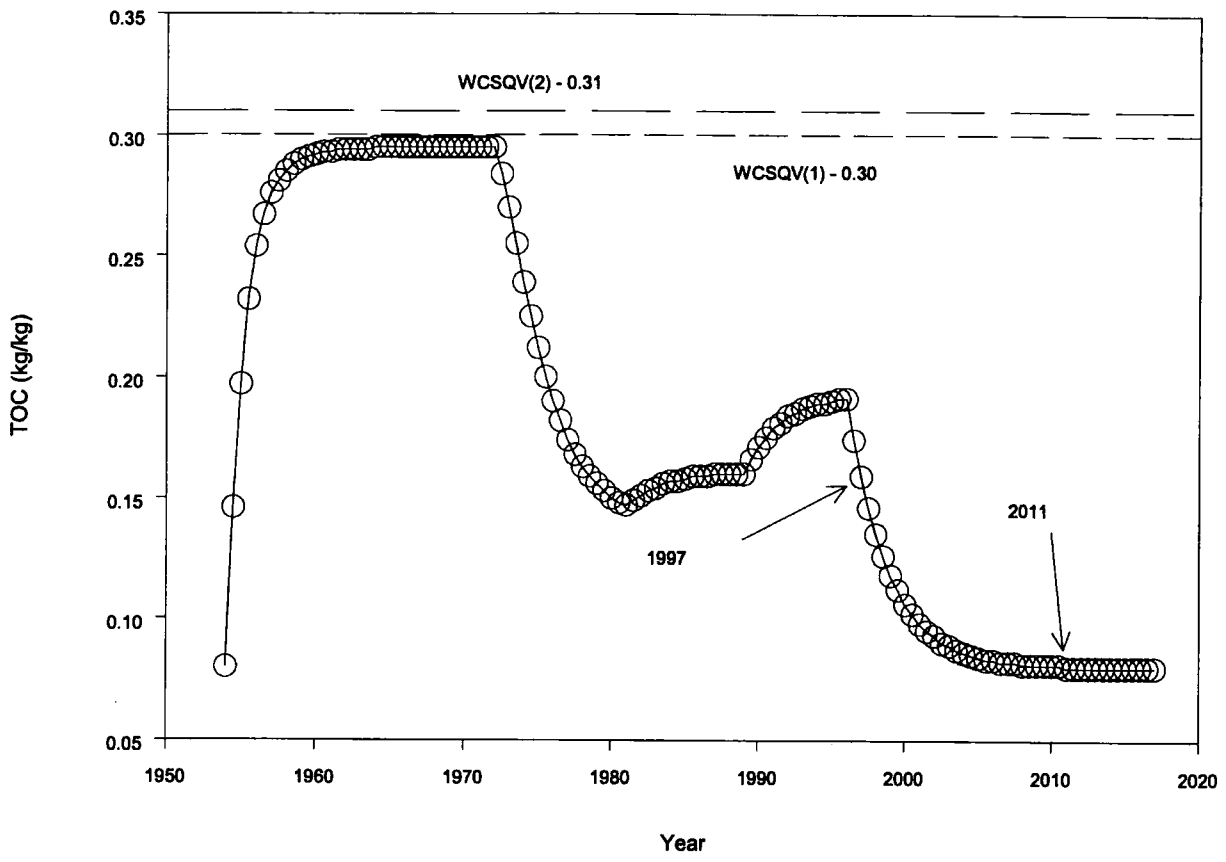
9.2.2.5 Sulfide

Sulfide was assumed to be produced principally by *in situ* reduction of sulfate from seawater during the anaerobic decay of sediment organic matter (Westrich and Berner 1984). Although sulfide has been detected in KPC effluent, this effluent and Ward Creek are considered to be negligible sources of sulfide to the sediment, because sulfide from these sources will be oxidized to sulfate in the surface water of Ward Cove. Sulfide production in the sediment was modeled as dependent on both the rate of diffusion of sulfate into the sediment and the rate of decay of organic matter (Appendix F). The exchange of sulfate and sulfide between seawater and sediment are both assumed to be governed by a diffusivity of 1.07×10^{-5} cm^2/s (the value for sulfate) and an assumed sediment tortuosity of 1.41 (Thibodeaux 1996). The diffusivity for sulfate was used for sulfide also to accommodate limitations of the TOXI5 model; sulfide diffusivity is actually somewhat higher, and use of the lower value will lead to a slight overprediction of sediment recovery times. The concentration of sulfate in seawater was assumed to be 3,648 mg/L (Snoeyink and Jenkins 1980). Sulfide is assumed to occur only in the dissolved state. Other input values are the same as used and calibrated for TOC. A sulfide production rate in the sediment of 5.4×10^{-5} g S/g effluent solid is required for simulated results in 1997 to match the observed concentration in the top 10 cm of sediment (Table 9-1).

9.2.3 Box Model Natural Recovery Simulation

To determine sediment recovery time, that is, the time for sediment concentrations to decrease to acceptable levels (i.e., below sediment quality values), model runs were made 20 years beyond the time when effluent discharges were terminated from KPC to Ward Cove (1996).

The percent TOC never exceeded the sediment quality values during the 43 years when KPC discharge was present. TOC recovery refers to the time it takes to return to the initial condition of 1 percent (Figure 9-1). This process takes 11 years in the box model runs. Six years is required for recovery of 4-methylphenol (Figure 9-2), whereas ammonia takes only 2 years (Figure 9-3). Although exchange with the water column is an



Note:

TOC concentrations are for the top 10 cm of sediment with two solids used in the simulation. Also indicated are the WCSQVs, sample date (1994), and year concentrations return to the initial condition (0.08 kg/kg), (2011).

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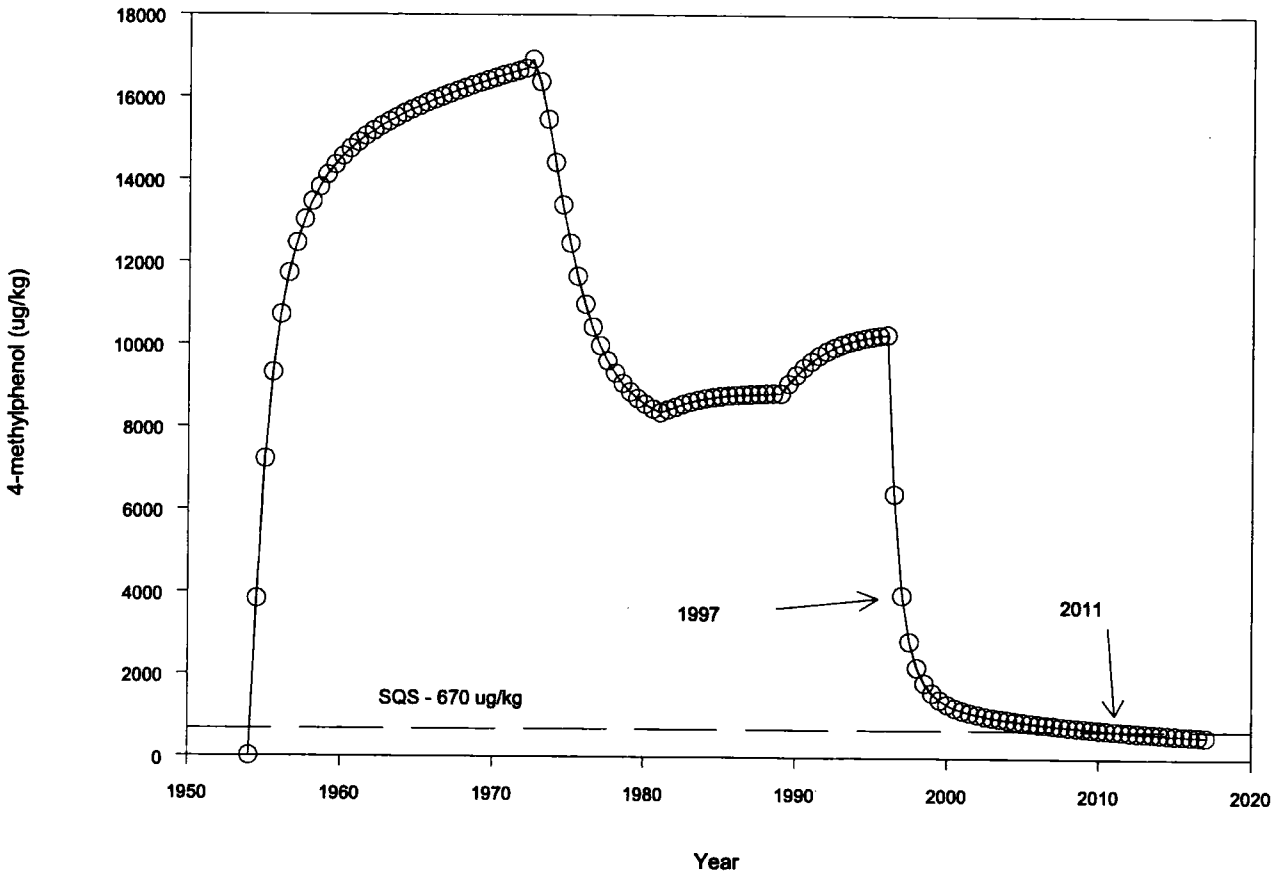
Figure 9-1.

Preliminary TOC modeling results

Exponent Environmental Group

Bellevue, Washington

DRAWN:	sb/km	DATE:	December 10, 1997	PROJECT NO:
FILE NO:	PTI73008	CHECKED:	T. Khangaonkar	5543-007-300



Note:

4-methylphenol concentrations are for top 10 cm of sediment with two solids used in the simulation. Also indicated are the SQS, sample date (1997), and the time when concentrations recover to below the SQS (2011).

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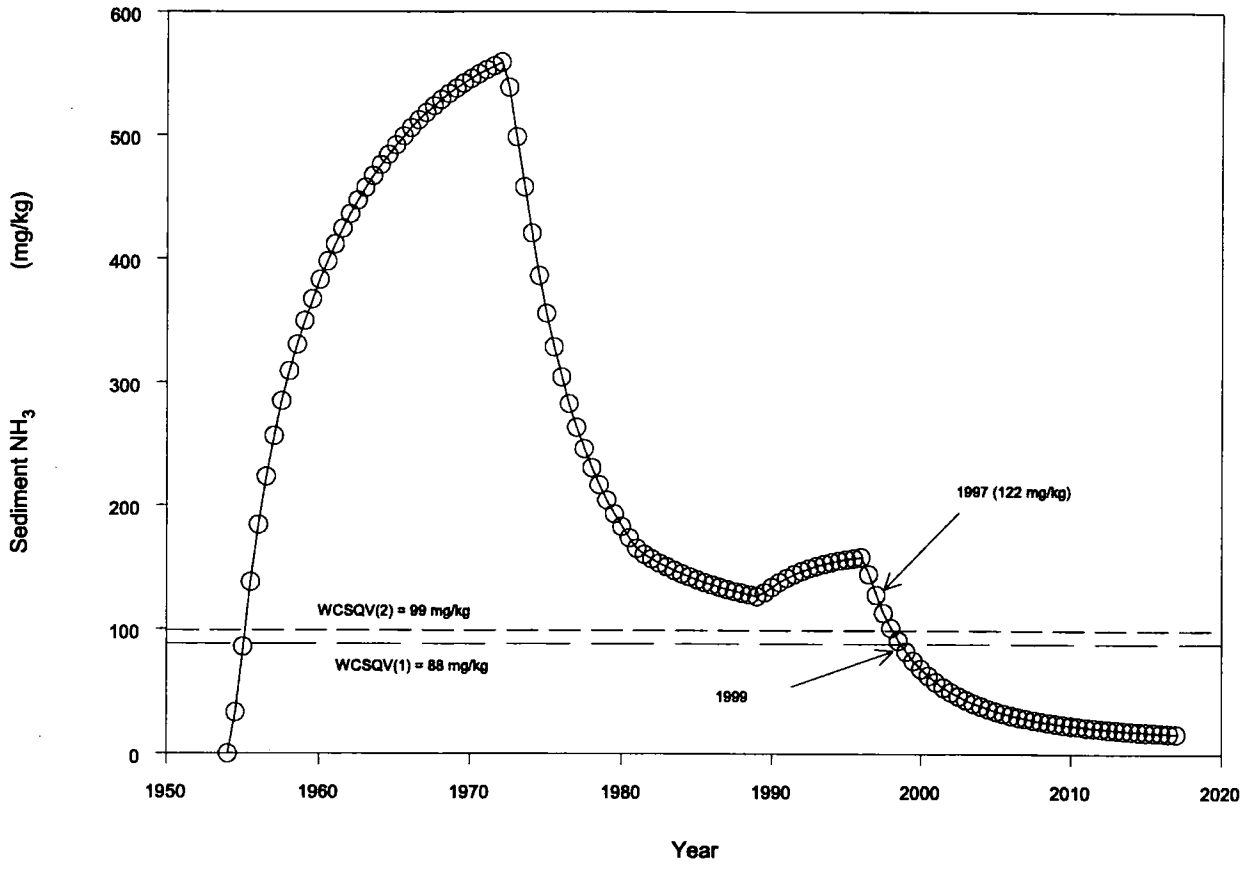
Figure 9-2.

Preliminary 4-methlyphenol modeling results

Exponent Environmental Group

Bellevue, Washington

DRAWN: sb/lon	DATE: December 10, 1997	PROJECT NO:
FILE NO: PTI7300C	CHECKED: T. Khangaonkar	5543-007-300



Note:

NH_3 concentrations are for the top 10 cm of sediment with two solids used in the simulation. Also indicated are the WCSQVs, sample date (1997), and time when concentrations recover to below the WCSQV(1) (1999).

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Figure 9-3.

Preliminary ammonia modeling results

Exponent Environmental Group

Bellevue, Washington

DRAWN:	sb/km	DATE:	December 10, 1997	PROJECT NO:
FILE NO:	PTI7300D	CHECKED:	T. Khangaonkar	5543-007-300

important process for both 4-methylphenol and ammonia, no sorption is assumed for ammonia; hence, it more readily escapes from the sediment.

An important limitation of the box modeling approach is its simplicity, which sacrifices the horizontal spatial resolution of the natural system and does not consider transport processes. This limitation is especially relevant to sediment accumulation data collected at one location and applied over all of Ward Cove. The effluent was discharged at a location over 1 km from the sampling site where sediment accumulation rate was estimated. It is expected that solids settling occurred as the plume traveled this distance, reducing solids flux to the bed at the sampled location. Elsewhere, the solids flux would likely have been greater, especially near the discharge. The transport cannot be quantified in this box model and requires qualification of the box model recovery results. The implication of lower solids flux is an overestimation of the calibrated organic decay rate, which affects the recovery results for all CoPCs. Thus, the recovery period is likely to be longer in localized parts of the study area than is indicated by the current box model.

The model also does not consider organic loading to the sediment from primary production in the water column and does not discriminate between organic material originating in KPC effluent, which is assumed to be biodegradable, and organic material in the form of woody debris. Organic loading from primary production is likely to extend the TOC recovery time beyond that predicted by the model. If TOC from the effluent and from woody debris could be distinguished, the effect could be to increase the calibrated 4-methylphenol and ammonia production rates but decrease the amount of substrate affected; the result is likely to be an increased recovery time for TOC, but recovery times for 4-methylphenol and ammonia might decrease.

9.2.4 3-Dimensional Model Approach (Phase 2)

Although the box model is an efficient tool for the determination of calibration parameters and estimation of overall recovery periods, a fully 3-dimensional model of Ward Cove is need to accurately describe spatial variation in conditions and recovery periods. The model Environmental Fluid Dynamics Code (EFDC; Hamrick 1996) was used to simulate the effects of tidal dynamics, Ward Creek inflow, and KPC discharge on circulation in Ward Cove. Circulation was modeled in each cell of a 3-dimensional grid superimposed on the water column of Ward Cove. EFDC output for each grid cell was linked to TOXI5, which was set up to model sediment and chemical dynamics in the same 3-dimensional grid. Together, these two models predict transport and fate at a higher spatial resolution than the box model.

Ward Cove was segmented using a rectangular grid of 17 segments and 3 water column layers, for a total of 51 water column cells. The thickness of the upper water column layer was set at 12.5 percent of the total thickness, and the thicknesses of the two deeper layers were each set at 43.75 percent of the total thickness. Actual (as opposed to proportional) water column layer thicknesses varied with the tide stage. Twelve sediment layers were used under each of the water column segments, as in the box model; the lowest

sediment layer thickness was increased from 1 m to 10 m to provide a larger reservoir for CoPC accumulation during the 20-year modeling period.

Tidal conditions at the downcurrent boundary of the Cove, which the EFDC model uses to predict current velocities, were established based on tide gauge data collected in 1997. Temperature and salinity of water at the downcurrent boundary were set to 6°C and 29 ppt. The temperatures of the Ward Creek and KPC discharge were set to 6°C and 15°C, respectively, and the salinity of both inflows was set to 0.05 parts per thousand.

Input data for the TOXI5 component of the 3-dimensional model were the same as for the box model, except that different initial sediment conditions were used for each of the 17 grid segments. The sediment organic solids were recalibrated to account for differential transport of effluent throughout Ward Cove. The initial condition for sediment constituents in each segment is based on the results of the box model for 1997, scaled by the actual sediment constituent concentrations measured in 1997. This approach combined the greater spatial resolution of the measured data with the greater vertical resolution of the box model predictions.

9.2.5 EFDC Model Calibration

Currents predicted by the EFDC model were calibrated to the observed 1997 currents (Orders Associates 1997) by varying the water column layer thicknesses. By setting the surface layer to be shallower than the other two layers, observed and predicted currents were made to match, producing a typical fjord circulation consisting of net inflow in the lower layers and net outflow in the surface layer. The subsurface layers near the mouth of Ward Cove show a counterclockwise circulation pattern that occurs during both ebb and flood tides. Model sensitivity tests show that this pattern is likely to be caused by Ward Cove geometry. The existence of higher concentrations of sediment constituents along the north shore of the Cove supports the circulation pattern predicted by the model.

9.2.6 TOXI5 Model Calibration

The overall organic solid settling and decay rates were recalibrated for the 3-dimensional model simulations, taking account of the differential transport of KPC effluent solids throughout Ward Cove. These model parameters were adjusted so that the net sediment accumulation rate in the cell containing Station 49 was similar to the measured value at that station, and so that the average TOC concentration in Ward Cove sediment produced by mill discharge was similar to the TOC concentration observed in 1997. Separate settling velocities were estimated for solids before and after the implementation of effluent treatment in 1971. The calibrated model parameters for organic solids are shown below. Both the organic solids decay rate and the net sediment accumulation rate are lower than used in the box model.

CALIBRATED 3-DIMENSIONAL MODEL PARAMETERS FOR ORGANIC SOLIDS

Parameter	Value
Organic solids decay rate constant	2×10^{-4} day^{-1}
Solids settling velocity	225 m/day (1954–1971) 6.4 m/day (1971–1997)
Net sediment accumulation rate at Station 49	0.26 cm/year

After calibration of organic solids, the model predicted spatial variability in sediment TOC similar to that observed in 1997. Yield coefficients for ammonia, 4-methylphenol, and sulfide were then recalibrated in the same manner as used for the box model, but incorporating the predicted spatial variability of TOC. The calibrated yield coefficients used for the 3-dimensional model are shown below.

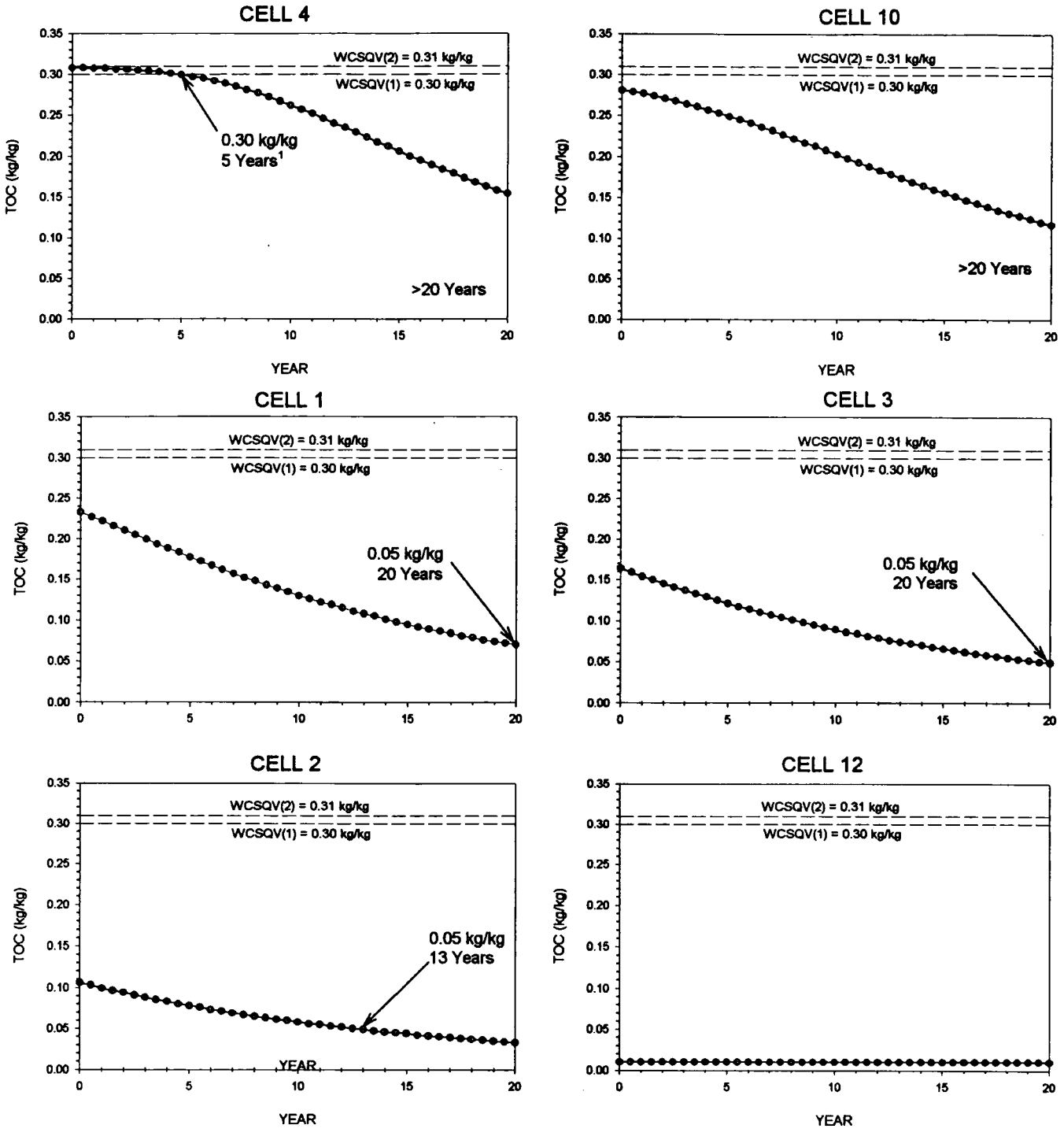
CoPC YIELD COEFFICIENTS USED FOR THE 3-DIMENSIONAL MODEL

CoPC	Yield Coefficient
Ammonia	3.06×10^{-2} g NH_3/g effluent solid
4-Methylphenol	1.09×10^{-4} g 4-methylphenol/g effluent solid
Sulfide	3.03×10^{-4} g S/g effluent solid

9.2.7 3-Dimensional Model Natural Recovery Simulation

The 3-dimensional model was run for a period of 20 years to predict future concentrations of TOC, 4-methylphenol, ammonia, and sulfide in the sediment. Recovery of TOC was considered to occur when the sediment TOC concentration reached the calibrated 1954 value of 1 percent (because most of the Cove is already below the sediment quality value for TOC), and recovery of 4-methylphenol, ammonia, and sulfide was considered to occur when their concentrations reached their respective sediment quality value or target value. Results of the 3-dimensional model are consistent with the box model: the recovery times predicted by the 3-dimensional model for different segments bracket the times predicted by the box model.

The rate of TOC recovery in selected segments is shown in Figure 9-4, and the estimated recovery time for each of the segments is shown in Figure 9-5. The region directly to the west of the mill, which had the highest initial TOC concentration, is anticipated to require more than 20 years to return to the 1954 TOC concentration. Areas in the center of Ward Cove are also expected to require an extended natural recovery period. Several of the model limitations described in Section 9.2.2 may lead to underestimation of TOC recovery times. Perhaps the most important of these limitations is the lack of distinction



Note:
Since TOC levels were less than the Ward Cove Sediment Quality Value (WCSQV) in all but Cell 4, the time to recovery was determined as the time to approach the estimated background TOC content of 0.05 kg/kg.
Results are for the top 10 cm of sediment.
Units are kg TOC/kg dry sediment solids.
[1]5 Year recovery time to WCSQV(1).

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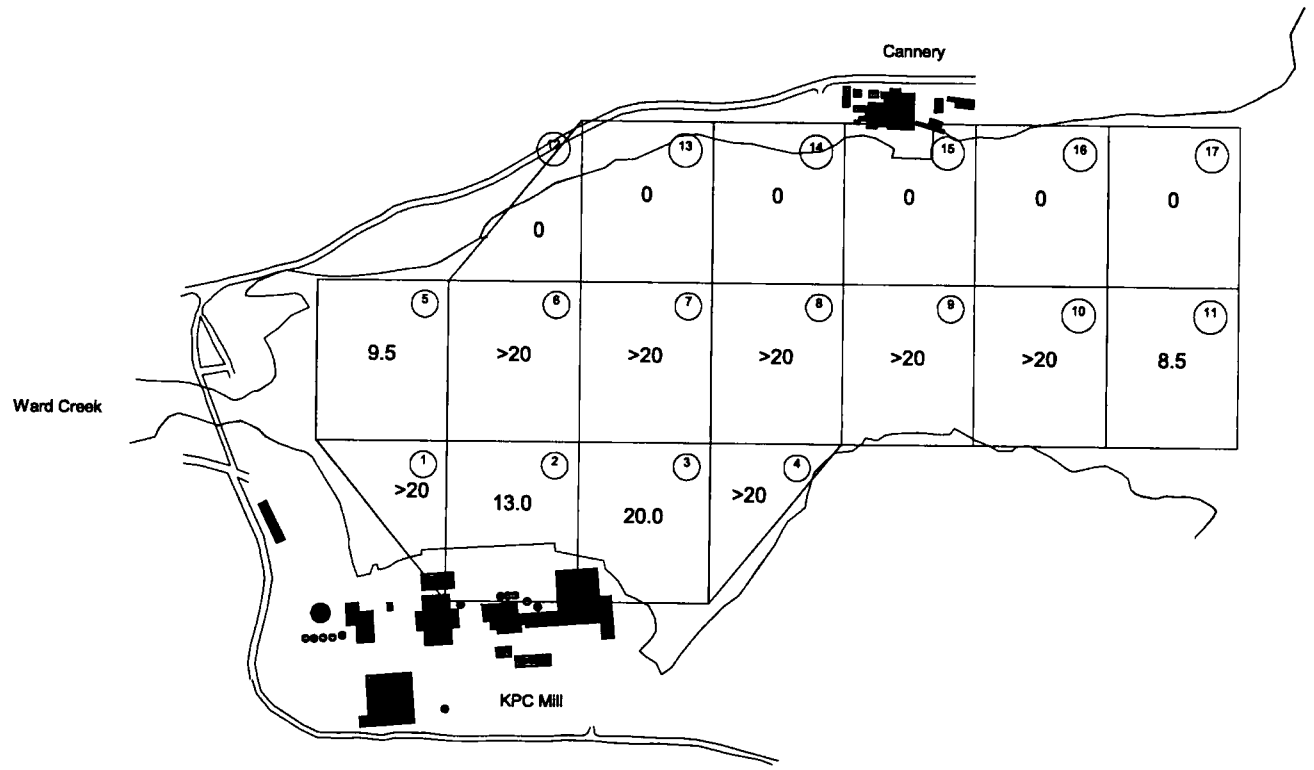
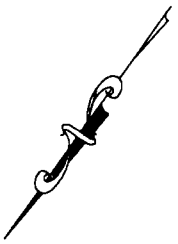
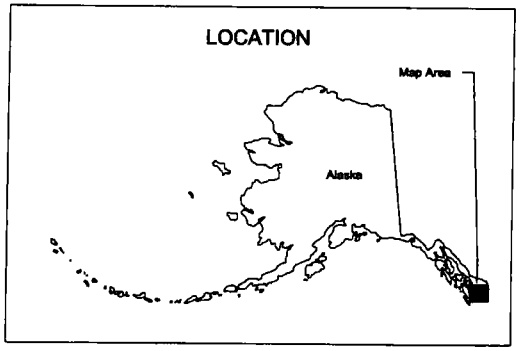
FIGURE 9-4
TOTAL ORGANIC CARBON SEDIMENT RECOVERY MODELING RESULTS
Exponent Environmental Group
Bellevue, Washington

DRAWN: N. Chin	DATE: May 29, 1998	PROJECT NO: 5543-007-700
FILE NO: 5543-007.jnb	CHECKED: C. DeGasperi	







Note: Numbers in upper right corner of segments are segment indices.

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FIGURE 9-5.
SUMMARY OF RECOVERY RESULTS
FOR TOC, WITH RETURN TO A LEVEL
OF 0.05 kg/kg - VALUES IN YEARS

Exponent Environmental Group
Bellevue, Washington

DRAWN:	K. Monger	DATE:	June 1, 1998	PROJECT NO:
FILE NO:	EEG0780U	CHECKED:	S. Breithaupt	5543-007-800

between organic material originating in KPC effluent and in woody debris. The latter category may persist for longer than the model predicts.

The rate of 4-methylphenol recovery in selected segments is shown in Figure 9-6, and the estimated recovery time for each of the segments is shown in Figure 9-7. Recovery is expected to require 20 years or more in the region offshore of the mill and along part of the north shore of the Cove. A high initial concentration of 4-methylphenol and continued *in situ* production from TOC are likely the principal causes of extended recovery times.

The rate of ammonia recovery in selected segments is shown in Figure 9-8, and the estimated recovery time for each of the segments is shown in Figure 9-9. Extended recovery times (20 years or more) are predicted offshore of the mill and along the north shore of Ward Cove.

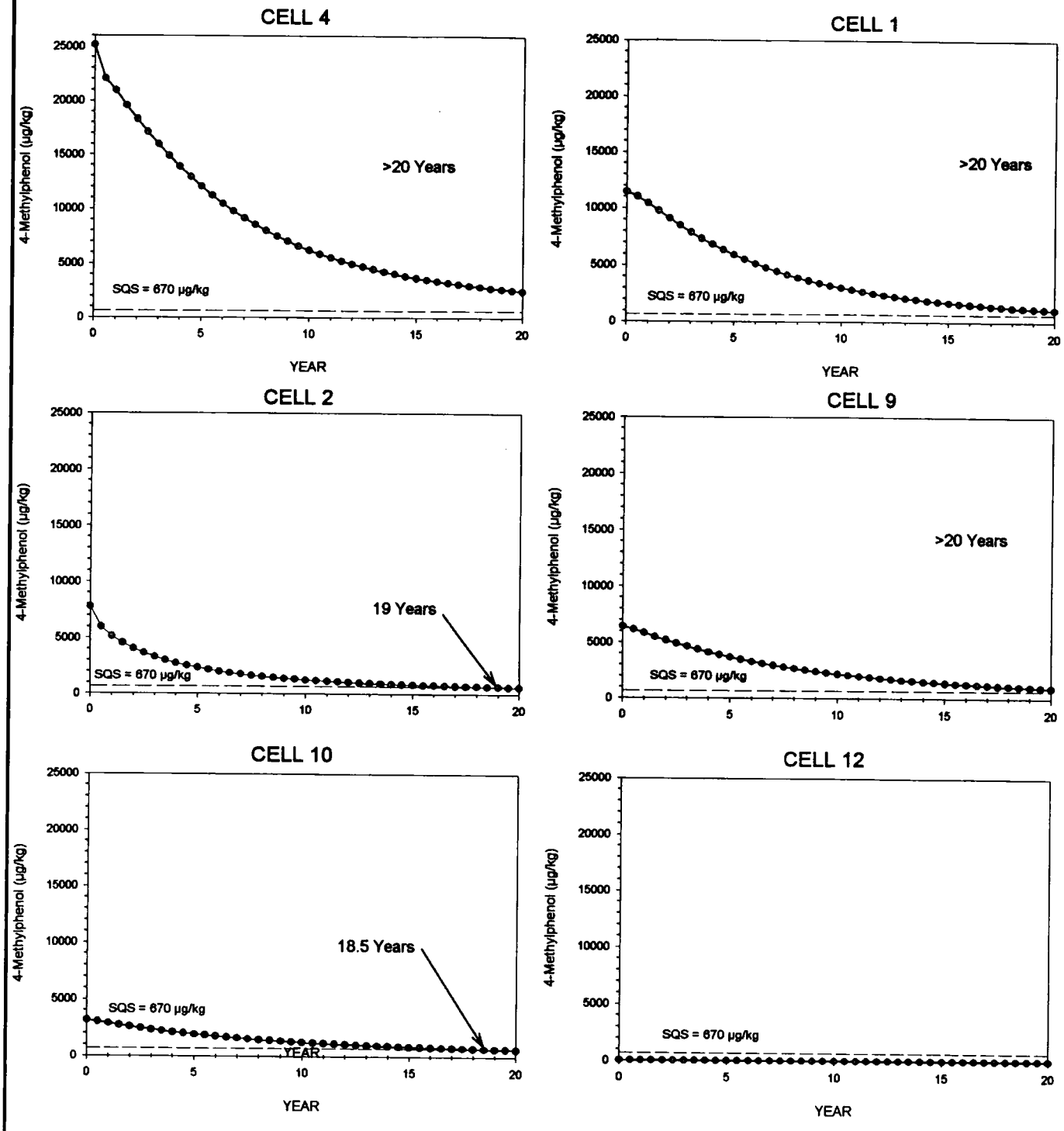
The rate of sulfide recovery in selected segments is shown in Figure 9-10, and the estimated recovery time for surface sediment in each of the segments is shown in Figure 9-11. The longest recovery times are expected in deep water offshore of the mill, where initial conditions were highest.

The 3-dimensional model results are not only consistent with the box model, they are also in accord with sampling data in that they show the longest recovery times along the northern shore, and they are internally consistent in identifying the same area as having the longest recovery time for all constituents. This level of consistency indicates that considerable confidence can be placed in the spatial distribution of estimates of relative recovery times. The spatial resolution of the 3-dimensional model clearly identifies an area to the west of the mill that is expected to require an extended period for natural recovery and provides a basis for selection of appropriate remedial alternatives for different parts of Ward Cove. Actual recovery times may differ from those predicted by the model because of uncertainties regarding the composition of organic matter and organic matter decay rates. Because 4-methylphenol, ammonia, and sulfide are produced as by-products of TOC degradation, these uncertainties in organic carbon conditions and processes also affect the predicted recovery rates for these CoPCs. The effect of uncertainties in the organic carbon degradation rate and other model parameters was evaluated by conducting sensitivity analyses of these model components.

9.2.8 Model Sensitivity Analyses

Sensitivity analyses were conducted to evaluate the effects of variability in four model components:

- Organic solids decay rate
- Thickness of the surface sediment segment (mixing layer)



Note:
Results are for the top 10 cm of sediment.
Units are µg/kg dry sediment solids.
Sediment Quality Standard (SQS)
= 670 µg/kg.

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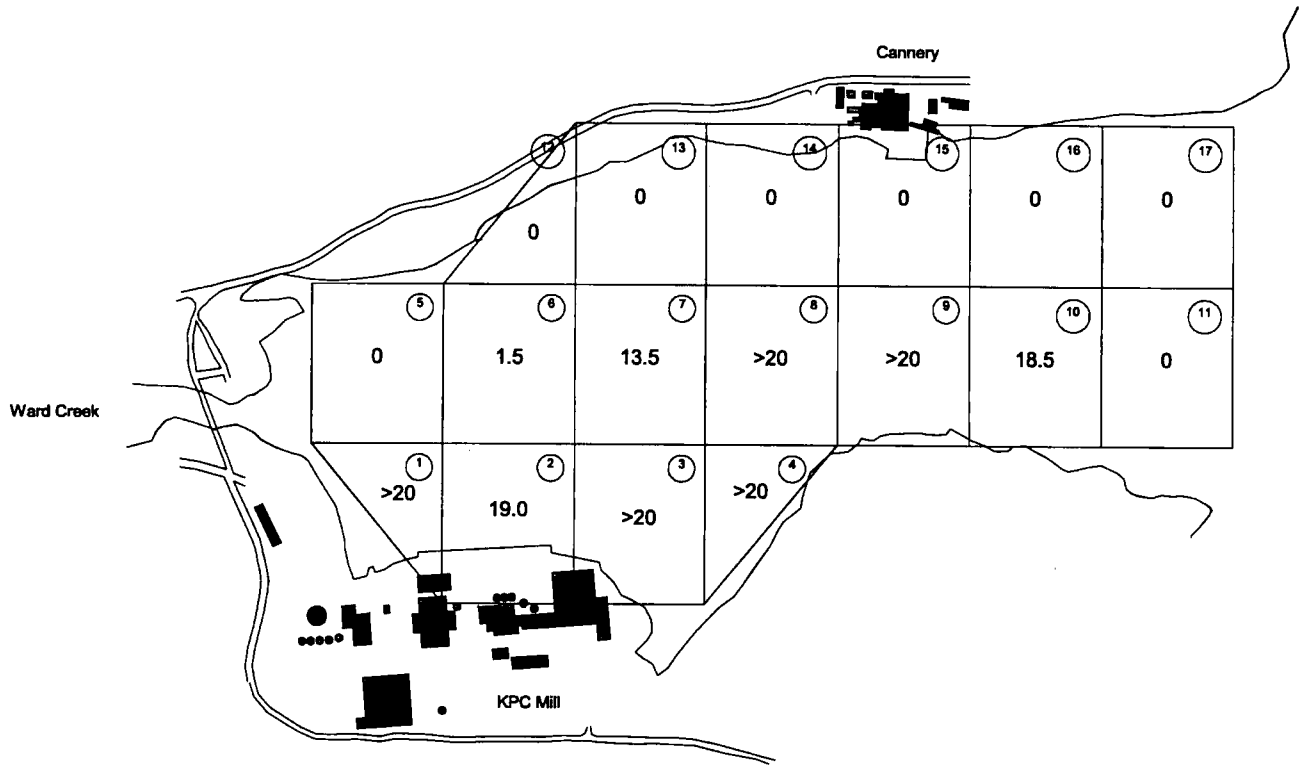
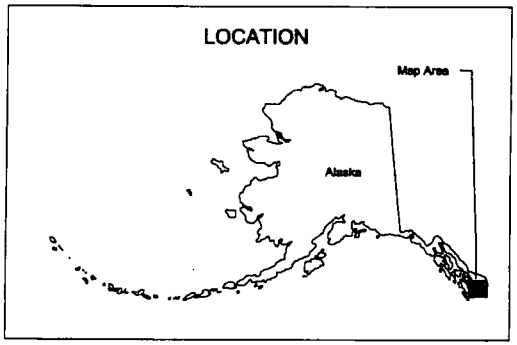
FIGURE 9-6
4-METHYLPHENOL SEDIMENT
RECOVERY MODELING RESULTS
Exponent Environmental Group
Bellevue, Washington

DRAWN: N. Chin	DATE: June 2, 1998	PROJECT NO: 5543-007-700
FILE NO: 5543-007.jnb	CHECKED: C. DeGasperi	





Note: Numbers in upper right corner of segments are segment indices.

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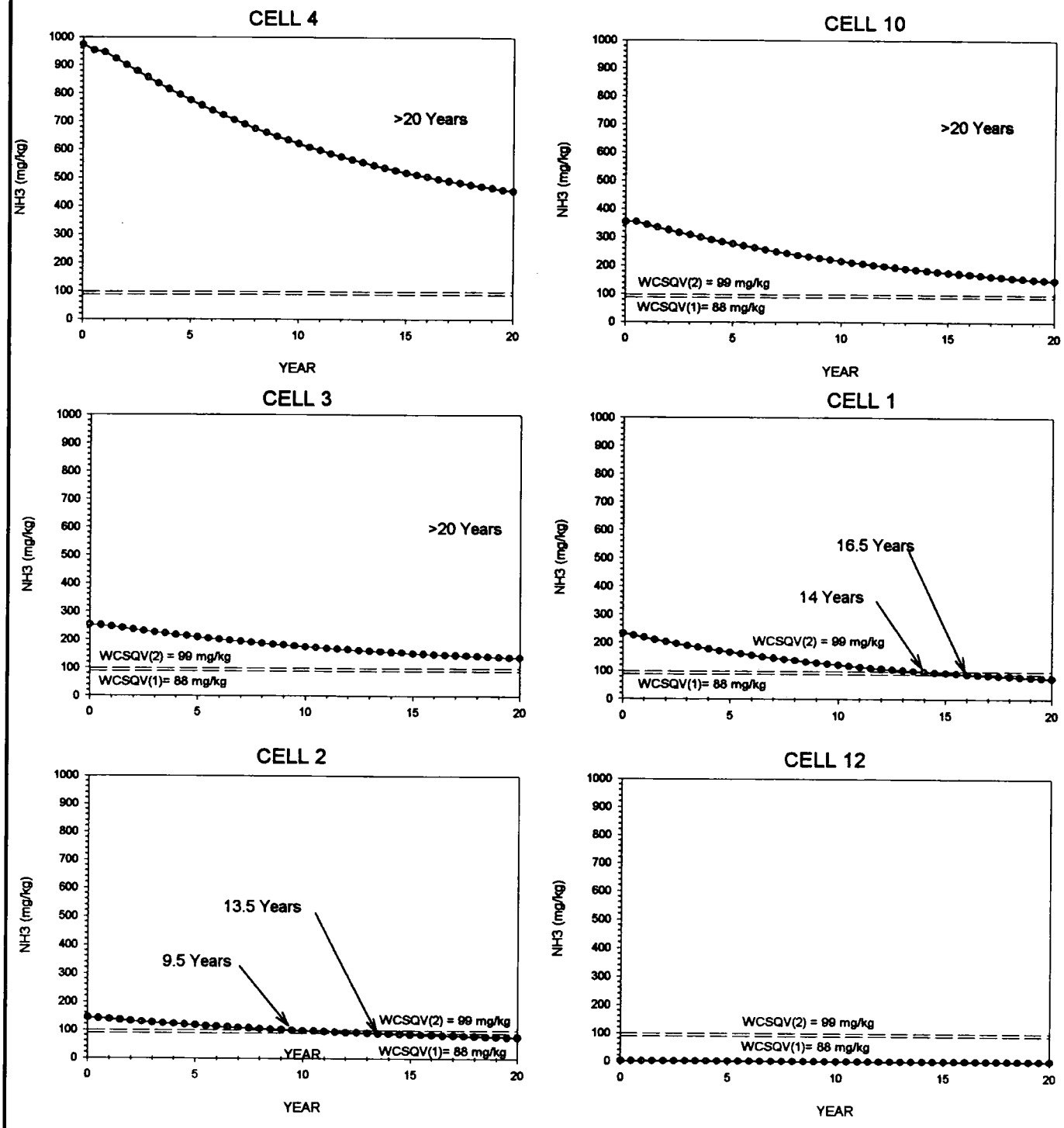
FIGURE 9-7.
SUMMARY OF RECOVERY RESULTS FOR
4-METHYLPHENOL, WITH RETURN TO LEVELS
OF <670 ug/kg - VALUES IN YEARS

Exponent Environmental Group
Bellevue, Washington

DRAWN:	K. Monger	DATE:	June 2, 1998	PROJECT NO:
FILE NO:	EEG0780V	CHECKED:	S. Brethaupt	5543-007-800



Note:
Results are for the top 10 cm of sediment.
Units are in mg/kg dry sediment solids.
Ward Cove Sediment Quality Value (WCSQV) = 88 AND 99 mg/kg.

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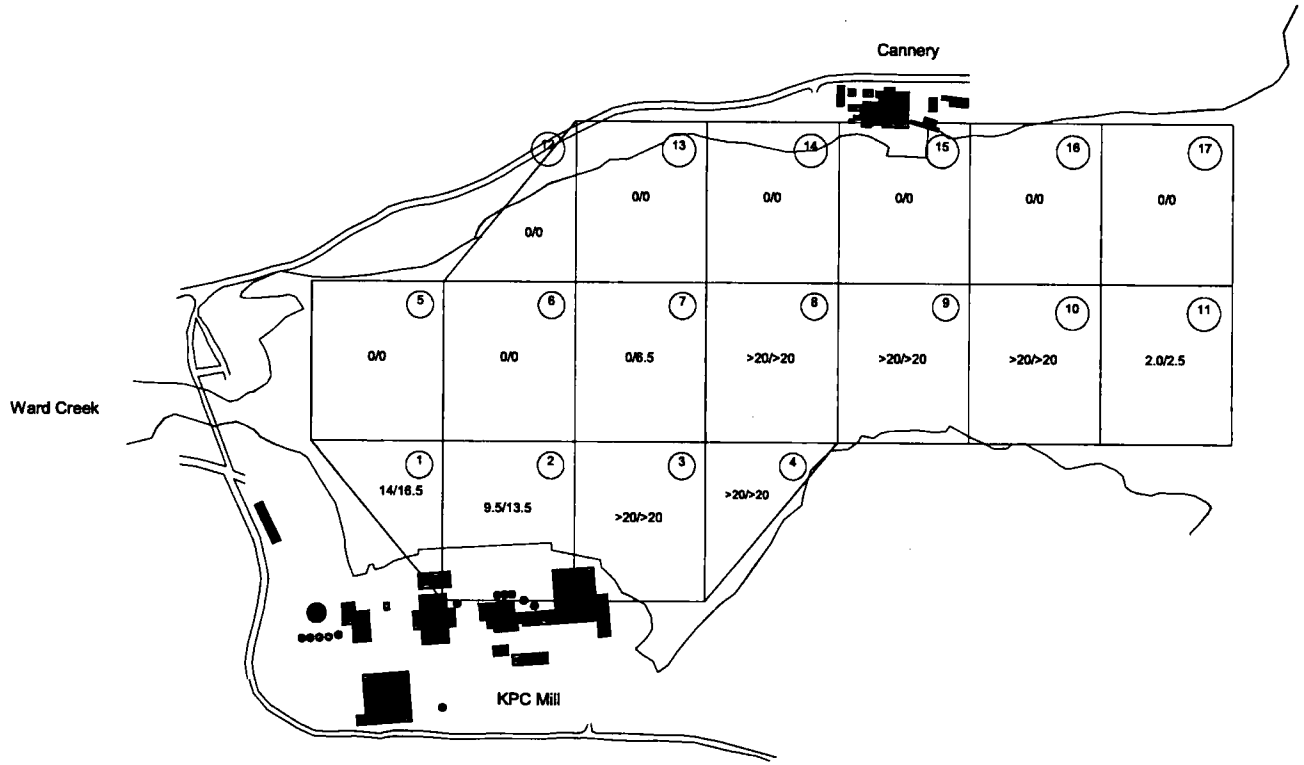
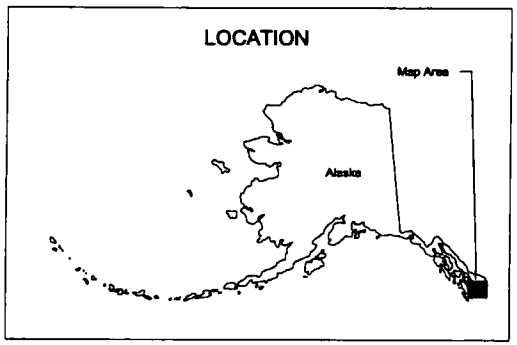
FIGURE 9-8
AMMONIA SEDIMENT RECOVERY MODELING RESULTS
Exponent Environmental Group
Bellevue, Washington

DRAWN: N. Chin	DATE: June 2, 1998	PROJECT NO: 5543-007-700
FILE NO: 5543-007.jnb	CHECKED: C. DeGasperi	





Note: Numbers in upper right corner of segments are segment indices.

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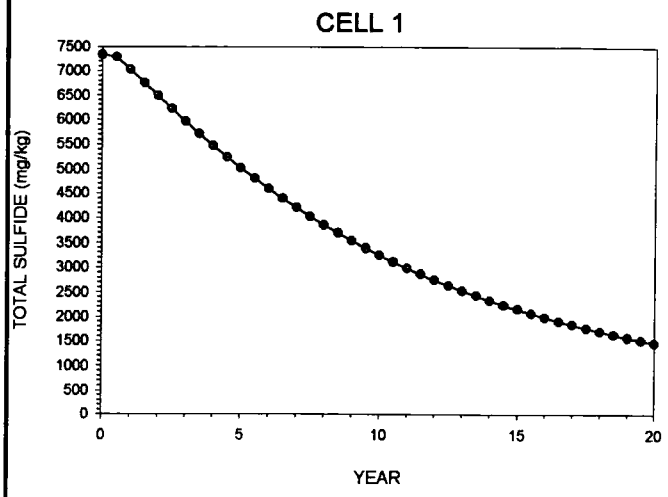
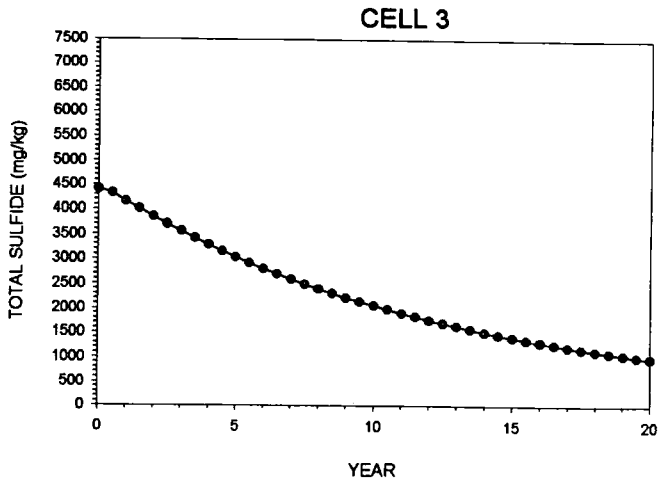
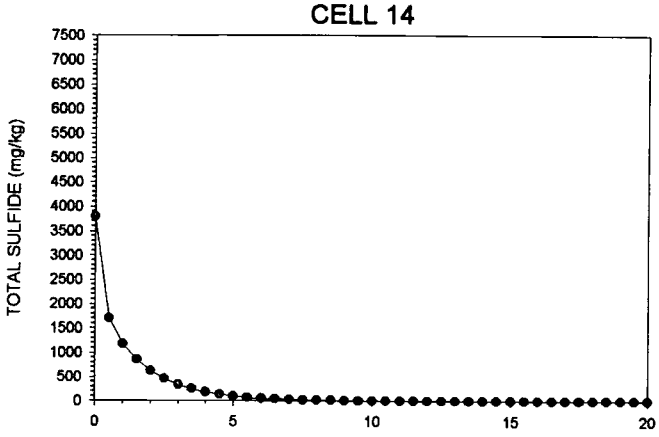
FIGURE 9-9.
SUMMARY OF RECOVERY RESULTS FOR
AMMONIA, WITH RETURN TO LEVELS
<99 AND <88 mg/kg - VALUES IN YEARS

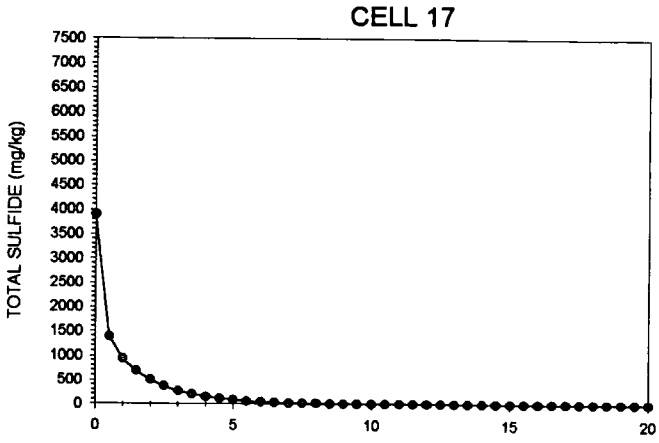
Exponent Environmental Group
Bellevue, Washington

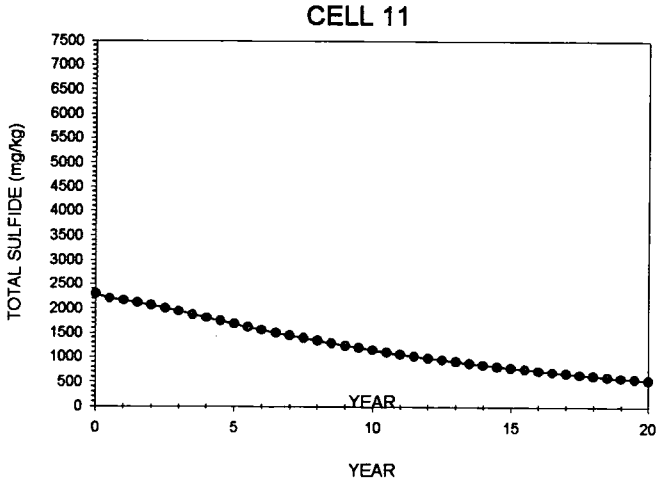
DRAWN:	K. Mongar	DATE:	June 2, 1998	PROJECT NO:
FILE NO:	EEG0780W	CHECKED:	S. Brethaupt	5543-007-800

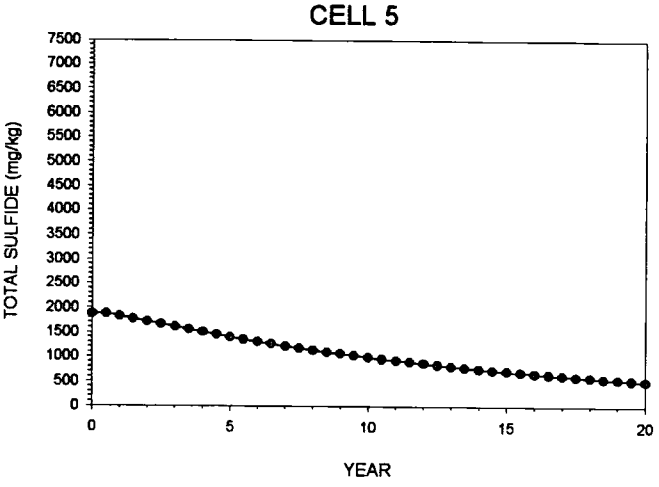












Note:
Results are for the top 10 cm of sediment.
Units are mg/kg dry sediment solids.

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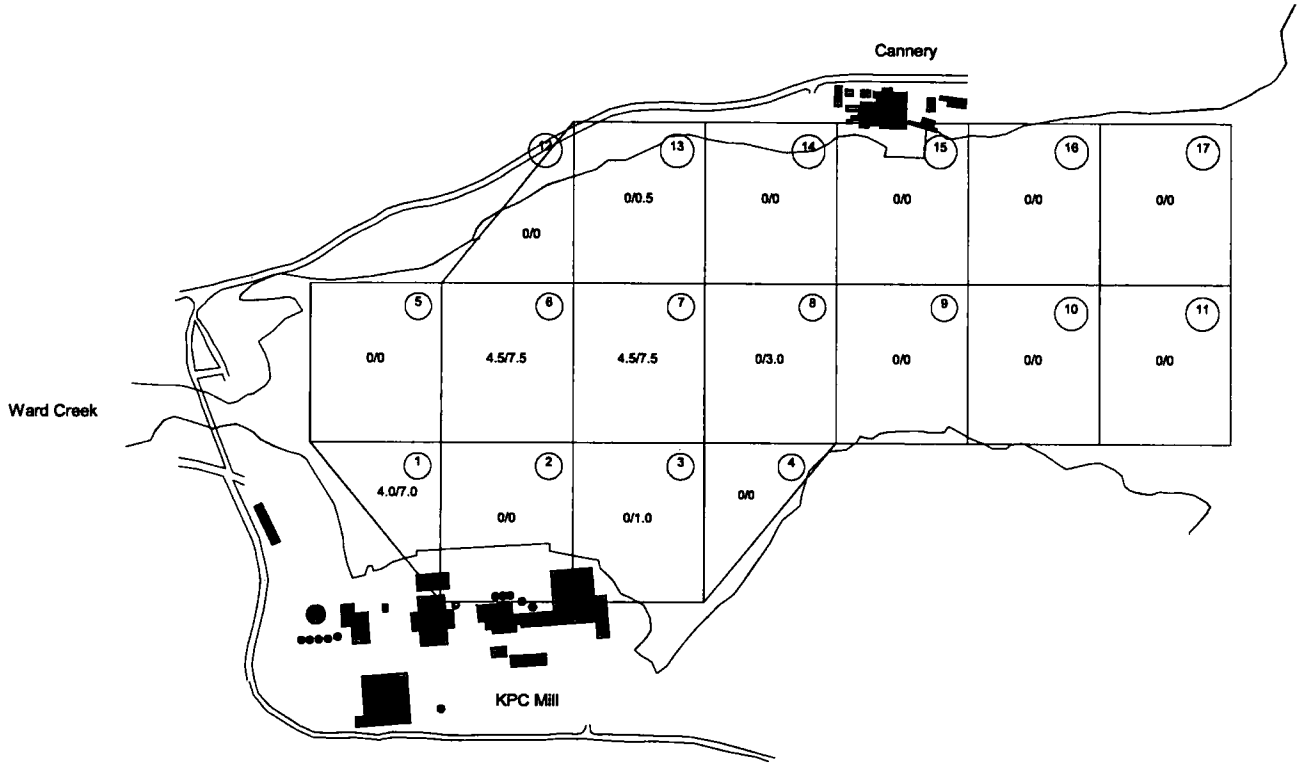
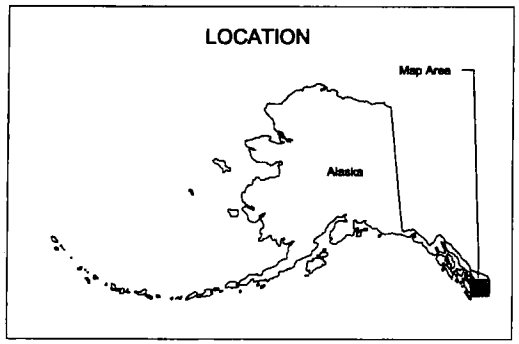
FIGURE 9-10
SULFIDE SEDIMENT
RECOVERY MODELING RESULTS

Exponent Environmental Group
Bellevue, Washington

DRAWN: N. Chin	DATE: June 1, 1998	PROJECT NO: 5543-007-700
FILE NO: 5543-007.jnb	CHECKED: C. DeGasperi	





Note: Numbers in upper right corner of segments are segment indices.

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FIGURE 9-11.
SUMMARY OF RECOVERY RESULTS FOR
SULFIDE, WITH RETURN TO LEVELS <5500
AND <4300 mg/kg- VALUES IN YEARS

Exponent Environmental Group
Bellevue, Washington

DRAWN:	K. Monger	DATE:	June 1, 1998	PROJECT NO:
FILE NO:	EEG0780X	CHECKED:	S. Breithaupt	5543-007-800

- Native solids settling velocity
- 4-Methylphenol decay rate.

The sensitivity analyses were conducted by varying these model components relative to the base case (described previously) and determining the effects on recovery times.

The organic solids decay rate constant was halved and doubled relative to the base rate (2×10^{-4}/day). These changes had an almost directly proportional effect on the recovery time of TOC throughout Ward Cove: the higher degradation rate led to shorter recovery times. For 4-methylphenol, ammonia, and sulfide, the lower solids decay rate constant resulted in a recovery rate that was initially more rapid than the base case (as a consequence of lower production of the CoPC), but a slower recovery rate in the long term (as a consequence of a more extended period of CoPC production). The effects on recovery time therefore varied throughout Ward Cove, depending on the relationship between the initial condition and the sediment quality value. In some locations, the initially rapid recovery rate associated with a lower solids decay rate constant brought the CoPC concentration below the sediment quality value, leading to a shorter recovery period, and in other locations, the extended period of CoPC production associated with a lower decay rate constant extended the recovery time. In model segments along the north shore of Ward Cove, a lower decay rate constant generally led to an increase in the recovery time. The greatest changes in recovery times were roughly proportional to the changes in the decay rate constant.

The thicknesses of the uppermost sediment segments in the model were changed to evaluate the effects of changes that might be associated with different amounts of sediment mixing (e.g., bioturbation). The base model runs used ten 2-cm layers at the sediment surface, and alternative model runs were carried out using two 10-cm layers. The former (base) case represents relatively little surface sediment mixing, as might be expected with a stressed infaunal community, and the latter (alternative) case represents the maximum mixing that might be expected to result from bioturbation. The change in sediment mixing depth had almost no effect on the TOC recovery period, because TOC recovery is controlled by *in situ* biodegradation and not by mixing. The recovery period for sulfide in surface sediment is dependent on the rate of transport of sulfide to a clean boundary (i.e., surface water), whereas the recovery periods for 4-methylphenol and ammonia in surface sediment are dependent on their rates of transport from a reservoir (i.e., deep sediment). Consequently, greater mixing of surface sediment resulted in a shorter recovery period for sulfide but a longer recovery period for 4-methylphenol and ammonia. For 4-methylphenol and ammonia, the total mass of these CoPCs in the sediment decreased faster with the greater mixing depth, although the surface sediment concentrations were higher within the 20-year modeling period.

The effect of the native solids settling velocity on recovery times was evaluated by doubling the velocity used in the base case (20 m/day). Because native solids originate from Ward Creek, the effect of this change was greatest near the mouth of Ward Creek. The effect of the increased settling velocity was more rapid burial of effluent solids within the

10-cm layer of surface sediments. Burial of the effluent solids results in an increase of the diffusion distance for sulfate (which is the source of sulfide), 4-methylphenol, and ammonia. The result of the increased native solids settling velocity on the 10-cm surface sediment layer is therefore a marked decrease in the recovery time for sulfide and a small increase in the recovery times for 4-methylphenol and ammonia. If the native solids settling rate were even higher, so that more of the 10-cm surface layer consisted of native solids, the recovery times for 4-methylphenol and ammonia would be expected to decrease. The recovery period for TOC was essentially unaffected by the change in the native solids settling rate.

The aerobic 4-methylphenol decay rate constant (in the water column) and the anaerobic 4-methylphenol decay rate constant (in the sediment) were both decreased by a factor of 2 to evaluate their effects on the 4-methylphenol recovery period. These changes had a negligible effect on the 4-methylphenol recovery period, indicating that the production and diffusion rates, rather than the degradation rate, control the concentration in surface sediment.

9.2.9 Summary of Natural Recovery Modeling

Modeling of CoPC dynamics—specifically, the effects of production and loss rates on the inventory and concentrations in surface sediments—indicates that extended recovery periods (20 years or more) for TOC, 4-methylphenol, and ammonia are expected immediately adjacent to the mill and along the northern shore of Ward Cove. Recovery times of all CoPCs are dependent on, and fairly sensitive to, the biodegradation rate of organic carbon. The biodegradation rate for organic carbon has been estimated from literature data rather than direct measurements in Ward Cove and therefore is somewhat uncertain. Nevertheless, the estimates of recovery times presented here are regarded as the best practicable, given available data and a reasonable approach to model setup and validation.

9.3 CASE STUDIES AND EMPIRICAL DOCUMENTATION OF NATURAL RECOVERY

As described in PTI (1996), historical studies (mainly qualitative) of benthic macroinvertebrate communities in Ward Cove suggest that recovery may have been occurring slowly over the past 20 years. In 1968–1969, FWQA (1970) conducted macroscopic evaluations of benthic communities in the Cove and found few benthic invertebrates. Following installation of the primary treatment system for wastewater at the KPC facility, U.S. EPA (1975) conducted macroscopic evaluations of sediment samples in Ward Cove in 1974 and found that polychaetes were common at all locations except immediately offshore from the KPC facility. After comparing the 1974 benthic community results with those found in 1968–1969 (FWQA 1970), U.S. EPA (1975) concluded that is was apparent that some improvement had occurred in the bottom sediments. Finally, EVS (1992) evaluated benthic macroinvertebrate communities in the Cove in 1992 and found that most communities were dominated by the opportunistic polychaete *Capitella capitata.*

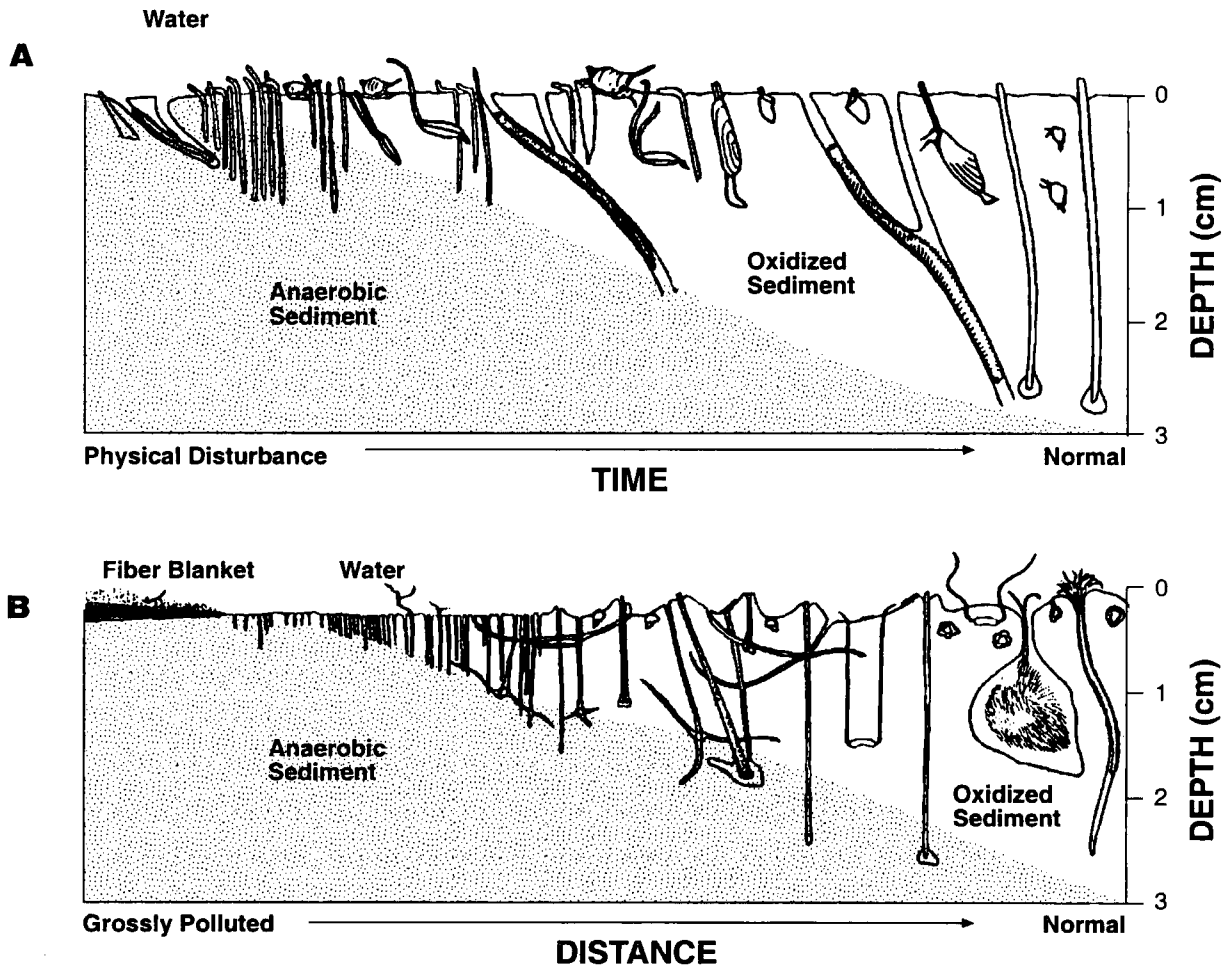
Because *C. capitata* is a well-documented indicator species for organic enrichment and one of the first species to colonize organically enriched sediments, its dominance of benthic communities in Ward Cove in 1992 supports the possibility that recovery was occurring.

Because most discharges from the KPC facility have been eliminated, recovery of benthic macroinvertebrate communities should proceed more rapidly than in the past and should follow the classical patterns of recolonization and recovery documented for organically enriched areas (Pearson and Rosenberg 1978) and dredged material disposal areas (Rhoads et al. 1977, 1978; Rhoads and Boyer 1982). Those patterns include initial colonization by "pioneering" species, subsequent modification of physical/chemical characteristics, and final colonization by deeper dwelling "equilibrium" species (Figure 9-12). In general, equilibrium species are associated with a deeply oxygenated sediment surface where the redox potential discontinuity commonly reaches depths of over 10 cm (Rhoads and Boyer 1982). The earliest benthic communities in the recovery process tend to consist of large numbers of a few species, whereas the equilibrium communities are characterized by a greater number of species and a more even distribution of individuals among species.

The first organisms to colonize a disturbed area are generally small, opportunistic, tube-dwelling polychaetes, followed by tube-dwelling amphipods (Rhoads and Boyer 1982). Most pioneering species feed near the sediment surface or from the water column and are thereby largely isolated from potentially toxic conditions in deeper sediments. The tube walls isolate the colonizing organisms from ambient surface sediments by controlling the diffusion rate of ambient porewater solutes into the tube environment (Aller 1982). In addition, by aerating the water in their tubes, organisms can effectively isolate themselves from oxidizable porewater constituents such as sulfide. In this manner, they can inhabit sediments that are toxic to free-burrowing organisms. The activities of the pioneering species modify the physical/chemical properties of the sediments so that additional species can colonize them. Such activities include bioturbation, irrigation, particle reworking, and progressively deeper penetration of subsurface sediments (Aller 1982).

Several aspects of the results of the Ward Cove sediment toxicity studies suggest that recovery of benthic macroinvertebrate communities in the Cove can occur more rapidly, now that most KPC discharges have been eliminated. Because the specialized toxicity test results indicate that sulfide may be the primary cause of toxicity in most sediments from the Cove, its rapid oxidation and subsequent toxicity reduction in the presence of oxygen suggest that pioneering tube-dwelling polychaetes and amphipods will be able to successfully colonize the surface sediments and isolate themselves from elevated sulfide concentrations in pore water by irrigating their burrows. After these pioneering species have colonized the surface sediments, the classical patterns of benthic recolonization and recovery should occur.

The ability of tube-dwelling organisms to successfully colonize Ward Cove sediments was demonstrated by results of the sediment toxicity tests conducted in 1996 using the tube-dwelling polychaete *Neanthes* sp. and the tube-dwelling amphipod *Leptocheirus*



Source for Inset A: Rhoads et al. (1977, 1978)
Source for Inset B: Pearson and Rosenberg (1978)

Figure 9-12. Classical patterns of benthic recolonization and recovery in disturbed environments.

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plumulosus. Tests based on both of those species showed that none of the sediments from the 28 stations sampled throughout the Cove were toxic, despite the fact that toxicity to the amphipod *Rhepoxynius abronius* was found at 16 (57 percent) of those stations. Because *R. abronius* is a free-burrowing organism, it cannot isolate itself from pore water and therefore would be expected to respond negatively to elevated concentrations of porewater sulfide. For *L. plumulosus*, the toxicity testing laboratory noted that a thin band of light brown sediment was found around the tubes of these organisms, compared to the general black color of the ambient sediment. It is likely that the brown sediments represented aerated sediments and that *L. plumulosus* was successfully isolating itself from the elevated concentrations of sulfide in the ambient sediment.

The time period over which benthic macroinvertebrate communities in Ward Cove can be expected to recover naturally can be estimated from historical studies that have monitored benthic recovery in areas affected by deposition of pulp mill material and deposition of sewage material. Four of the best documented cases of natural recovery following reductions in the discharge of organic material are related to the closure of a sulfite pulp mill in Sweden (Rosenberg 1976; Pearson and Rosenberg 1978) in 1966, the closure of a combined sulfite and kraft pulp mill in British Columbia in 1967 (Waldichuck 1988), the substantial reduction of discharges from a sulfite pulp mill in British Columbia (Cross and Ellis 1981), and the substantial reduction in sewage discharges off Los Angeles in 1970 (Swartz et al. 1986; Stull 1995). The results of those studies suggest that recovery of benthic macroinvertebrate communities in Ward Cove can be expected to occur within approximately 10 years. Each of the case studies is described below.

9.3.1 Pulp Mill Closure in Sweden

In 1966, a sulfite pulp mill located at the head of the Saltkallefjord in Sweden was closed. During operation, the mill had discharged large amounts of wood pulp fiber to the head of the estuary. At the time of mill closure, no benthic macroinvertebrates were found in the innermost part of the estuary, benthic communities in the middle portion the estuary were considered severely altered, and communities near the estuary mouth were considered largely unaffected by the discharges. This site is similar to Ward Cove in that it is a fjord type of estuary in a cold water marine environment, with pulp mill effluent solid accumulations at the head of the cove and unaffected areas nearby to provide recruitment stock.

The benthic macroinvertebrate communities in the Saltkallefjord were monitored for 8 years after the pulp mill was closed to document their recovery. Benthic species began recruiting into the middle portion of the Saltkallefjord (where communities previously had been severely altered) within 2 years of mill closure and into the innermost part of the estuary (where no organisms had been found previously) within 3 years of closure. Benthic succession was then rapid during the ensuing 2–3 years, and in 1971, the numbers of species in all parts of the inner and middle portions of the estuary were similar to the numbers of species found at the estuary mouth. After 1971, the numbers of species at all stations stabilized at the values found in 1971.

The polychaete *Capitella capitata* was the numerically dominant species of the severely altered communities in the inner half of the estuary prior to mill closure. This species also became a numerically dominant species in the formerly azoic areas in the innermost part of the estuary after mill closure. In all parts of the inner and middle portions of the estuary, polychaetes were the primary colonizing species during the first years of recovery, with echinoderms species becoming important later on. The wood pulp fibers that had accumulated in the sediment appeared to initially inhibit recolonization as their decomposition resulted in oxygen deficiencies. However, the fibers later became an important food source for benthic organisms through the probable mediation of bacterial decomposition. Rosenberg (1976) concluded that basic recovery of benthic communities in the Saltkallefjord occurred within 5 years of mill closure, and after 8 years of recovery, it was not possible to distinguish between a normal and a recovery-influenced community.

9.3.2 Pulp Mill Closure in British Columbia

In 1967, a combined sulfite and kraft pulp mill located at the head of Cousins Inlet in British Columbia ceased production, although a groundwood operation for newsprint production continued until 1980. During its 55 years of operation, the mill discharged effluent to the Link River, which ultimately drains to the head of Cousins Inlet. The seafloor near the head of the inlet was found to be covered with wood fibers that had been discharged from the mill. This site is also similar to Ward Cove in that pulp mill effluent solids were deposited at the head of an inlet in a cold-water environment. Continued operation of the groundwood facility in Cousins Inlet is a difference that is likely to lead to longer recovery times in Cousins Inlet than in Ward Cove.

Four benthic macroinvertebrate surveys were conducted between 1972 and 1975 to evaluate recolonization of the inlet. During the initial survey, no organisms were found at the head of the inlet, within approximately 2 km from the point at which the pulp mill effluent entered the inlet. In all four surveys, benthic communities in the inlet were dominated by polychaetes and the number of polychaete species per station increased with increasing distance from the head of the inlet. In addition, there was some indication that the overall numbers of polychaete species in various parts of the inlet increased between 1972 and 1975. A more recent benthic survey was conducted in Cousins Inlet in 1981, and it was found that benthic macroinvertebrates had begun colonizing the head of the inlet. However, the numbers of benthic taxa continued to be low within 3 km from the discharge point of the pulp mill, relative to stations located farther down the inlet. In addition, the total abundance of macroinvertebrates was found to be lowest at the station closest to the discharge point.

9.3.3 Pulp Mill Effluent Improvement in British Columbia

In 1977, a sulfite pulp mill located at Port Alice, British Columbia, implemented a major discharge abatement system for the collection and incineration of spent sulfite waste liquor, which was previously discharged to the head of Neroutsos Inlet. This site is

similar to Ward Cove in the type of impact and environmental conditions. Impacts at this site were apparently less than at the previous case study sites, because initial surveys did find benthic species living adjacent to the discharge. A shoreline biological monitoring program was implemented in 1978 and 1979 and included qualitative estimates of enchytraeid/tubificid oligochaete species, evaluation of epifaunal amphipod abundances associated with the kelp *Fucus*, and qualitative surveys of the benthic macroinvertebrates in the rocky shore communities of the inlet. Sampling stations were located from the head of the inlet (near the pulp mill) to immediately outside the inlet mouth, at increasing distances from the mill.

Fourteen species of enchytraeid/tubificid oligochaetes were found in Neroutsos Inlet. Results of a classification analysis based on oligochaete abundances showed a distinct cluster of species at the two stations closest to the pulp mill. At stations farther from the mill, a continuum of species assemblages was found with distance from the mill that could not be divided into distinct clusters. Cross and Ellis (1981) concluded that the oligochaete surveys documented a division into two distinct regions. The first region was represented by stations within 1 km from the mill and the second region was represented by the remaining farfield stations. The first region was characterized by *Lumbricillus lineatus*, an oligochaete found to occur in British Columbia only near pulp mills. Comparisons with prerecovery surveys indicated that *L. lineatus* was present at the farfield station closest to the mill in 1976, but was replaced by three other oligochaete species in 1978. The authors concluded that the biological recovery at that station was likely related to improvements in the quality of mill discharges that were implemented in 1977.

Sampling of the epifaunal organisms associated with *Fucus* showed that the gammaridean amphipod component of the assemblages consisted exclusively of *Allorchestes angusta* (an opportunistic species) at most stations within 10 km from the mill. However, no amphipods were found at the two stations located closest to the mill. At distances greater than 10 km from the mill, *A. angusta* was replaced by three other amphipod species. Comparison with prerecovery surveys showed that the peak abundance of the opportunistic amphipods moved closer to the mill each year between 1976 and 1979, with the highest abundance in 1979 occurring at the station closest to the mill (i.e., where no amphipods were found in 1976 and 1977). Cross and Ellis (1981) concluded that the movement of the peak of opportunists was likely related to improvements in the mill discharges and provided additional documentation of biological recovery in the area.

The species diversity of the rocky intertidal benthic macroinvertebrate communities showed a nearly monotonic increase with increasing distance from the pulp mill and included nearly linear increases in the number of species of major taxonomic groups such as gastropods, decapods, and asteroids. Results of a classification analysis conducted using these data also documented substantial changes in community composition with increasing distance from the mill.

9.3.4 Sewage Treatment Abatement in Los Angeles

The Los Angeles County Sanitation District serves over 5 million people in the Los Angeles area and has discharged sewage through an outfall to the continental shelf off of Palos Verdes for more than 50 years. This site differs from Ward Cove and the other case study sites in both the type of organic enrichment and in the type of environment (warm water vs. cold water; open circulation vs. confined). Nevertheless, it shows similarities to the other case study sites in the initial colonization by tubificid worms and in recovery in a period of a few years.

Beginning in 1970, substantial reductions in the emission of suspended solids (primarily organic matter) occurred following changes in treatment practices. In 1970, sediments near the outfall were highly enriched in organic matter and contained elevated concentrations of chemical contaminants. Benthic macroinvertebrate assemblages in the affected areas were severely degraded and dominated by opportunistic polychaetes such as *Capitella capitata* and *Schistomeringos longicornis*. Crustaceans and echinoderms were rare and numerous benthic species were absent near the outfall.

The benthic communities on the Palos Verdes shelf have been monitored since 1970. In general, the monitoring studies have documented recovery following the reductions in sewage discharges. These studies have found that the number of species and the balance of individuals among species have increased considerably. Monitoring results for one station near the outfall showed that the number of species per grab sample (0.1 m^2) increased from 16–24 species in the early 1970s to 48–56 species in the early 1980s and then fluctuated between 40 and 64 species from that period until the early 1990s. These results indicate that recovery of benthic macroinvertebrate communities on the Palos Verdes shelf had occurred approximately 10 years after substantial reductions in sewage discharges were achieved.

9.4 CONCLUSIONS

Natural recovery is an integral part of EPA's contaminated sediment management strategy (U.S. EPA 1998a). Ward Cove is an ideal site for considering natural recovery for all or part of the AOC for several reasons: the source of pulp mill effluent was eliminated with shutdown of the mill in 1997; the CoCs in sediments are natural products of organic matter degradation and are not persistent as are chemicals such as metals and hydrophobic organic compounds; sediment chemicals are within acceptable limits for human health and wildlife and of limited toxicity to the benthos; and existing sediment and hydrodynamic modeling indicate that offsite sediment transport is not a concern. Conclusions regarding chemical and biological recovery in Ward Cove, and predicted recovery periods, are presented below.

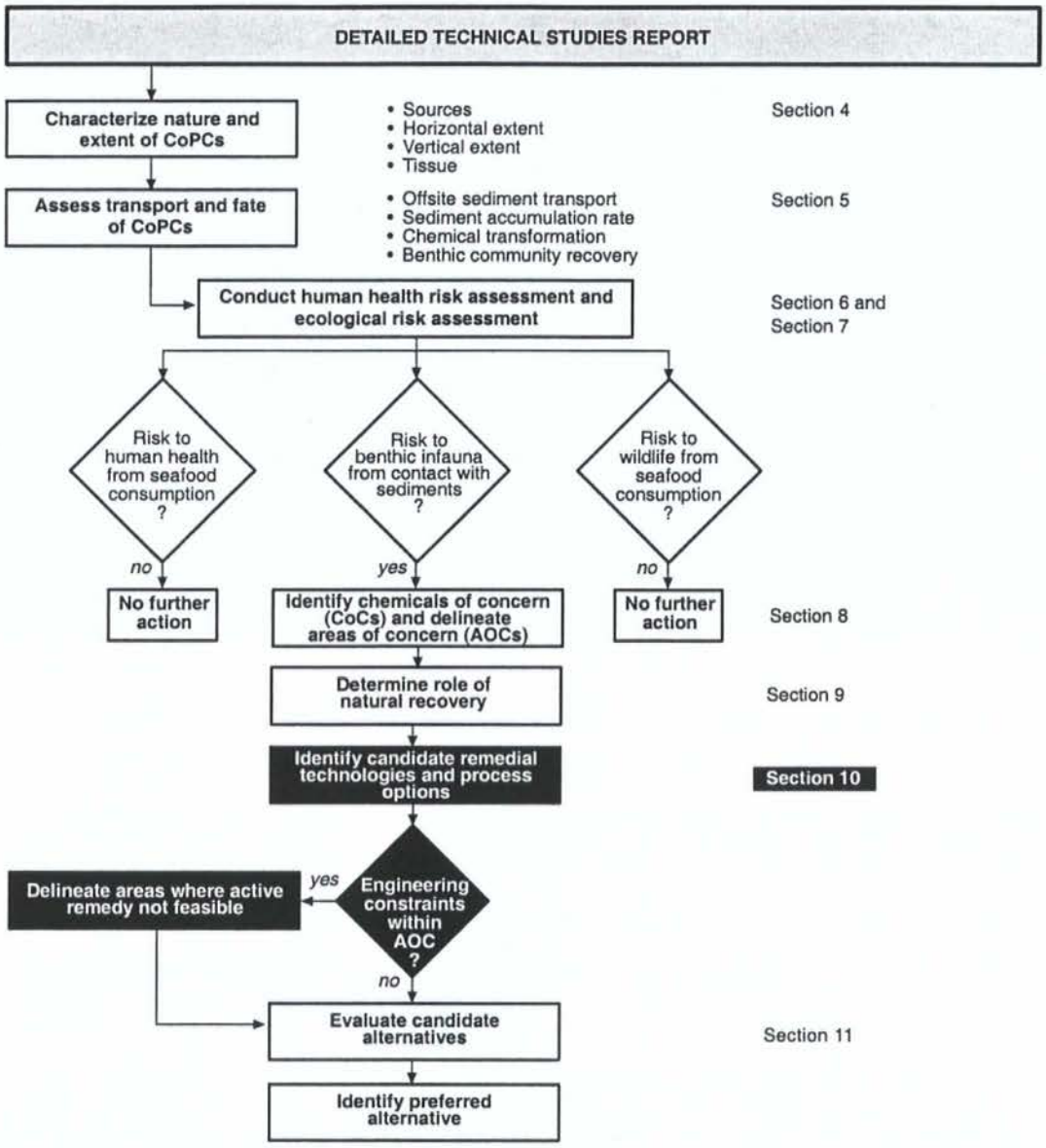
Numerical modeling of quantifiable natural recovery processes indicates that the recovery period is likely to be longest directly offshore of the KPC mill and along the north shore to the west of the mill. Recovery of ammonia and 4-methylphenol to levels below the sediment quality values used in the numerical modeling is expected to take more than

20 years in this region. Recovery of sulfide, in contrast, is expected to require less than 8 years. The absolute durations of the predicted recovery periods are somewhat uncertain, as a result of their dependence on the organic carbon decay rate, for which there are no site-specific data. Differences in degradation rates of effluent solids and woody debris, in particular, could result in either an increase or decrease in the recovery period. In addition, the steep slopes along the north shore of Ward Cove, the spatial variability of sediment deposition processes, and the positive feedback between chemical and biological recovery processes may all reduce recovery periods from those predicted. Despite these limitations in the model results, the predictions of areas requiring extended natural recovery periods are consistent with each other, consistent with 1996 and 1997 field data, and plausible with regard to current knowledge of conditions in Ward Cove. Because organically enriched sediment is confined to the inner part of Ward Cove even after decades of mill discharges and because field measurements indicate that there is little potential for sediment transport (Section 5.1), the areal extent of affected sediment is not expected to increase as a result of sediment transport during the recovery period.

Results of the specialized toxicity tests, observations made on the benthic communities in Ward Cove, and case studies of other sites with organic-rich sediment provide compelling arguments for natural recovery in a reasonable time frame. The potential for benthic macroinvertebrate communities to recover naturally in Ward Cove is supported by the results of the specialized toxicity tests. Specifically, the results of the specialized toxicity test on pore water using *Rhepoxynius abronius* (Section 7.1.4) indicate that sulfide appears to be the major cause of sediment toxicity in sediment samples from most areas of the Cove and that simple aeration can render most sediments nontoxic.

The observed characteristics of benthic macroinvertebrate communities in Ward Cove are consistent with those documented for organically enriched areas and dredged material disposal areas in other studies, in which initial colonization by opportunistic species is followed by colonization by equilibrium species. The likely pattern of future recolonization in Ward Cove is illustrated by the results of several case studies of recolonization following closure of a pulp mills in Sweden and British Columbia, improvements in the effluent quality of a pulp mill in British Columbia, and sewage treatment abatement in Los Angeles, California. Based on the theoretical models of benthic recovery and the results of case studies, recovery of benthic macroinvertebrate communities in Ward Cove is predicted to occur within approximately 10 years.

10. TECHNOLOGY SCREENING



Once the AOC has been identified (Section 8) and the potential for natural recovery evaluated (Section 9), remedial technologies that are potentially appropriate can be evaluated. The limited hazards and unique characteristics associated with sediments within the AOC are critical considerations throughout the evaluations of candidate technologies (this section) and candidate remedial alternatives (Section 11). In the following subsections, potential remedial technologies are introduced, site-specific constraints and screening criteria are described, and remedial technologies and disposal locations are evaluated. This section concludes with a summary of technologies that are potentially

applicable to sediments within the AOC. Technologies are combined into candidate remedial alternatives in Section 11 and subjected to a more detailed evaluation.

10.1 OVERVIEW OF POTENTIAL SEDIMENT REMEDIAL TECHNOLOGIES

Sediments that do not currently meet the RAOs can be dredged, treated or capped in place, or left to recover naturally. If sediments are to be dredged, they could be disposed of in various ways. The range of disposal options that should be considered includes upland disposal (in an appropriate landfill or by land application), near-shore disposal (in a constructed facility along the shoreline), and confined aquatic disposal (CAD) (which includes confinement by capping in place with clean material). Treatment is an option that is usually considered only for sediments with high concentrations of persistent substances that are toxic or have the potential to bioaccumulate. An overview of sediment remedial technologies and process options is summarized in Figure 10-1.

10.1.1 Dredging

The removal or excavation of sediments from a water body, commonly called dredging, is a routine process. The most common purpose of dredging operations is to remove large volumes of subaqueous sediments as efficiently as possible within a specified operational and environmental restriction (Palermo and Hayes 1992). The term "environmental dredging" has evolved in recent years to distinguish dredging operations for the purpose of environmental remediation from maintenance or navigational dredging. Environmental dredging operations, such as those being considered in Ward Cove, must attempt to remove problem sediments as effectively as possible, while minimizing environmental risk and other adverse consequences.

Dredging involves active disturbance of the bed to dislodge sediment by mechanically penetrating, grabbing, raking, and cutting or by hydraulically scouring by water jets. Once the bed sediment is dislodged, the sediment is transported to the water surface mechanically (e.g., by clamshell) or hydraulically (e.g., by pipe slurry). Dredges are categorized as either mechanical or hydraulic, depending on the method of transporting the sediment. Various types of dredges have been described elsewhere (Averett and Francingues 1994; Zappi and Hayes 1991; U.S. EPA 1996d).

10.1.1.1 Hydraulic Dredging

Hydraulic dredges are usually barge-mounted systems that use centrifugal pumps to remove and transport the sediment and water mixture. Pumps may be either barge-mounted or submersible. The cutterhead dredge, a type of hydraulic dredge, uses a mechanical device (called a cutterhead) for dislodging the sediment (U.S. EPA 1993d). Resuspension at the cutterhead is a common problem for a hydraulic dredge working in fine-grained sediment. Certain hydraulic dredges such as the "clean-up," "matchbox," "refresher," or "modified dustpan" typically add an enclosure around the suction end of

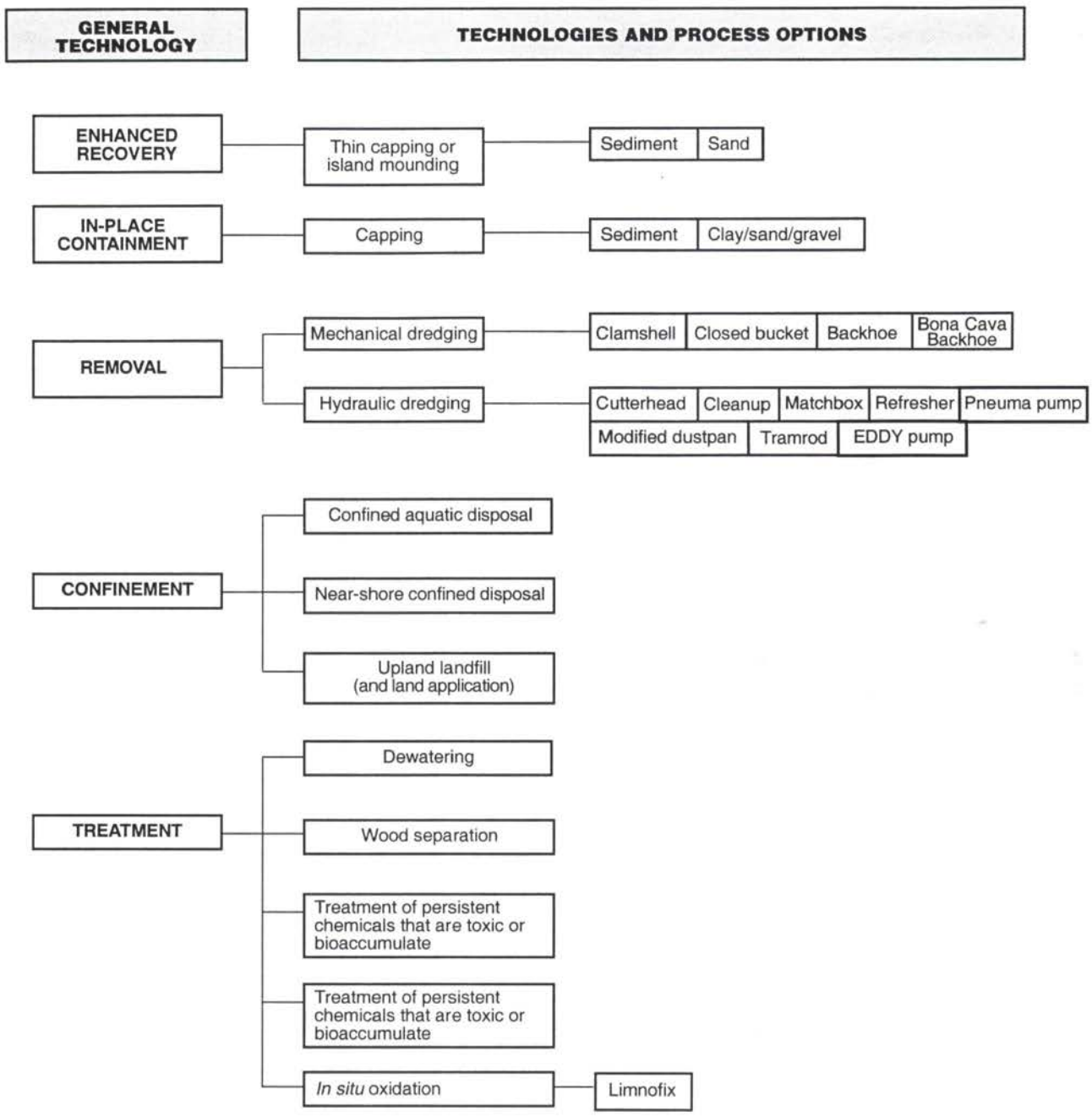


Figure 10-1. Potentially applicable technologies and process options.

the dredge to reduce resuspension of sediment. Hydraulic dredges pump the bed sediment to the surface where it is usually transported via a pipeline to a confined disposal facility, either near-shore or upland.

To hydraulically dredge in depths greater than 80 ft, it is necessary to use specialty dredges. Standard hydraulic dredges are now equipped with submerged pumps to allow deeper dredging than in the past. The submerged pump located on the dredge ladder allows hydraulic dredging to depths approaching 100 ft.

The Tramrod dredge is a special hydraulic dredge unit that operates on the bed of deep water sites by remote control. The dredge has a submerged pump and a small rotary cutterhead or auger head mounted on a platform that is propelled by two rolling tracks much like a small bulldozer. This unit can operate on fine sediment beds typical of deeper water, where limited trash, debris, or steep slopes would exist. The unit's movement is controlled from operators on the surface that use video and a global positioning system to control the location. The bed sediment is pumped to the surface for transport and disposal.

The EDDY Pump and the Pnuema pump are variations of hydraulic dredging that use a vortex pump or air pressure to suction sediments without entraining as much water as conventional hydraulic dredges. Although designed to dredge at depths up to 30 m, the equipment is comparatively small, with a maximum discharge pipe diameter of only 0.36 m and a maximum production rate of about 100–250 m^3 per effective hour. The vortex pump (used in the EDDY Pump) consists of a rotor inside a volute with a suction nozzle and discharge outlet (Harrison and Weinrib undated). The only moving parts are the rotor and its shaft (which is turned by electric or hydraulic motors, with power supplied by diesel generator). The volute and suction tube are designed to create a vortex at the inlet, so that sediment can be drawn into the pump without using water jets or cutterheads to first loosen sediment.

10.1.1.2 Mechanical Dredging

A mechanical dredge uses equipment such as a clamshell bucket to excavate material from the bottom and haul it to the surface, where it is placed either directly into a confined disposal area or into a barge or truck to be hauled to a disposal site. The mechanical dredging process adds substantially less water to the dredged sediment relative to hydraulic dredging, but is generally thought to operate in a manner that leads to higher resuspension rates in the water column.

A variation of the conventional bucket, the enclosed dredge bucket, has been developed to limit spillage and leakage from the bucket (Hartman and Goldston 1994). Enclosed bucket dredges have been used routinely in various Great Lakes ports for the maintenance of navigation channels. They have also been used in sediment remediation projects in the Black River near Lorain, Ohio, in 1990; the Sheboygan River, Wisconsin, in 1990 and 1991; and in the Brazos River channel in Freeport, Texas, in 1992. Use of the cable arm bucket (an enclosed bucket) was demonstrated by Environment Canada on

contaminated sediments in the Toronto and Hamilton harbors in Canada (Buchberger 1993).

Other mechanical dredges, such as backhoes or dipper dredges, can be used for removing problem sediments under certain circumstances. The Bona Cava, a specialty backhoe dredge system, was developed and used to remove contaminated sediment from Bayou Bonfouca in Louisiana (Taylor 1997, pers. comm.). This specialty dredge had automated controls that allowed sediment removal to within 3-in. accuracy from a water depth of approximately 35 ft. The Bona Cava is the only dredge that has been actually operated to such narrow vertical tolerances. The typical vertical accuracy of dredging for mechanical dredges at depths less than 50 ft is 2–3 ft.

10.1.2 In-Place Capping

In-place capping is the most straightforward and least intrusive of sediment remedial techniques. Capping material, typically clean sediments, sand, or silty to gravelly sand, is placed on top of problem sediments (Figure 10-2). Capping material is generally brought to the site by barge and put in place using a variety of methods, depending upon the selected remedial action alternative. The issues generally associated with in-place capping are 1) obtaining an appropriate cap thickness over the entire problem sediment area, 2) placing the capping material without displacing sediment, and 3) maintaining long-term cap integrity. Placement methods include the following:

- *Surface release from barges* is a technique where the clean sediment is slowly released from a split hull barge as the barge is slowly towed over the problem sediment area.
- *Tremie tube or submerged diffuser placement* of capping material is a method to control the capping material as it passes through the water column for deep water capping sites. A tremie tube is a large diameter tube, usually greater than 10 ft in diameter. The tube contains the material as it travels through the water column and allows for relatively accurate placement. A diffuser is a velocity dissipater used with a submerged hydraulic pipeline discharge. The material is pumped from the barge and discharged through the diffuser or tremie tube placed under water, near the bed surface.
- *Hydraulic washing* is a technique where the clean sediment is washed off of a barge with large water hoses. This technique has been successfully used in shallow water (10–30 ft) at the Eagle Harbor project at Bainbridge Island, Washington, where bed material was predominantly sandy silt and silty sand. This method allows the clean sediment to rain down over problem sediment.
- *Pipeline with baffle box or diffuser placement* of capping material uses a pump-out system to transport the capping material from the barge to the capping area. The material is pumped from the barge through a

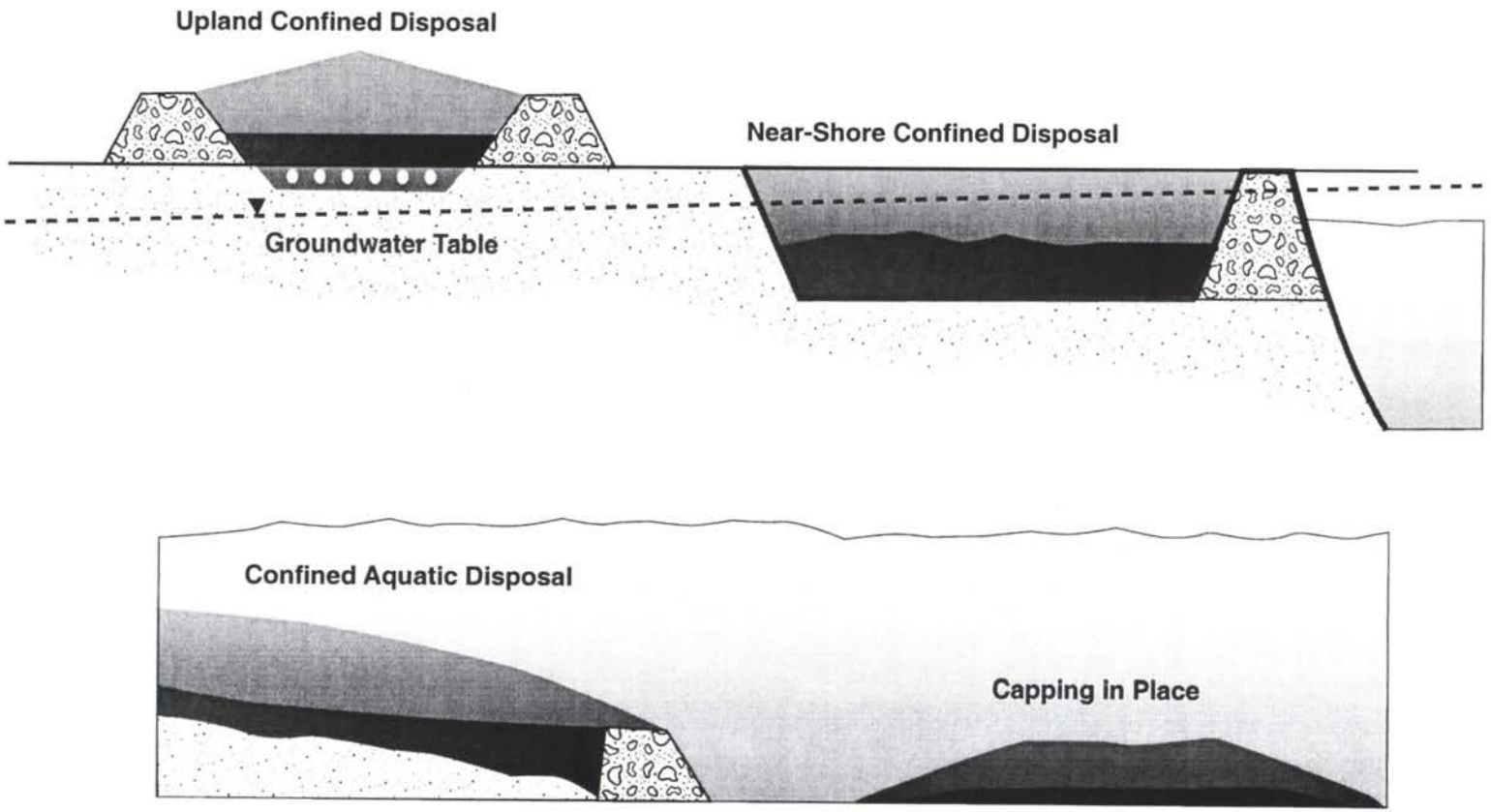


Figure 10-2. Confined disposal alternatives

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floating pipeline and into either a surface baffle box or a submerged diffuser, which reduces the slurry velocity and allows the capping material to fall gradually over the problem sediment area.

- *Direct mechanical placement* of capping material uses a clamshell dredge to lower and release the capping material near the bed surface.

There are two general categories of capping: thick capping and thin capping. The goal of thick capping is to isolate problem sediment and replace the benthic habitat. The thick cap is typically a minimum of 3 ft thick. The goal of thin capping, also known as enhanced recovery, varies depending on the type of site. For some environments, the goal could be isolation, but to a more limited extent (e.g., a 1-ft target thickness) than is typical for thick capping. In other environments, the goal could be to improve the chemical or physical properties of surface sediments, which constitute the biologically active zone. With thin capping, surface coverage is expected to vary spatially, providing variable areas of capped surface sediments and amended surface sediment (i.e., where mixing between capping material and problem sediment occurs) as well as areas where no cap is evident. Thin capping offers an option for environmental improvement in areas where the environmental or human health hazard posed by sediments is low.

In addition to thin capping, other capping approaches could be used to implement enhanced recovery. For example, the cap material could be placed as a series of mounds that extend out of the soft, organic sediments. Spot dumping of capping material could be conducted to create a discontinuous, island-like cover that provides areas where benthic communities would likely recover at an accelerated rate. The sand (or other capping material) could be placed carefully, one clamshell at a time, to create the isolated mounds of sand.

10.1.3 Confined Disposal of Dredged Material

Confined disposal sites fall into three general categories, depending on their general location and/or future use: upland, near-shore disposal, and CAD (Figure 10-2). Geotextile bag containment is also described.

10.1.3.1 Upland Disposal

Dredged sediment could be placed into approved upland landfills for disposal. Sediments within the Ward Cove AOC would not be classified as hazardous wastes because of the types of CoCs present and the low concentrations (discussion in Appendix L). The sediments could be disposed in a solid waste landfill pending landfill operator and agency approval. Upland disposal would involve sediment removal by dredging and transport by truck or barge to an off-loading site near a landfill with the capacity to accept the sediment.

Another possibility for upland disposal is application of the dredged sediments to agricultural lands. This type of disposal could also be considered a treatment technology depending on the type of problem sediments being remediated and the objectives for applying it to the land. The disposal area would need to be sufficiently large and dry so the sediments could be applied in thin layers and runoff could be prevented. Agricultural lands in arid regions would be most appropriate to accommodate large volumes of sediment with high water content. For marine sediments, placing them in thin layers would be necessary so that rainwater could leach out the salts.

10.1.3.2 Near-Shore Confined Disposal

Near-shore confined disposal facilities (NCDFs) are constructed adjacent to the shoreline. The problem sediment is confined using retaining dike structures that are constructed to extend out of the water. The problem sediment can be placed into the NCDF by a variety of methods. These methods include release from a split hull barge, direct mechanical placement, hydraulic placement via a pipeline directly from a dredge, and slurrying of mechanically dredged material in the barge with subsequent pumping over the dike into the NCDF. Depending on the placement method, a temporary opening in the retaining dike may be used to allow access by the disposal barges during subsurface placement of the problem sediment. Typical retaining structures are berms (constructed with sand, sandy gravel, or other fill material) and sheet-piling structures. Finer-grained sediment such as sand or silty sand is used as a core to ensure minimum movement of soluble contaminants through the berm. For problem sediments containing mobile, toxic metals, the problem sediment is placed below groundwater level to keep the metals bound to sediment and in a stable anaerobic environment. Loss of contaminants in this condition occurs only from groundwater movement through the contaminated fill and the capping materials. Once the sediment has settled, the site can be filled to grade and put to a variety of uses. NCDFs are distinguished from CADs in that the final grade of an NCDF allows for future upland use.

NCDFs are one alternative with the lowest impact on water quality; however, constraints on berm and sheet pile construction are a critical consideration (see Appendix K for a more detailed discussion).

10.1.3.3 Confined Aquatic Disposal

CAD is the placement of dredged material followed by capping material in an aquatic (i.e., submerged) disposal site. Problem sediment is either placed on the bottom in a mound and then covered with clean material to create a CAD site or it is placed within a subaqueous bermed area on the bottom and then clean material is placed within the berm over the problem sediment to create a CAD site. The thickness of the cap is based upon the need to limit convection of chemical contaminants through the cap, prevent biological contact with the sediment, and resist erosion forces. The issues associated with CAD site capping are the same as those for in-place capping: 1) obtaining a sufficient cap thickness over the entire area, 2) placing the capping material without displacing the problem

sediment, and 3) maintaining long-term cap integrity. In high energy environments or areas where navigation may impact the cap, a suitable armor layer of gravel or rock is required.

10.1.3.4 Geotextile Bag Containment

Geotextile bag containment has been used in conjunction with hydraulic dredging to provide temporary containment of problem sediment. In a typical process, permeable geotextile fabric bags are placed inside barges and then the sediment slurry is pumped into the bags. When the bag is full, the open end is sewn shut and the bag is ready for disposal. The geotextile bags are custom made for each project but are generally the length and width of the barge bottom opening. At the disposal site, a bottom dump barge is opened to allow the geotextile bag filled with sediment to fall out.

10.1.4 Sediment Treatment

Sediment can be treated in a variety of ways, ranging from simple dewatering techniques or wood separation methods to more elaborate treatment technologies that are designed to immobilize or eliminate hazardous constituents.

10.1.4.1 Dewatering

Sediments may be dewatered for remedial alternatives that involve dredging and upland disposal or as a pretreatment step prior to additional treatment. Dewatering may be conducted using settling basins, clarifier tanks, or filter presses. The determination of whether dewatering is needed and, if so, the type of dewatering, is dependent on the characteristics of the sediments and the ultimate disposal or treatment method.

10.1.4.2 Wood Separation

Technologies to separate wood debris and bark from sediments are also available. The goal of separation processing is to separate wood or bark from sediment, so that these materials can be reused independently. Separation is typically accomplished using conventional material processing equipment such as vibrating screens, conveyors, or flotation separators. The work could be done on barges or at an upland work site.

For cases where logs are mixed with sand and fine-grain sized sediment, screens can be effective in separating material. Material sorted by screens of various sizes is then carried off by separate conveyors leading to separate stockpiles. Sediments with finer organic material can be separated by flotation chambers where material lighter than water floats to the surface and heavier sediment settles to the bottom. Following flotation separation, the wet materials are sent to dewatering equipment and then sent to separate

stockpiles. Separation costs generally range from $10/cubic yard (cy) to more than $70/cy.

10.1.4.3 Treatment of Hazardous Substances

Sediment with elevated concentrations of highly hazardous substances can be treated by a variety of methods, depending on the sediment characteristics, specific contaminants, and levels of contamination. Potential sediment treatment technologies and process options are the same as those used for upland solid waste (either soil, sludge, slag, or debris). The main difference between marine sediment and soil is that marine sediments are mixed with salt water and the sediments have a much higher initial water content than upland soil. Ward Cove marine sediment has greater similarity to pond sludge than to typical upland soil, because of the high organic matter content and the high water content.

Sediment or soil treatment can be grouped by the type of treatment. The most common categories are thermal treatment, chemical treatment, biological treatment, extraction, and stabilization (Garbaciak 1994). All of these types of treatment are designed to remove high concentrations of hazardous substances from soil or sediment. Most research and development have been focused on treating soil with contaminants that are the most toxic and most mobile in the environment, because they pose the greatest threat to human health or the environment.

Soil can be treated in place or after it is excavated and removed (Averett and Francingues 1994). Sediment, however, is rarely treated in place because of the difficulty in working under water. Based on experience with controlling dredging accuracy, it may not be possible to effectively treat sediment in water depths exceeding 20–30 ft because of the difficulty in accurately controlling the treatment equipment or chemical additions (Swatko and Berry 1989). The National Water Research Institute of Environment Canada and Limnofix Inc. (a member of the Golder Associates group of companies) are developing technologies to treat contaminated sediment in place (*in situ*) by injecting oxidants and amendments into the near-surface sediment to obtain specific chemical and physical conditions (Golder 1998). The *in situ* sediment remediation technology is sometimes referred to as the "Limnofix" technology.

10.1.5 Log Removal

Removal of logs exposed on the bottom surface may be needed in selected areas of Ward Cove to facilitate or complement dredging. A mechanical dredge mounted on a derrick barge could be used to remove logs from Ward Cove. The logs exposed on the bottom surface could be picked up with a grapple (orange peel) or standard 5-cy clamshell bucket. The logs would be transported to shore by haul barge where they can be disposed of appropriately. Three potential options for reuse or disposal of the logs are evaluated in this report: chipping the logs for use in Ketchikan, chipping the logs for use in the Puget Sound region in Washington State, or sending the logs to a solid waste landfill in Washington State.

10.2 SITE-SPECIFIC CONSTRAINTS AND SCREENING CRITERIA

As in any project, site-specific conditions in Ward Cove serve to immediately narrow the choices in the decision-making process. The unique physical and chemical characteristics of Ward Cove sediments play an integral role in identifying and evaluating potentially applicable technologies and process options. The physical features of Ward Cove are also important in determining whether a particular technology or disposal site is feasible. Additional site-specific constraints include the isolated location of the Cove, (i.e., the Ketchikan area is accessible only by air or water and all material transport must be conducted by haul vessels), limited local road access, a steep surrounding terrain resulting in limited suitable upland area for disposal or treatment, and lack of a local source of clean capping material.

In the following subsections, the physical and chemical properties of Ward Cove sediments and the major features of Ward Cove are described. The properties of the sediment and features of the Cove are then used to develop criteria that are used to screen sediment remedial technologies and process options later in this section.

10.2.1 Important Properties of Ward Cove Sediments

In Sections 6 and 7, it was demonstrated that the chemicals in Cove sediment are within acceptable limits for human health and wildlife. A limited risk to benthic infauna is observed (i.e., a limited degree of sediment toxicity is observed); however, a benthic community is present, with characteristics consistent with those documented for organic-rich areas. CoCs are ammonia, sulfide, and 4-methylphenol, all natural degradation products of organic matter and wood debris. Concentrations of persistent chemicals or chemicals that bioaccumulate (e.g., mercury or PCDDs/Fs) are low and within acceptable limits for human health and wildlife. The cessation of pulping activities in March 1997 (i.e., elimination of pulping effluent), the nature of the CoCs, and the effectiveness of natural recovery processes (demonstrated in Section 9) all indicate that aggressive remedial efforts are not warranted.

The physical characteristics of the sediments are also important considerations. Sediments affected by releases from the KPC facility are distinctly different from the underlying native sediments and from sediment in most marine environments. The sediment horizon affected by historical wood pulping releases with historical and ongoing log handling activities is generally found near the head of the Cove offshore of the KPC facility and along the north shore and is generally 4–9 ft thick. Affected sediments contain wood debris, have high water and organic content, and are black in appearance. The TOC content of these sediments is high, typically ranging from 20 to 40 percent.

A number of tests were performed on sediment samples taken from Ward Cove to define the engineering properties of the sediment. This information is important for the development and analysis of sediment management alternatives. The following is a discussion

of the tests conducted on the sediments, the findings of the tests, and their implications for sediment management. Soil Technology, Inc., performed water content, specific gravity, void ratio, volatile solids, grain size, Atterberg limits, column settling, consolidation, and elutriate tests on representative sediment samples. The laboratory test results are presented in Appendix A5.

There are two general types of sediment tests. The first are "index" tests which are used to classify sediment. Index tests performed for this project include water content, specific gravity, void ratio, volatile solids, grain size, and Atterberg limits. The second type are engineering tests, which are used to directly measure characteristics of sediments that affect settlement rates, settlement magnitudes, and dewatering ability. Engineering tests performed or attempted for this project are the column settling, consolidation, elutriate, and desiccation tests.

10.2.1.1 Index Test Results

The results of the index tests are presented below, followed by an explanation of the significance of the results relative to this project. Values for Commencement Bay in Washington State (Port of Tacoma 1992) are used to represent typical marine bed sediments.

- **Water Content**—The water content generally ranged from 290 to 660 percent, with one sample at 137 percent.[9] In Commencement Bay, water content tests on 254 samples showed an average of 50 percent, with a standard deviation of 26 percent, compared with 290–660 percent for Ward Cove sediment. In Commencement Bay, composite samples of silty fine sand had water contents of 41–48 percent and composite samples of clayey silt had water contents of 62–79 percent. The implication of the high water content in Ward Cove is extremely important to the development of screening criteria for dredging and sediment management. The target sediment has a high water content and it lacks strength and stability, indicating that it will not likely support capping material.

- **Specific Gravity**—The specific gravity of the dry solid material ranged from 1.93 to 2.52, much lower than the specific gravity of inorganic sediment and rock, which typically has a narrow range of 2.6–2.7.

[9] Water content can exceed 100 percent because it is calculated by dividing the weight of water by the weight of *dry* solids. For example, a sample that weighs 400 g and contains 300 g of water and 100 g of dry material has a water content of 300 percent.

- **Void Ratios**[10]—The void ratios generally varied from 7.2 to 14.4, with one sample at 3.59, which means that the volume of water in the samples was 7–14 times the volume of the same material after drying and compaction. In comparison, the Commencement Bay sediment void ratios were 2–4, or 2–4 times the volume after drying and compaction.

- **Volatile Solids**—The volatile solids ranged from 16 to 86 percent, which means that 16–86 percent of the weight of *dry* solid material volatilized when heated to 440°C. In Commencement Bay, tests on 238 samples showed that the average percent volatile solids was 3.0 percent, with a standard deviation of 1.8 percent, compared with 16–86 percent for Ward Cove sediment.

- **Grain Size**—The sieve analyses show that the organic sediment particles are predominantly in the range of sand-size (between 0.06 and 4.75 mm in diameter), although the percentage of grain sizes finer than sand ranges from 9 to 61 percent.

- **Atterberg Limits**—The Atterberg limits for sediments are the liquid limit, plasticity limit, and plasticity index, which is the difference between the liquid limit and plasticity limit. The tests show that the liquid limit ranged from 92 to 324 percent and the plasticity limits ranged from 32 to 234 percent. Typical sediment and soil has liquid limits below 100 percent and plasticity limits below 60 percent (ASTM D-2487 Standard Classification of Soils for Engineering Purposes, Unified Soil Classification System).

The index tests clearly demonstrate that the Ward Cove sediments are atypical. The high water contents and corresponding high void ratios mean the sediments are predominantly water, not solid matter. The high water content indicates that the sediment has extremely low shear strength and would be highly compressible.

The significant amount of volatile solids and low specific gravity indicate that the sediment consists of organic material that has a much lower density than typical sediments. The high Atterberg limit values also are indicative of organic material, rather than typical sediment or soil. Plasticity indices greater than 100 demonstrate that the wood fibers can adsorb several times their weight in water. Organic material with more water than dry solids would be very difficult and costly to dewater and dispose in upland disposal sites compared to more typical sediments such as those in Commencement Bay. These test results are consistent with the observations in the sediment core logs that described the sediment as organic in nature with a very high water content.

[10]The void ratio is the ratio of the volume of air and water in the sample divided by the calculated volume of solid, completely compacted dry material.

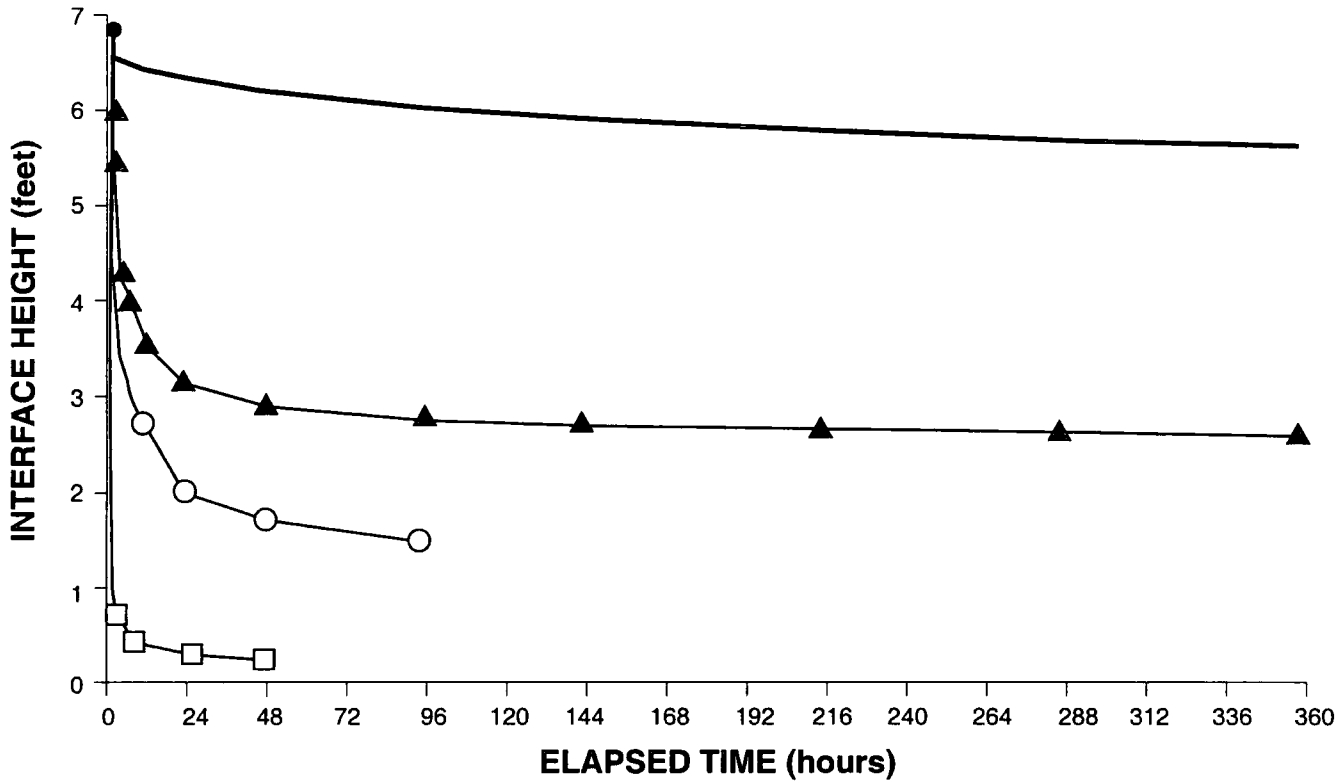
Each of these results plays an important role in the screening of dredging and disposal technologies by limiting or excluding those alternatives that involve significant handling, capping, and/or dewatering. From an engineering and remediation perspective, these sediments have limited strength or have essentially no strength, depending on the water content. The difficulty in dredging, transporting, and disposing this very soft material limits the range of feasible remedial options. Placing cover material over the extremely soft, organic, fine-grained sediments presents a number of problems and further limits the range of feasible remedial options. The tests also indicate the need to include a screening criterion limiting disposal on slopes, because the material is unstable and will be displaced, especially if capping on slopes is attempted.

10.2.1.2 Engineering Tests

The results of the engineering tests are presented below, followed by an explanation of their significance to this project.

- **Column Settling**—The purpose of the column settling test is to measure the rate that the sediment would separate from water by gravity settling in an upland or NCDF site. In this test, an interface develops between relatively clear water in the upper top of the column and the slurry below the interface. Figure 10-3, *Comparison of Elapsed Time vs. Interface Heights*, shows the results of the column settling test for the Ward Cove sediment compared to clayey silt and silty fine sand from Commencement Bay. As shown, the Ward Cove slurry was still 3 ft thick in the 6-ft high column after 360 hours (15 days). By contrast, the silty fine sand from Commencement Bay dropped to less than 0.5 ft within 24 hours. The clayey silt from Commencement Bay dropped to about 1.5 ft within 96 hours (4 days). The results of the column settling tests are consistent with the index tests that show the sediment consists of organic wood fibers. The sediment adsorbs significant amounts of water and essentially does not separate from water by gravity settling.

- **Consolidation**—This test is conducted to determine the magnitude of self-weight settlement of the sediment. The results of this test are used to determine how much consolidation of the disposed problem sediment should take place before capping an upland or NCDF and how much water would be released and have to be collected, tested, and if necessary treated before being discharged. Figure 10-4, *Comparison of Consolidation Test Results*, shows the results of the Ward Cove sediment compared to typical sediments from Commencement Bay. The results indicate that Ward Cove sediment would be compressed by 25 percent (an axial strain of 0.25) at the relatively low stress of 0.2 tons/ft^2. In contrast, sediments from Commencement Bay had essentially no compression at the same stress. As the sediments are compressed, large volumes of water would be released, which would



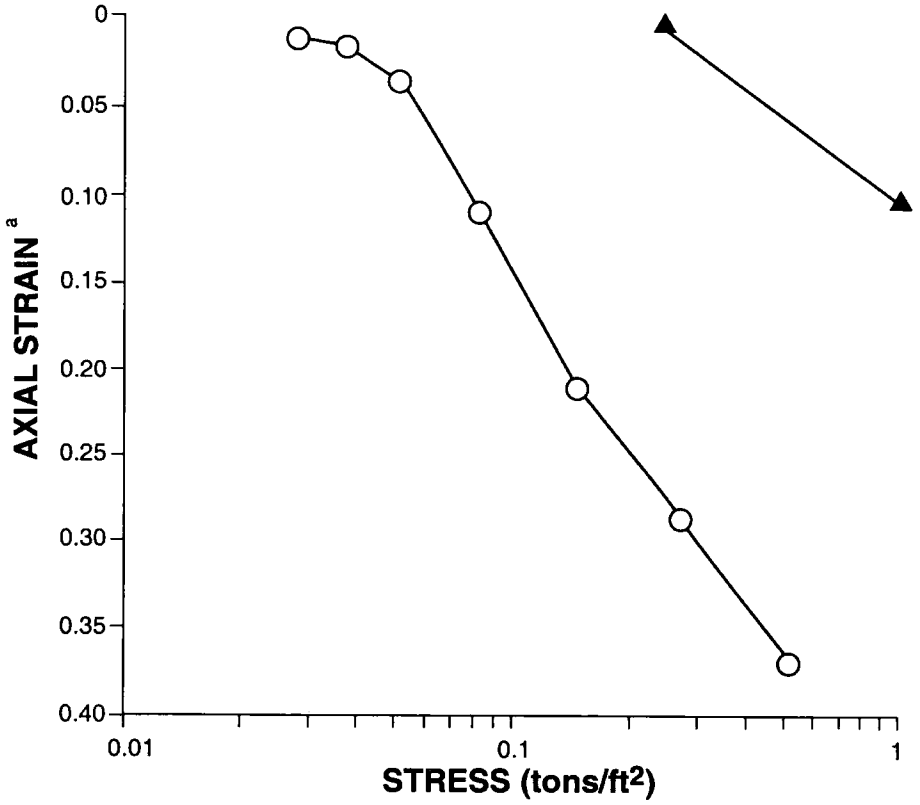
LEGEND

- Water surface height[a]
- Sediment from Ward Cove
- Test initiation
- Clayey silt from Commencement Bay
- Silty fine sand from Commencement Bay

[a] Apparent decrease in water surface height reflects cumulative loss due to sample extractions.

Note: Testing is discontinued when the interface height is stable (no further settling).

Figure 10-3. Comparison of elapsed time vs. interface heights for column settling analyses.



LEGEND

Sediment from Ward Cove

Clayey silt from Commencement Bay

[a]Calculated as: $\frac{\text{(Initial column height – Final column height)}}{\text{(Initial column height)}}$

Figure 10-4. Comparison of consolidation test results.

have to be collected, tested, and if necessary treated before being discharged.

- **Modified Elutriate Test**—The MET is used to determine the quality of the effluent that would be returned to waters of the United States from placement of dredged sediment into confined disposal areas. The test is important if sediment is to be dredged by hydraulic dredging and discharged into a confined near-shore or upland site that has an overflow weir. Specifically, the MET is used to determine TSS and soluble and total CoCs that would be released from the disposal site. The results of the tests for Ward Cove sediment indicate that a mixing zone would likely be needed where effluent is released from the disposal site because of the relatively high TSS detected in the supernatant (229 and 338 mg/L).

- **Dredging Elutriate Test**—The DRET provides an indication of the relative resuspension of sediment that would occur at the point of dredging. The DRET used approximately 10 percent sediment by volume, with 1 hour of mixing and aeration and 1 hour of settling. The DRET is considered to be an optimistic indicator of sediment resuspension (low resuspension). If significant amounts of TSS are measured using the DRET, dredging activities are likely to cause significant resuspension. The results of the tests for Ward Cove sediment indicate that a mixing zone would likely be needed at the point of dredging because of the relatively high TSS detected in the supernatant (140 and 167 mg/L).

- **Desiccation**—The purpose of the desiccation test is to determine the rate at which the sediment will dry so that it can be handled as a solid or semi-solid material. This test is important if the sediment is to be transported and disposed of at an upland site or applied as a soil amendment. Desiccation tests were started but were not completed because of the length of time required to dry the Ward Cove wood waste sediment by simple air drying. It was determined that air drying a thin layer of this sediment would likely require several months because the outer surface dries and forms a solid crust that prevents the interior sediment from drying. By contrast, drying a thin layer of typical sand sediment requires 1–2 days, and drying a thin layer of typical clay sediment requires 2–4 weeks.

The nature of Ward Cove sediment has a major impact on both the technical and economic feasibility of remediation technologies. The column settling tests indicate that Ward Cove sediment cannot be dewatered using conventional gravity settling methods, which is essential for cost-effective hydraulic dredging. The consolidation results demonstrate that the sediment consists of more water than solids and has very low strength. The problems with laboratory-scale desiccation indicate that field-scale desiccation would not be successful. Even if extremely long drying periods were allowed, the lack of

land area at the KPC site would place further constraints on any management strategies involving drying. Partial drying prior to removal and disposal would have similar constraints.

10.2.2 Physical Features of Ward Cove

The physical features of Ward Cove that are most relevant to sediment remediation are bathymetry and slope, wind and waves, and the presence of sunken logs.

10.2.2.1 Bathymetry and Slope

Detailed bathymetric and geophysical surveys were conducted in Ward Cove in May 1997. The shoreline is mostly rocky, with steep slopes (Figure 10-5). Water depth typically ranges from 10 ft at the head of the Cove to 200 ft at the mouth. Slope and water depth are critical considerations for evaluating the technical feasibility of technologies and disposal options.

Slope Considerations—Portions of Ward Cove have steep slopes (e.g., greater than 25 percent, or 4H:1V [4 horizontal units for every 1 vertical unit]), especially along the shoreline (Figure 3-1). The portion of the AOC where the bottom slope is greater than 25 percent is shown in Figure 10-5. As discussed in Section 9.1, minimal problem sediment is expected to be found on the slopes steeper than 25 percent based on the difficulty in finding sufficient soft sediment on these slopes to collect samples.

Capping is not expected to be successful for those areas where problem sediment could exist on slopes steeper than 4H:1V. During the process of placing capping material on the bottom surface, the capping material gains momentum as it falls through the water and is inclined to flow downslope when it reaches the bottom. Even if the capping material is released near the sediment bed, the problem sediment will still slough because it has no cohesive strength. Using a method that results in a slow, gentle deposition to place a thin cap (e.g., 6–12 in. of capping material) would also displace the problem sediment, but to a lesser degree. The maximum slopes on which sand caps can be placed are discussed in more detail in Appendix K. Further evaluation of capping the steep slopes may be conducted during the remedial design and may include a conventional bearing capacity and slope stability analysis. The steepness of the slopes also presents technical difficulties for constructing disposal facilities in those areas.

Water Depth Considerations—Overall, dredging is not considered to be a practicable technology for water depths greater than 100 ft for a number of reasons. The largest deep draft ships require navigation dredging to depths of approximately 50–80 ft; therefore, deeper dredging is rarely done and equipment is not readily available. At depths greater than 100 ft, dredging would be difficult because of the problems of controlling the location of the dredge head or bucket, even with sophisticated positioning

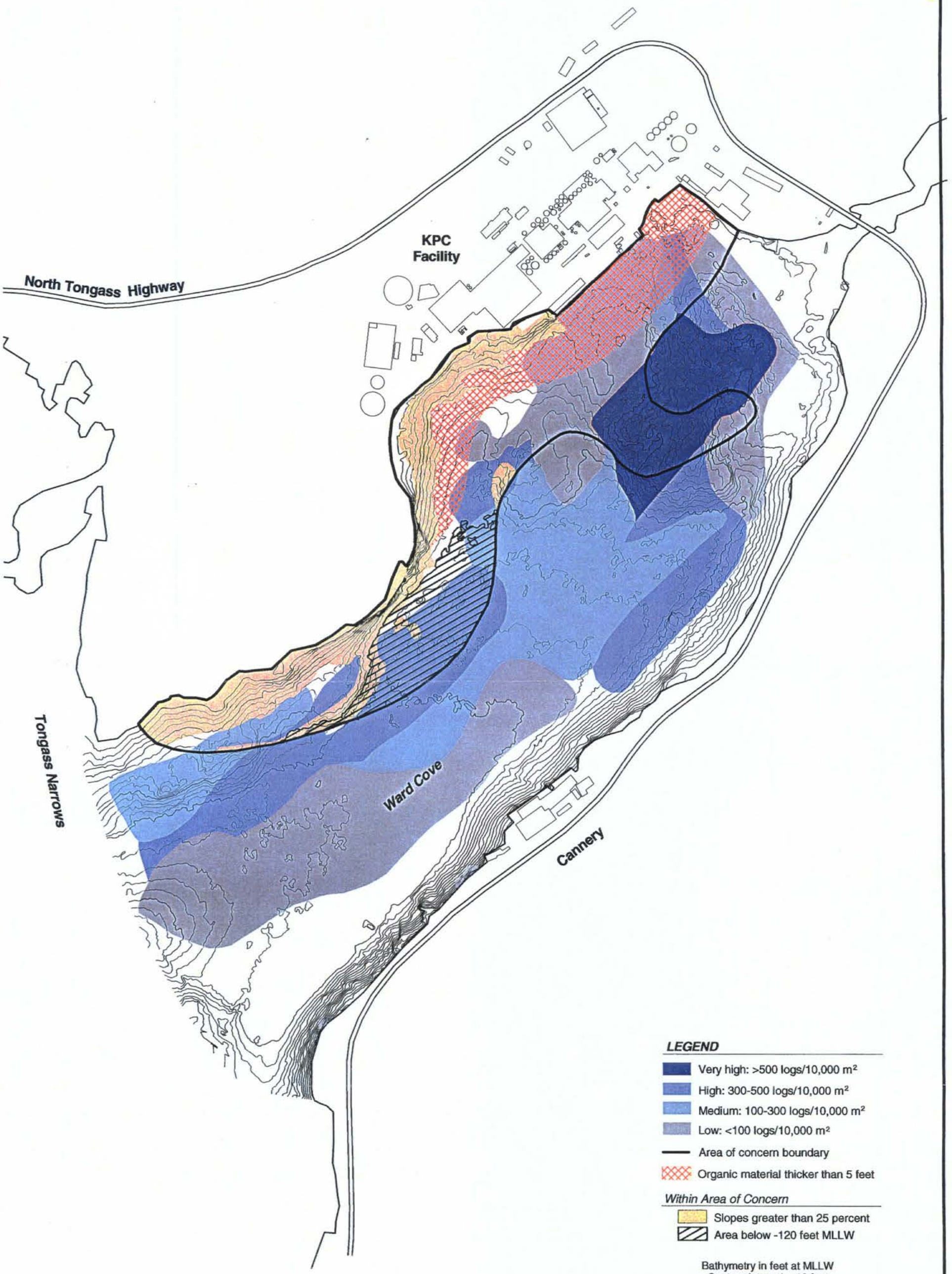


Figure 10-5. Major physical features of Ward Cove.

8600B0W.001 1601 | May 11,1999 | Figure 10-5 Log Distribution | Figure 10-5 | d:\images\5_5_99.apr

equipment, because it is affected by tides, currents, and the variable bed elevations. Dredging costs would also be extremely high because the production rate for lifting sediment over 100 ft would be very low. As a consequence, one of the screening criteria identified is to exclude dredging in depths greater than 100 ft.

Successful *in situ* capping of contaminated sediments is generally limited to waters shallower than –60 ft MLLW (Sumeri 1996). Previous capping projects that achieved success had a solid engineering basis for the cap design (i.e., specifications for the thickness and coverage of the cap were defined at the onset of the capping project) and had appropriate monitoring and management programs in place to ensure that the specifications for the thickness and coverage of the cap were met (U.S. EPA 1999a). EPA's *Assessment and Remediation of Contaminated Sediments (ARCS) Guidance for In-Situ Subaqueous Capping of Contaminated Sediments* (U.S. EPA 1998b) also indicates a maximum water depth of about –70 ft (presented as below sea level) for nine *in situ* capping projects presented in that document. The water depths for the capping projects range from 3 to approximately 70 ft.

Past experience with other projects also indicates that substantially more clean material is needed to ensure coverage in waters greater than about –60 ft MLLW. To illustrate the occurrence of greater spreading of capping material with increasing water depth, the U.S. Army Corps of Engineers (Corps) (Seattle District), in support of U.S. EPA (1999a), has performed computer simulations using its STFATE model. Modeling that incorporated a bottom dump barge with medium sandy silts showed that a single deposition in 60 ft of water effectively covered a 240-ft radius circle with a 6-in. cap, while the same deposition in 140 ft of water effectively covered only a 160-ft radius circle with a 6-in. cap, because the dumped material spread with depth (U.S. EPA 1999a).

The maximum depth for *in situ* capping at Ward Cove will be determined during remedial design. For the purpose of this report, a depth of –120 ft MLLW is used as an estimate for the maximum water depth limit for *in situ* thin capping in Ward Cove. This estimate is necessary to select and refine options for remediation in Section 11. The portion of the AOC where the water depth is below –120 ft MLLW is shown in Figure 10-5.

10.2.2.2 Wind and Waves

Wind and wave analyses were performed to evaluate the wave regime in the northern end of Ward Cove. These analyses are necessary to evaluate the size and type of armor protection needed for the confined disposal technologies.

Wind data obtained from the KPC meteorological station at the site include the following:

- 1995 quarterly wind roses, wind speed percent frequency of occurrence by direction
- 1990–1996, yearly wind frequency distribution, frequency by number, frequency in percent of total observations.

Interpretation of this 7-year record yielded maximum sustained wind speeds, by direction, for the site. Direction radials in 22.5-degree intervals centered on the north end of Ward Cove were drawn on a bathymetric contour plot of the area to determine fetch lengths and average water depths for incident wind and wave directions. Based on the geometry and orientation of Ward Cove and the available wind speed data for the site, it was determined that wind directions from the south-southeast through southwest would generate the maximum wave heights at the north end of the Cove.

Waves are generated as winds move across the surface of Ward Cove and the Tongass Narrows from the southwest. With time and distance, the waves gain energy and increase in height. The Wind Adjustment and Wave Growth module of the Corps Automated Coastal Engineering System (ACES) software was used to approximate the open water wave heights, as they approach the project area in the north end of Ward Cove. The wind speed, fetch length, and average water depth data used for wind directions of interest are summarized in Table 10-1, as are the corresponding wave height results.

The model results indicate that the north end of Ward Cove should not experience wave heights greater that 1.0 ft with the highest wind speeds expected at the site. These results are based on the maximum sustained wind speeds reported in the 1990–1997 record. The highest waves expected under these conditions, 0.93 ft with a 1.94-second period, occur when winds are blowing from the southwest (225 degrees) at 22.4 miles per hour (mph). From this direction, winds have the longest fetch (11,400 ft) over which to generate waves.

In the absence of historical wind data previous to 1990, two hypothetical cases, for wind speeds of 40 and 60 mph from the southwest (225 degrees), were also evaluated in the analysis to account for possible maximum wind speeds having impact on the site. For sustained wind speeds of 40 and 60 mph, the maximum wave heights in the north end of the Cove were estimated to be 1.64 ft with a 2.43-second period and 2.72 ft with a 2.85-second period, respectively.

No appreciable transformation (growth) of the estimated open water design wave heights is expected as they travel from deeper water at the south end of the Cove to the north end of the Cove where the depths average 30 ft. This was confirmed using the Linear Wave Theory module of ACES.

Vessel wakes should also be considered as part of the wave environment in the project area. No vessel wake analysis has been performed in this study; however, assuming

TABLE 10-1. RESULTS OF WIND AND WAVE ANALYSIS

Direction	Degrees North	Maximum Wind Speed (mph)	Fetch Length (ft)	Average Depth over Fetch (ft)	Depth at North End of Ward Cove (ft)	Wave Height at North End of Ward Cove	
						Wave Height (ft)	Wave Period (sec)
Wind Speed Record[a]							
SSE	157.5	34.0	1,200	45	30	0.48	1.09
S	180	33.6	2,100	60	30	0.62	1.3
SSW	202.5	33.6	3,800	100	30	0.78	1.66
SW	225	22.4	11,400	200	30	0.93	1.94
Hypothetical Cases							
SW	225	40.0	11,400	200	30	1.64	2.43
SW	225	60.0	11,400	200	30	2.72	2.85

[a] Based on 7 years.

tugboat, barge, and cruise vessel operations at reduced speed for berthing in Ward Cove, vessel wakes on the order of 2–3 ft can be expected.

The purpose of the wind/wave analysis was to determine whether any special shore protection measures would be required for an NCDF constructed in the Cove. The results of the analysis indicate that no special design features or siting requirements are required for NCDFs to address the wind and wave conditions at the Cove.

10.2.2.3 Sunken Logs

Ward Cove has varying densities of logs and wood debris over much of the bottom surface (Figure 10-5). The presence of logs complicates several aspects of sediment remediation.

A thick cap for isolation purposes cannot be successfully constructed (i.e., a uniform cap approximately 3-ft-thick) where log densities are classified as medium or higher, because capping material would fall into holes around the logs and leave thin or bare areas. In medium- or high-density log areas, removal of the logs would need to precede thick capping. Thin capping (amending surface sediments) could be implemented in low- to high-density log areas because thin capping requirements allow for varying degrees of capping and physical mixing with surface sediments. Thin capping could not be conducted in very high-density log areas because it would have limited beneficial effect (i.e., little or no capping material would reach and amend the surface sediments). No information is available concerning thickness of the log piles.

10.2.2.4 Climate

The Ketchikan area has a maritime climate, characterized by relatively mild, wet conditions. The average minimum/maximum January and July temperatures are 29/39°F and 51/65°F, respectively. NCDFs would not require any special protection to address ice formation in the Cove because of the relatively mild climate.

10.2.3 Summary of Site-Specific Screening Criteria

The technologies and process options will be screened using the primary criteria of effectiveness, implementability, and capital cost. In addition, the following criteria were developed for capping, dredging, and disposing dredged material at the Ward Cove site. These criteria are based on current dredging and dredged material disposal practices, the nature of problem sediments, and the site-specific characteristics of Ward Cove.

- Dredging is not considered a feasible technology for water depths greater than 100 ft.
- Thick capping is not feasible where log densities are classified as medium or higher.
- Thin capping would have a maximum water depth limit of −120 ft MLLW (for the purpose of this report). Thin capping could be examined further with a pilot study during remedial design.
- Capping may not be successful where the existing slope is steeper than 4H:1V. Further evaluation of capping the steep slopes may be conducted during the remedial design.
- Berm construction for NCDF and CAD sites is not considered to be feasible where the existing slope is greater than 8H:1V (i.e., 8 horizontal units for every 1 vertical unit).
- CAD is not a feasible technology for water depths greater than 100 ft.
- Deep water CAD is not a feasible technology for areas where the existing slope is steeper than 100H:3V (Ecology 1990).
- Confined disposal sites, NCDF and CAD, with capacities less than 10,000 cy are not considered practicable because of the high unit costs of berm construction.
- NCDF capacity is based upon problem sediment fill to a maximum elevation of +7 ft relative to MLLW (mean lower low tide water level, or approximate ground water elevation). Filling above elevation +2 ft MLLW would require subsequent handling of sediment, because it would be higher than could be placed with bottom dump barges at highest tide.
- CAD capacity is based upon fill to a maximum elevation of −30 ft MLLW in navigation areas to avoid interference with vessel traffic (i.e., for all but large general cargo vessels). If deeper draft vessels are anticipated to be used at the site, the CAD site maximum elevation must be lowered.

10.3 EVALUATION AND SCREENING OF TECHNOLOGIES AND PROCESS OPTIONS

In this section, potential technologies and process options are described, evaluated, and retained or eliminated from further consideration for the Ward Cove site. Table 10-2 lists the confinement technologies that are potentially applicable to this project.

TABLE 10-2. SUMMARY DESCRIPTIONS OF CONTAINMENT TECHNOLOGIES

Technology	Process Option	Construction Methods	Capacities/ Elevations
In-Place Containment	Capping (thick cap)	Placement of 3-ft-thick clean imported sand cap. Several placement methods available.	NA[a]
	Enhanced recovery (thin cap)	Placement of thin cap (e.g., 0.5- to 1-ft-thick) or isolated mounds (approximately 3- to 6-ft-thick) of clean imported sand. Several placement methods available.	NA[a]
Containment Facility	Sawmill dock NCDF at Site 1	Clamshell dredge and clamshell placement for berm.	155,000 cy problem sediment; elevation –20 to +7 ft MLLW
		Clamshell dredge and bottom-dump barge for problem sediment to +2 MLLW. Subsequent handling of problem sediment to +7 MLLW.	Clean imported sand from +7 to +18 ft MLLW
		Hydraulic pump-out of imported sand for cover.	
	Ward Creek NCDF at Site 2	Same as Sawmill Site 1	175,000 cy problem sediments; elevation –35 to +7 ft MLLW
			Clean imported sand from +7 to +18 ft MLLW
	Rail barge NCDF at Site 5	Not feasible because of steep slopes and deep water.	Not feasible because of steep slopes and deep water.
	CAD at Site 2	Clamshell dredge and clamshell placement for berm	80,000 cy problem sediments;
		Clamshell dredge and surface release bottom dump barge for sediments and cap	below 0 ft MLLW finish elevation
		Cap with 3-ft-thick sand	
	KPC dock CAD at Site 3	Clamshell dredge and clamshell placement for berm	156,000 cy problem sediments;
		Clamshell dredge and surface release bottom dump barge for sediments and cap	below –30 ft MLLW finish elevation
		Cap with 2-ft-thick sand and 1 ft gravel armor	
	Deep CAD at Site 4	Clamshell dredge and tremie placement of sediments	Contaminated sediments placed at –130 to –150 ft MLLW
		Hydraulic placement of 3-ft-thick sand cap	
	Upland disposal	Clamshell dredge; truck or barge	Less than 25,000 cy problem sediments at KPC flyash landfill or Washington landfill

Footnotes on following page.

TABLE 10-2. (cont.)

Note: CAD - confined aquatic disposal
cy - cubic yard
KPC - Ketchikan Pulp Company
MLLW - mean lower low water
NCDF - near-shore confined disposal facility

[a] Area not required for technology screening purposes.

10.3.1 Dredging Technologies

The loss of chemicals to the surrounding waters is of particular concern when dredging problem sediments. Because the CoCs in Ward Cove are generally bound to fine particles, which are easily resuspended, the focus of technology development efforts is to minimize the amount of resuspension through equipment design and/or operational controls (Palermo and Hayes 1992).

10.3.1.1 Hydraulic Dredging

Problem sediment in Ward Cove is a soft, organic material, with high water content. The hydraulic dredging process adds significant volumes of water to the sediment, aggravating an already substantial handling problem. Slurry water would typically make up 90 percent of the *in situ* sediment and water discharged through the pipeline. The discharge of soft, organic fine-grained sediment with up to 90 percent additional water entrained in the sediment would result in a sediment without any structural strength, making the discharged sediment impossible to cap. In a CAD application, either pumping the material directly to a site with release at mid-depth or higher in the water column, or pumping into a barge to haul to the site for release at the surface, the ratio of sediment to water would likely be so low that the slurry solids would dissipate as suspended matter in the water column before reaching the bottom surface during dumping or pump discharge. Any release of slurried sediment must be near the bed; however, the condition of excessive water in the sediment will still exist and make capping of the sediment virtually impossible.

Ward Cove has varying densities of logs and wood debris over much of the bottom surface. Without the removal of logs, a hydraulic cutterhead dredge cannot be used because of the difficulty in maneuvering the dredge and cutting through the logs. Cutterhead dredges operate on a system of anchor wires and/or anchor spuds. The logs would make it very difficult, if not impossible, to set the spuds or anchors to swing the cutterhead and move the dredge forward or to move the cutterhead through a cut. The cutterhead action would be extremely difficult or impossible to control and could not grind up the excessive log and wood debris. A hydraulic suction head dredge would not be able to dredge around, through, and under the buried logs.

The loose nature of the affected sediment, the sloping bottom of the Cove and other uneven sediment surface features, and the wood debris would prevent a hydraulic dredge from removing all of the organic-rich sediment. Hydraulic dredges that are designed to remove large volumes of sediment (i.e., thousands of cubic yards) in deep water (i.e., greater than 40 ft) have relatively rigid suction pipes and swing in an arc at a fixed elevation to remove the sediment. The movement of the suction end of the dredge through the soft sediment in Ward Cove would displace some of the soft sediment, which would then flow back into the depression made by the dredge. In addition, sediment immediately adjacent to the depression would also flow into the depression. Multiple passes could be made, but the soft sediment would continue to be displaced and flow back into the depression. Also, the sloping bottom, other uneven sediment surface features, and the

wood debris would prevent adequate positioning of the suction head to effectively remove all of the soft sediment. After extensive dredging is conducted, a residual layer of the soft, organic-rich sediment would remain in the dredged area on the bottom of the Cove.

As discussed in Section 10.1, the EDDY Pump is a variation of hydraulic dredging that uses a vortex pump. A demonstration project using the EDDY Pump was performed in 1994 in the Cresta Reservoir in Plumas County, California. A 0.25-m diameter discharge was used with a 224-kW electric motor. Fine to medium sand was dredged to a depth of 15 m. Short-term maximum production was 205–271 m^2 per hour, for periods of 2.3–6.5 hours per day (23–65 percent effective time). Slurry densities of 70 percent solids were sustained during the demonstration.

The EDDY Pump and other similar specialty designs, such as the Pnuema pump, appear to produce less resuspended sediments at the dredge site and can produce higher slurry densities, for very short time periods. However, they still suffer the same problems as hydraulic dredges (i.e., introducing additional water and inability to dredge through logs). They cannot move large sunken logs, and the smaller pieces of logs and branches would plug the smaller intake of these specialty dredges, likely damaging the pump rotor. In addition, any type of hydraulic pump will add water to the sediment, which has to be separated and treated at the discharge site.

In summary, hydraulic dredging is not considered an acceptable method of dredging for Ward Cove based on the amount of logs and wood debris and the handling problems that would result from the addition of water to the organic-rich sediment. Based on the results of the engineering analyses, the sediments cannot be dewatered using conventional gravity settling methods that are essential for cost-effective hydraulic dredging. An examination of the sediments at Soil Technology, Inc., also indicated that air-drying the sediments would not be successful. Even if extremely long drying periods were allowed, the lack of land area at the KPC site would place further constraints on any sediment remediation alternatives involving drying. The use of former KPC pulp mill facilities to conduct wastewater treatment of water from a dewatering or drying process was considered but rejected, given that there is no longer a wastewater treatment facility at the site. Hydraulic dredging is not retained for further consideration.

10.3.1.2 Mechanical Dredging

Sediment loss is an important consideration for mechanical dredging. Sources of sediment loss during clamshell dredging can be attributed to the following:

- Sediment resuspension occurs when the bucket impacts the bottom, when the bucket is closed and removed from the bottom, and when the bucket is dragged across the bottom of a completed cut to smooth out irregular surfaces.

- Sediment loss from the bucket occurs as it is retrieved through the water column and through rapid drainage of the entrapped water and slumping of material when the bucket clears the water surface.
- Sediment spills occur from accidental overflowing of the disposal barge, from leakage from the barge, and from the bucket remaining partially open from caught logs during retrieval.

Accidental overflow from the disposal barge, leakage from the barge, and sediment resuspension due to loss from the bucket during retrieval are conditions that could be corrected by operational controls of the dredging. As an example, the contractor could be required to avoid full barge loading, which would reduce or eliminate any overflow. If split hull barges are used, the contractor could be required to replace seals before dredging and maintain them during the project, and thereby prevent leakage from those barges during loading and transport. Use of an enclosed bucket for the dredging would reduce or eliminate loss of sediment from the bucket during retrieval, and slower production rates could be used to further reduce loss of sediment, as necessary.

The use of a backhoe mechanical dredge in Ward Cove would be limited by the depth at which it can operate. Until the early 1990s, the deepest a backhoe could operate was 20–30 ft. Recent construction of extremely large backhoe dredges has allowed removal at depths to 45–50 ft. Currently, there are only two dredges of this type operating in the United States (Juhnke 1997, pers. comm.).

A clamshell dredge would be expected to remove some material from the areas with low log densities. This dredge operation would tend to result in high levels of suspended sediment, because the clamshell bucket would resuspend sediment as it hits logs at the sediment surface. Resuspension also is increased when the bucket is retrieved through the water column in a partially open position with log debris in the bucket mouth.

Similar to hydraulic dredges, mechanical dredges would not be able to remove all of the organic-rich sediment because of the loose nature of the affected sediment, the sloping bottom of the Cove, and the relative accuracy of the dredging equipment. A clamshell bucket would be lowered into deep water and into the soft sediment. When the bucket is placed on the bottom, it would displace some of the loose, soft sediment, which would then flow back into the depression made by the dredge. In addition, sediment immediately adjacent to the depression would also flow into the depression. Multiple passes could be made, but the soft sediment would continue to be displaced to some extent and flow back into the depression. Also, the sloping bottom and other uneven sediment surface features would prevent adequate positioning of the bucket in all locations to remove all of the soft sediment. After dredging is conducted, a residual layer of the soft, organic-rich sediment would remain in the dredged area on the bottom of the Cove.

A significant advantage of the mechanical dredge over hydraulic dredging is that less water is mixed with the sediment, thus minimizing bulking and dewatering requirements. Although there are some disadvantages with mechanical dredging, it is technically

feasible for this project and is retained as a representative removal technology for the development of alternatives.

10.3.2 In-Place Capping

The success of capping is highly dependent on the structural strength of the sediment being capped. If the sediments have sufficient strength, then the cap can be supported by the problem sediment to provide a complete cover over the sediments (i.e., a thick cap of 1–3 ft overlying the sediment). If the sediments do not have sufficient strength to support a complete cap cover, they may be amended or mixed with capping material or subjected to sediment mounding techniques (i.e., thin capping). Both thick and thin capping were evaluated for Ward Cove problem sediments. A variety of placement techniques are potentially applicable to the AOC (see summary in Section 10.1 and Appendix K).

10.3.2.1 Thick Capping

The long-term integrity of a thick cap will depend upon the ability to cover the area with a consistent cap thickness. Capping of *in situ* sediment over large areas has been successfully accomplished for other projects with a higher bed sediment density than that of Ward Cove. The thickness of very soft organic material, the presence of partially buried logs, and the steep slope in Ward Cove make thick capping not technically feasible for most of the AOC. Placement of a thick layer of sand or other capping material on the soft organic-rich sediment in Ward Cove would be expected to displace the soft sediment laterally from the placement area, rendering capping for confinement as ineffective. Capping of sediment for confinement in the presence of buried logs or on steep slopes is also expected to be ineffective (see discussion in Section 10.2). Because it would not be technically feasible to place a thick cap over Ward Cove problem sediment, thick capping is eliminated from further consideration.

10.3.2.2 Thin Capping (Enhanced Recovery)

Thin capping, also known as enhanced recovery, differs from other confinement technologies in the degree of spatial isolation and coverage of the target sediments. The goal for thin capping in Ward Cove is to amend surface sediments (i.e., the biologically active zone) through partial surface cover or dilution. Existing surface conditions within the AOC vary spatially, which would result in variable areas of capping or mounding and amendment (mixing) within the targeted area upon placement of cap material.

Thin capping would be accomplished by distributing a thin layer (e.g., 6–12 in.) of cap material via a placement method that results in a slow, gentle deposition on the sediments, such as using a diffuser or washing the material off a barge. Slow, gentle deposition on the sediment surface is necessary because of the high water content, high organic content, and low strength of problem sediments. Ideally, the cap material would mix in with the surface sediments in the biologically active zone, improve habitat quality, and

eliminate sediment toxicity. Some displacement of bottom sediment would likely occur. A thin cap would amend the surface sediments and provide enhanced recovery. However, if the sediments lack structural strength, the capping material will displace the sediments or will sink into the sediment and below the active surface layer.

In some portions of the Cove, problem sediments are expected to have minimal structural strength and will not likely support sufficient volumes of capping material in the surficial, biologically active zone to effectively amend problem sediment. This condition is most likely in areas where core logs show the presence of a thick layer of soft, organic-rich sediment. If there is low sediment strength in these areas, it is not likely that application of 6–12 in. of capping material would provide any significant amendment to the sediment surface. Rather, capping material might be expected to sink through the unconsolidated surface sediment. A pilot study would need to be performed during the design phase to determine if thin capping would effectively amend the surface sediments and to evaluate the thickness of organic-rich sediments that could support the cap. The water depth at which thin capping could be conducted could also be examined during the pilot study, as well as other design issues such as placement method.

In the event that placing a thin layer of cap material is unsuccessful, the cap material could be placed as a series of mounds that extend out of the organic sediments. Spot dumping of capping material could be conducted to create a discontinuous, island-like cover. The sand (or other capping material) could be placed carefully, one clamshell at a time, to create isolated, mounded islands of sand. Alternatively, the sand can be placed by hydraulic washing to create isolated, linear low ridges (Figure 10-6). The cap material will displace the organic material laterally when it is placed on the bottom. This type of capping is most appropriate for portions of the AOC where the layer of organic-rich sediment is less than 5 ft. It allows the possibility of a 5.5–6 ft mound that displaces the organic sediment and would result in a small island of clean material above the organic-rich sediment (Figure 10-6). Mounding of cap material in areas where the organic-rich sediment is greater than 5 ft thick would be very costly because of the amount of cap material needed to create emergent mounds. If spot dumping of capping material were to be implemented, then placement methods, placement locations within the AOC, and the number of mounds to be created would be evaluated further during remedial design. Also, the initial construction of the mounds would need to be monitored to ensure proper implementation.

Thin capping is not expected to result in a continuous, even blanket of material over the target area. Once thin capping has been completed, the bottom surface may be expected to look like a series of islands covered with sand with areas of sand mixed with sediment between the islands. Those areas with the structural properties to support the thin cap (thinner layers of soft sediment or sediment containing larger pieces of wood debris) are expected to form the nucleus of the islands, while areas of deeper or softer sediment would make up the surrounding areas, where mixing with surface sediment could occur.

There may be the potential for uncapped organic-rich material to become resuspended, transported, and deposited onto an area that has been capped. The potential for

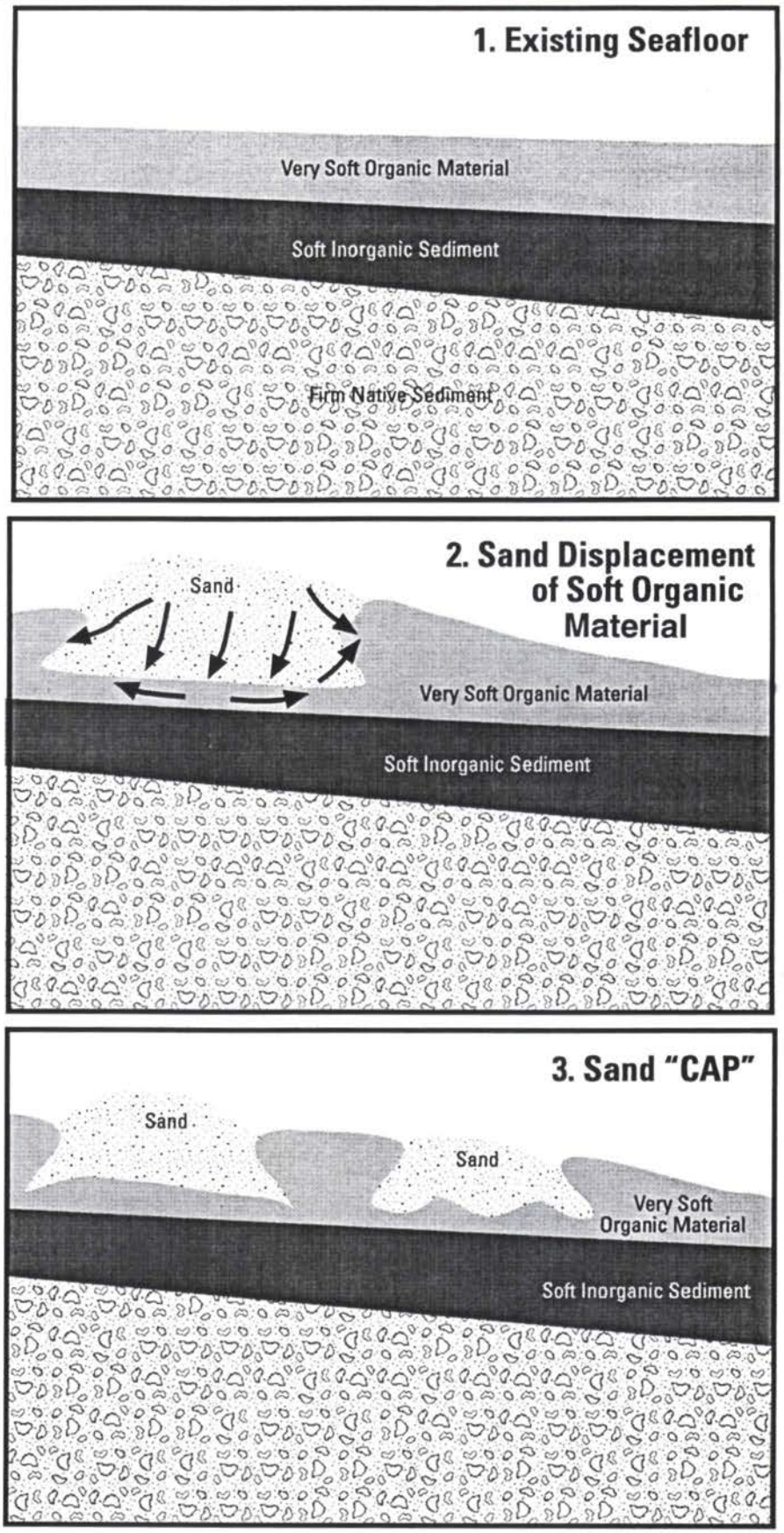


Figure 10-6. Placement of sand on soft sediments.

remobilization and transport of sediment was evaluated in Section 5.1 based on current velocities and sediment grain size data. The evaluation indicated that the potential for sediment resuspension is very low. Therefore, there is no reason to believe that the surrounding organic-rich sediment would resuspend and cover a capped area. However, resuspension of organic-rich material onto a capped area would be considered further during the remedial design and remedial action.

Thin capping is retained as a technology for development of the remedial action alternatives. However, a pilot study would need to be conducted during remedial design to determine the appropriate approach for implementing thin capping and the expected outcome.

10.3.3 Containment Facilities

Confined disposal sites fall into three general categories, depending on their general location: upland, CAD, and near-shore disposal. Geotextile bag containment is also evaluated.

10.3.3.1 Upland Disposal

This section evaluates upland disposal technologies for dredged sediment. The technologies include disposal of the dredged sediments in an upland solid waste landfill and land application of dredged sediments over open land.

There is limited suitable upland area in the Ketchikan vicinity for the disposal of sediment. The land in the vicinity of the Ketchikan area is steep and forested. Of the more than 800,000 acres of land and water within the boundaries of the Ketchikan Gateway Borough, the Tongass National Forest accounts for 95 percent. The mountainous character of most of the terrain limits community expansion and has effectively restricted settlement in Ketchikan to the narrow strip of land about 30 miles long that borders Tongass Narrows. There is no place where residential or commercial development extends as much as 1 mile inland from the coast because of the features of local topography. Because of the high costs of services associated with dispersed commercial and industrial uses and the scarcity of appropriately sloped and located land for those uses, Ketchikan's future land use development will probably continue to be concentrated around the existing city (Martinson and Kuklok 1977).

The surrounding steep terrain also limits potential upland disposal because of landslides and the shallow soils. Landslides are relatively common in the Ketchikan area for several reasons. Glacially polished bedrock may be very close to the surface with only a thin covering of unconsolidated material over it. During periods of heavy precipitation, the probability of landslides on these slopes increases because water adds weight and acts as a lubricant. Receiving approximately 151 in. of precipitation annually, Ketchikan is one of the wettest locations in the United States. In addition, landslide potential increases manyfold when the protective vegetation and organic soil layers are removed because the

holding power of plant roots is destroyed and water runoff and erosion increase (Martinson and Kuklok 1977).

KPC operates a landfill at the mill site. Upland disposal of dredged sediments in a landfill could occur at the KPC landfill or at an existing location in the continental United States. Each of these disposal options is evaluated in more detail later in this section. At this time, with the exception of the KPC landfill, no landfills have been identified in southeastern Alaska with a capacity for thousands of cubic yards of sediment. The Ketchikan area does not have a local municipal solid waste landfill for household waste. The household waste in the Ketchikan area is currently being shipped for disposal in eastern Oregon. Ketchikan does have a landfill for construction debris, but it does not accept hazardous materials and the city council has given direction to close the landfill. If formally requested by KPC to accept sediment at the landfill, the City has indicated that they would deny the request (Voetberg 1998, pers. comm.).

In addition to the limited suitable land in the Ketchikan area, the results of the column settling tests, consolidation tests, and desiccation observations for the Cove sediments indicate that upland disposal would be difficult. The sediments could not be dried using conventional, cost-effective methods. Gravity settling basins and air-drying would not separate the water from the organic-rich sediment. The sediments would form a slurry that would have to be contained inside a pond or bermed area, and the sediment would be too soft to support even low-ground-pressure earthmoving equipment. Even though upland disposal would be difficult, it has been considered further for small volumes of sediment.

KPC Ash Landfill—The KPC landfill is currently permitted (ADEC Solid Waste Permit No. 9713-BA0001) to receive approximately 600 yd^3 of solid waste per month including dredge spoils. In recent months, the active landfill cell has been used primarily for disposal of dredge spoils from maintenance dredging activities and soil/sediment from the remediation of the access road ditch. Even though the landfill permit allows dredge spoils to be placed in the landfill, KPC would seek approval from the state prior to disposing any sediments in the landfill that are dredged as part of the sediment remediation project.

The wet, soft organic material would be off-loaded from barges into trucks at the dock and then transported by truck to the landfill. The haul trucks should have waterproof liners to prevent loss of water and sediment from the trucks along the haul route. At the landfill, the wet sediment would be dumped into designated areas of the landfill. The landfill is constructed with high berms, a low permeability geomembrane liner, and a leachate collection and treatment system. These characteristics may allow sufficient dewatering at the landfill to occur.

Availability of the landfill for dredged material disposal is dependent upon future operating scenarios for the KPC site and on state acceptance. Continued operation of the existing power boilers is uncertain. However, potential wood processing operations

could necessitate operation of a modified or new power boiler. Disposal of flyash generated by the new power boiler may require the existing capacity of the landfill.

KPC is currently evaluating future operating scenarios for the Ward Cove facility. Until firm decisions are made regarding future operations that may or may not eliminate this site, this option is retained for further consideration for small volumes of sediment.

Offsite Landfill—Another option for upland disposal would be placement in an approved solid waste landfill. In this process option, the sediment would be transported by barge to an off-loading site near a landfill with the capacity to accept the sediment. The total disposal cost would be very high because of the cost of shipping by barge hundreds of miles, transporting by truck, and incurring landfill disposal fees. This option is retained for further consideration for small volumes of sediment. Potential sites are located near Roosevelt, Washington, and Arlington, Oregon.

Land Application—Agricultural lands in arid regions are generally the most appropriate areas to accommodate large volumes of sediment with high water content. There are no suitable arid agriculture areas in the state of Alaska. The closest areas would be eastern Washington and eastern Oregon, along the Columbia River. Sediment could be barged up the Columbia River and off-loaded. Large pieces of wood debris in the sediment would need to be removed and properly disposed of prior to applying the sediment to land. While the sediments would not need to be confined, they would need to have the salt (from seawater) removed. The removal of salt would require special washing of the sediment or long-term conditioning of the sediment in the open environment. Typical conditioning of saltwater sediments requires placing the sediment in thin layers over large areas so that rainwater can leach out the salts. Silty sand sediments require from several months up to 1–2 years for leaching. The fine-grained nature of the organic-rich sediment does not lend itself to washout of salts within reasonable periods of time. Application of the sediments to land is not retained for further evaluation because of the long transport distances to suitable land and associated high costs, the problems and costs associated with removal of wood debris, and the difficulty with leaching out the salt in the sediment.

10.3.3.2 Confined Aquatic Disposal

CAD sites have been used successfully to contain problem sediment at many sites and, like NCDF, are one of the most commonly used disposal options. Design factors for CAD sites include water depth, bed slopes, water column velocities, bed stability, and physical characteristics of problem sediment. Given the extremely soft, organic, fine-grained nature of Ward Cove sediment, both placement and capping of problem sediment are expected to pose particular challenges. Capping of a slurried sediment discharged to the open bottom is not technically feasible even with adequate berming to control sediment movement and mudwaves during capping operations. Berming issues are discussed in greater detail in Appendix K.

Because the density of the organic-rich problem sediments is less than that of typical sediments, the material would fall through the water column at a slower velocity. The dredged organic material would tend to "float" down at a relatively slow rate because there would be a small difference between the density of the dredge material and salt water. Disposal would need to be restricted to periods of slack tide, on days with low levels of wind and waves. If water quality monitoring during disposal indicates that the dredged organic material is dispersing, then other more elaborate and expensive placement methods may be required.

Even though CAD is not likely to be technically feasible for Ward Cove problem sediments, potential CAD sites are evaluated further in Section 10.4 because CAD has been used successfully to contain problem sediments for other projects. Three sites were considered as potential CAD locations (Figure 10-7). One of the three CAD sites (Site 2) is adjacent to the shoreline and is evaluated concurrently as a potential NCDF site. The final elevation of the CAD at Site 2 would be at or below 0 ft MLLW; the final elevation of the NCDF at Site 2 would be above 0 ft MLLW.

10.3.3.3 Near-Shore Confined Disposal Facilities

NCDF sites have been used successfully to contain problem sediment at many sites and, like CAD, are one of the most commonly used disposal options for problem sediments. The long-term integrity of the sites can be ensured with appropriate design. Design factors include physical characteristics of sediment, such as average grain size and moisture content, groundwater and tidal elevations, foundation materials for dikes, and bed stability of the site. NCDF is retained for further evaluation in this section. Three sites were considered as potential NCDF locations (Figure 10-7). These sites are evaluated in greater detail in Section 10.4.

10.3.3.4 Geotextile Containment

For geotextile bags to be successful, the material used to fill the bags must contain a sufficient proportion of solids and minimal fine materials so that the bag will dewater over time without significant risk of fabric clogging and loss of permeability. The fibers in the Ward Cove organic sediment would tend to stick to the bag and form a layer of material with low permeability, so that the water could not flow out.

A technical note on the state of the art for use of geotextile bags was prepared by the Corps, Waterways Experiment Station (Corps 1996). The technical note states that there has been limited use of geotextile bags for contaminated sediment disposal. Most of the projects involving geotextile bags have been used for "shallow-water, low-energy breakwaters and as dikes." Two projects have been done in water depths less than 30 ft.

The primary purposes for using geotextile bags are to reduce spread of contaminated sediments at disposal sites and to reduce short-term water quality impacts during

disposal. The Corps technical note states that "the Corps has performed nearly 30 capping projects using conventional hopper or barge surface-release techniques. No adverse environmental impacts have been documented, even though some losses to the water column and resuspension have occurred."

Installing the geotextile bags into barges and sewing the bags together when they are full significantly increases cycle time, which increases construction costs. For the Marina Del Rey project in Los Angeles, production was one-half that of normal production for dredging and disposal. The Corps estimated that the increased cost for using geotextile bags was approximately \$65–\$78/m^3 of contaminated material (Corps 1996).

For the Ward Cove site, the sediments have very low strength, which would result in an increase in the lateral sediment pressures pushing the bags apart. In addition, because Ward Cove is deep, the loaded bags would fall a considerable distance. Under these conditions, the bags could rupture upon impact with the bottom.

Geotextile bags can be used in conjunction with hydraulic or mechanical dredging. Hydraulic dredging is not considered an acceptable method of dredging for Ward Cove because of the physical nature of the problem sediment and the excessive debris and logs on the bed as discussed in Section 10.1. In view of the problems associated with the deep water deposition of geotextile bags, the high-cost of geotextile bags, the difficulty in filling the bags with loose, soft sediment, and the limited ability for the bags to dewater because of the fibrous nature of solids, the bags are not considered appropriate for this project and are therefore eliminated from further consideration.

10.3.4 Sediment Treatment

Candidate treatment technologies appropriate for Ward Cove sediments fall into three general categories: dewatering, wood separation, and *in situ* treatment.

10.3.4.1 Dewatering

Dewatering the Ward Cove sediments would be very difficult and expensive because they are fine-grained and have high water and organic content. Tests conducted by the laboratory indicate that the sediments do not readily settle. Dewatering would also likely involve removing wood debris from the dredged sediments. Active dewatering technologies will be considered only as a last resort for small volumes of dredged sediments that may require upland disposal. Some dewatering would also occur during the implementation of other confined disposal options such as NCDF.

10.3.4.2 Wood Separation

Although no treatability tests have been conducted for Ward Cove sediment to determine the ease with which wood could be separated from the inorganic sediment, it appears that

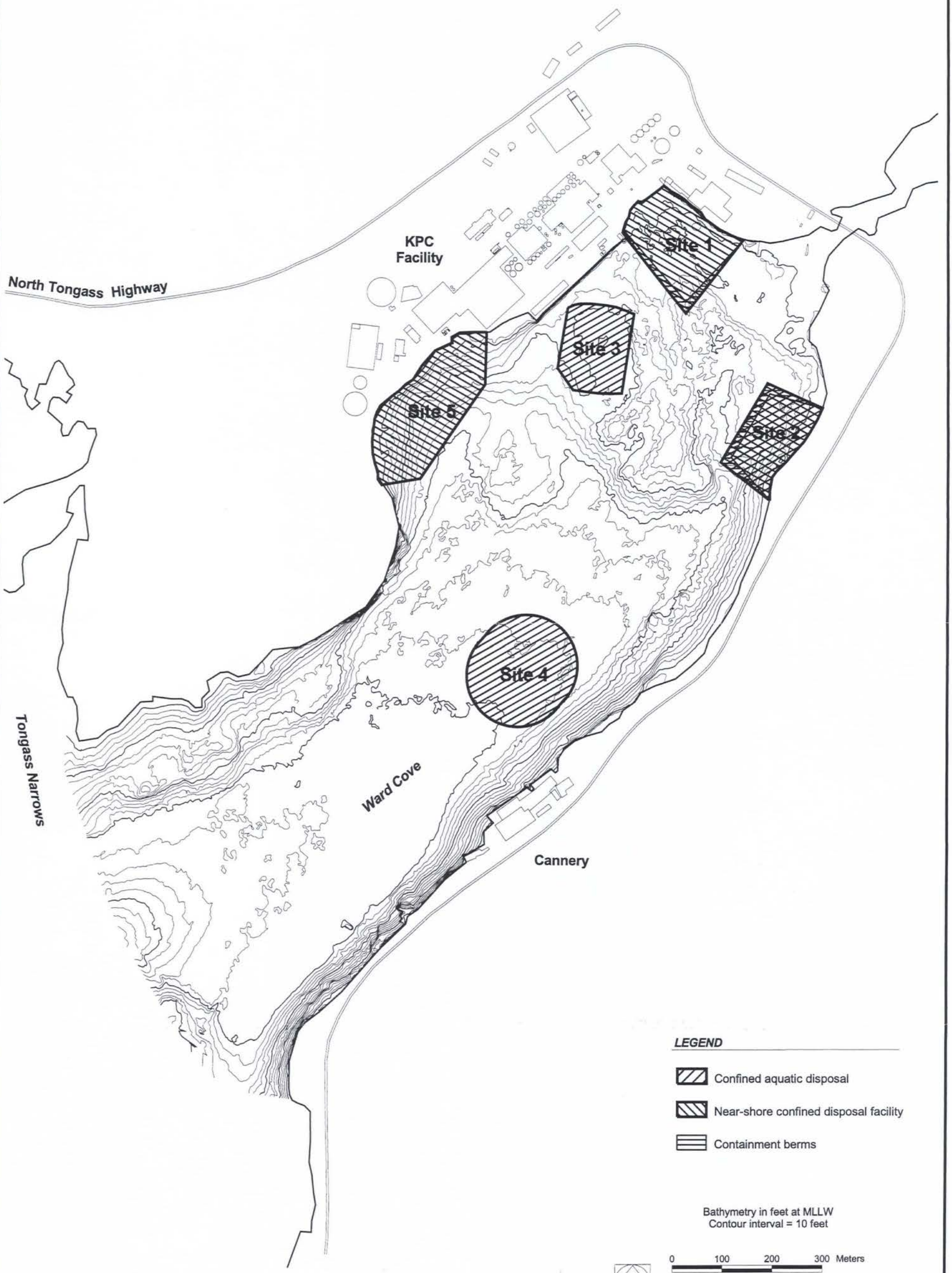


Figure 10-7. Preliminary disposal site locations.

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logs and larger wood pieces (i.e., larger than 3 in.) could be separated (see Section 10.3.5). However, it would be impractical to separate and remove smaller wood pieces and particles from the inorganic sediment because the problem sediments predominantly consist of small organic wood material (gravel-, sand-, and silt-sized wood particles) mixed with small amounts of inorganic material. There are a number of separation techniques available that could be tested, but these techniques are usually used to concentrate a waste material that is present in small quantities in a medium or to remove small quantities of debris to facilitate treatment of the medium. In addition, large amounts of wastewater would be generated during separation and would require treatment. Wood separation is not retained for alternative development because it would be difficult to implement, little or no benefit would be derived, and large amounts of wastewater would be generated and require treatment.

10.3.4.3 *In Situ* Treatment

The Limnofix technology could potentially be used to treat sulfides and similar constituents in the sediments. However, Limnofix is in the developmental stage and there is limited information regarding its potential application to a large site with the sediment characteristics, sediment thicknesses, large amounts of wood debris, and other physical constraints such as those found in Ward Cove. To date, the Limnofix technology has been used only on small pilot-scale projects.

Murphy et al. (1995b) describe bench-scale and pilot-scale work to treat sediments by injecting ferric chloride. This work was done in the St. Mary's River, which flows from Lake Superior to Lake Huron. The St. Mary's River sediments contain wood fibers 20 cm to 2 m thick, and the site is contaminated with PAHs, total petroleum hydrocarbon (TPH), and sulfides. Murphy et al. (1995a) describe bench-scale and pilot-scale work to inject calcium nitrate into sediments in Hamilton Harbor, which is located in the western end of Lake Ontario. The Hamilton site consists of silt underlain by clay and is contaminated with PAHs, TPH, coal tar, and sulfides from historical steel mill operations and other industrial and municipal sources. The oxidants added to the sediments can oxidize sulfides and organic compounds to some extent and thereby potentially reduce sediment toxicity. The pilot-scale work involves the use of a boat to tow an 8-m-wide injection boom through the surface sediments. The chemicals are injected in the upper 5 to 10 cm of surface sediment and then sink deeper into the sediments. The pilot-scale tests have been conducted in water depths of up to 22 m in Hamilton Harbor.

The St. Mary's River sites were 12 by 90 m (0.27 acres) and 36 by 200 m (1.78 acres). The Hamilton site was about 50 by 100 m (1.24 acres). Murphy et al. (1995a) report that in 1988, 3,000 m^3 of highly contaminated sediment was dredged and taken to an upland landfill at a cost of \$200/m^3. They projected that the cost for *in situ* treatment would be on the order of 20 percent of the confined disposal cost (e.g., in an upland landfill). Limnofix costs are expected to be \$20–\$30/m^2 (\$81,000–\$121,000 per acre) with a treatment thickness of about 0.5 m (Babin 1998, pers. comm.).

The Great Lake sites are areas where the sediment is contaminated with high levels of chemicals from a long history of heavy industrial discharges. Sediment (and soil) treatment is primarily effective and beneficial for those sites where there is a relatively small volume of material with high levels of contamination (i.e., principal threats or hot spots). In Ward Cove, however, there is a large area and large volume of sediments with low levels of contamination. In addition, the Ward Cove sediments have high BOD, and a significant percentage of the oxidants applied could be consumed by the natural organic matter, not the CoCs (assuming the treatment technology could be implemented). Even if the CoCs in the surface sediments were oxidized, it is likely that the CoCs would soon recur from degradation of the organic matter in the sediment. Repeated applications of large amounts of oxidants would likely be needed to address the CoCs in the long-term, resulting in high remediation costs and little overall benefit to the Cove ecology.

The Corps has also evaluated the potential application of Limnofix to Ward Cove sediments (Corps 1998). According to the Corps, the projects to which this technology has been applied are not very relevant to the Ward Cove conditions. The Corps has indicated that there could be potential problems with delivery of the nitrate to the Cove sediments due to obstructions such as logs, angled rocky bottoms, and depth of injection. The Corps expressed concerns over the permanence and effectiveness of this treatment at Ward Cove, because there appears to be a large reservoir of organic material in the thick, flocculent sediments, possibly requiring treatment several times. The Corps provided preliminary cost estimates for treating 162,000 m^2 (40 acres) that range from $3.2 million (for a one-time treatment at $20/$m^2$) to $29.6 million (for treating 6 times at $30/$m^2$) (Corps 1998).

Limnofix is not retained for further evaluation because there are too many issues at Ward Cove that would impede full-scale implementation of Limnofix. The logs and wood debris, water depth, thickness of problem sediments, steep slopes, currents, and similar site characteristics would make implementation of Limnofix very difficult or impossible. Even if the technology could be implemented, the CoCs would recur from degradation of the organic matter in the sediments. Remediation costs would be high, and there would be little or no overall improvement to the Cove ecology.

10.3.5 Log Removal

Log removal is not a critical component of the Ward Cove sediment remediation; however, the feasibility of log removal within the AOC and associated costs were estimated to allow for the assessment of potential log removal actions that complement or enhance any proposed dredging effort. EPA has also evaluated the removal of sunken logs at Ward Cove (U.S. EPA 1999b), and a summary of EPA's findings is presented in this section.

Estimates of log distribution in Ward Cove are described in Section 3.1. To provide a preliminary indication of log removal costs that may be associated with dredging, estimated costs were prepared for log removal in the zones with "very high" and "high" log densities within the AOC (Figure 10-5). It is assumed that the average log would weigh

about 4 tons. The area of each of these zones within the AOC and the estimated weight of the logs are shown below:

Zone	Log Concentration (no. per acre)	Area (acres)	Tons of Logs
Very high	over 200	8	about 6,400
High	120 to 200	17	about 11,000
Total			17,400

The estimated costs for removing the logs from the "very high" and "high" zones within the AOC are shown below:

Remove, chip, and use in Ketchikan—$1,900,000
Remove, chip, and use in Puget Sound—$2,400,000
Remove and place in landfill in Washington State—$3,100,000.

The above costs are based on work conducted by Hartman Consulting Corporation for a log removal project in Puget Sound and include construction cost and a 30 percent contingency, but do not include other costs such as those for design, permitting, and monitoring during construction.

The process of removing surface logs may cause high levels of suspended sediment. Use of silt curtains, at a minimum, should be anticipated during this operation. Also, it may be necessary to remove logs only during slack tides if silt curtains are found to be inadequate at higher tide current conditions. Silt curtains are not functional in currents greater than 1–2 ft per second and in depths that require excessive skirt lengths.

It is assumed that the logs are not decomposed or infested with marine borers to the point where they cannot be lifted and handled with standard equipment. For areas to be dredged, whole logs would be removed before the sediments would be dredged. Any remaining sunken logs that are located in the sediments to be dredged would be removed by the clamshell as part of the subsequent dredging.

Chipping costs are included in the above costs, but no other processing costs prior to use are included. It is assumed that the cost for handling, storing, drying, or other preprocessing would equal the salvage value of the chips; therefore, there is no credit for value of the chips. Even though use of the chips was included in the options, it is questionable whether the wood could be reused because of decomposition, infestation with marine borers, and salt content.

EPA evaluated the sunken logs at Ward Cove and concluded that they do not pose a toxic risk to human health, and based on information available to EPA, aged sunken logs do not pose a known or suspected toxic risk to the environment (U.S. EPA 1999b). Specifically, it appears that sunken logs are not toxic to benthic communities in sediments. The sunken whole logs may alter the bottom substrate habitat and may cause a shift in species

composition (i.e., different types of organisms may colonize the altered habitat). EPA also concluded that removing sunken logs is not necessary to meet the RAOs for a sediment cleanup at Ward Cove. RAOs for Ward Cove sediments are presented in Section 8 of this document. Based on its findings, EPA recommends removal of sunken logs in areas of Ward Cove to be dredged, but does not require removal of sunken logs in areas not proposed for dredging.

Based on the evaluation presented in this section and on EPA's evaluation (U.S. EPA 1999b), log removal is retained for the development of remedial action alternatives for those areas of the Cove where dredging is included as a component of the alternative.

10.4 ENGINEERING ANALYSIS OF CANDIDATE DISPOSAL SITES

There are limited sites available for sediment disposal in the vicinity of Ward Cove. Five locations in Ward Cove were subjected to an engineering analysis to determine if they were feasible disposal sites for problem sediment.

10.4.1 Near-Shore Confined Disposal Facilities

Three locations were considered for near-shore confined disposal: Sites 1, 2, and 5 (Figure 10-7).

10.4.1.1 Sawmill Dock Apron Area (Site 1)

An NCDF site with a trapezoidal shape having outer dimensions of approximately 600 by 600 ft could be constructed with select fill berms located in this area. Assuming a mean depth of –20 ft MLLW at this site and disposal of problem sediments to a level of +7 ft MLLW, the capacity could confine approximately 155,000 cy of problem sediment. The site would be capped with a fine-grained sediment (silt, sand, silty sand) with a minimum cover cap of 6–10 ft for a minimum fill to serve the sawmill operations. Engineering analysis of this fill is required for final design to confirm sawmill operations are possible. This site was selected because it is located in shallow water on relatively low slope and avoids construction on submerged log piles. The site, when eventually stabilized, could provide additional space for a number of upland uses.

The berm along the south side of the site would be constructed of imported sand at a 3H:1V slope held with quarry spall training dikes. The berm would be constructed using a clamshell dredge to place the material. The problem sediment would be placed into the NCDF to a final elevation of +7 ft MLLW. The problem sediment would be placed into the NCDF to an elevation of +2 ft MLLW using a bottom dump barge. The remaining 5-ft-depth of problem sediment would be placed after dike closure. The material would be dredged with a clamshell dredge. The NCDF would be capped with imported sand. The cap would be placed with a hydraulic barge pump-out system using a diffuser. When the cap surface level reaches near sea level, the cap will continue to be hydraulically

placed, moving the material away from the discharge pipe with a dozer. The final elevation of the fill would be +18 ft MLLW. The conceptual level cost for this disposal option, including dredging, constructing the berm, and capping the NCDF, is $27 million.

The NCDF at the sawmill dock apron could provide the KPC facility with additional operation storage area. This NCDF is constructed in relatively shallow water and as a result presents the least construction difficulty, but it is also the site with the least capacity.

10.4.1.2 Mouth of Creek Area (Site 2)

An NCDF site with a rectangular shape and having outer dimensions of approximately 400 by 600 ft could be located at the mouth of the creek. Assuming that problem sediments could be filled to an elevation of +7 ft MLLW with a mean depth of 35 ft, approximately 175,000 cy of problem sediments could be placed here. As with Site 1, the cap would be 6–10 ft thick, and the final elevation of the fill would be about +18 ft MLLW. This site was selected because it is located in shallow water on relatively low slope and avoids construction on submerged log piles. The site, when eventually stabilized, could provide additional space for a number of upland uses.

This NCDF site would be constructed and filled in the same manner as the NCDF site at the sawmill dock apron. The conceptual level cost for this disposal option, including dredging, constructing the berm, and capping the NCDF, is $30 million.

The NCDF site at the creek mouth is constructed in deeper water than Site 1 and has a greater capacity. It also provides the potential opportunity for upland site development. Any development would have to consider the long-term settlement of the organic fill. Construction issues and long-term integrity considerations would be similar to those for Site 1.

10.4.1.3 Rail Barge Terminal Area (Site 5)

The slope in the area west of the rail barge terminal is greater than 8H:1V out to a water depth of approximately –90 ft. This site was evaluated because the upland that could be created would benefit plant operations. A submerged training berm constructed at this depth, with the inside toe at the –90 ft contour would extend into approximately 120 ft of water. Conditions at this site exceed the berm criterion for construction in less than 100 ft of water as well as the criterion for constructing berms on steep slopes. Therefore, this site is eliminated from further consideration.

10.4.1.4 Summary Evaluation of Near-Shore Disposal Sites

Although it appears possible to construct near-shore disposal facilities at Sites 1 and 2, the cost is very high and construction activities would be limited by the tide range. The

high costs are the result of high tidal fluctuation, the steep slopes along the shoreline, and the expense of imported large volumes of sand and gravel for the perimeter berm and clean soil cover. Although there are disadvantages to construction of NCDFs in Ward Cove, this technology is technically feasible at Sites 1 and 2; therefore, it will be carried forward in the development of remedial alternatives to provide a range of alternatives for detailed evaluation.

10.4.2 Confined Aquatic Disposal Facilities

Three locations were considered for CAD: Sites 2, 3, and 4 (Figure 10-7).

10.4.2.1 Shallow Water CAD (Sites 2 and 3)

Sites 2 and 3 are the only shallow water CAD sites available in water depths of less than 100 ft with existing slopes that allow the construction of containment berms (i.e., slopes less than 8H:1V, Figure 10-5). Site 2 is located at the creek mouth and is also a candidate site for near-shore confined disposal. The capacity of this site is approximately 80,000 cy, with a minimum 3-ft cap. The capacity of the CAD at Site 2 is less than that of the NCDF at Site 2 because of the difference in final elevations of the two disposal options. The final elevation of the CAD, including the cap, would be less than 0 ft MLLW, whereas the elevation of the problem sediments in the NCDF would be as high as +7 ft MLLW. (The final elevation of the cap for the NCDF would be as high as +18 ft MLLW.) The conceptual level cost for CAD Site 2 is $14 million.

Site 3 is located in the vicinity of the KPC dock area. The capacity of this site is approximately 156,000 cy, with a minimum 3-ft cap. The top of the CAD cap would be limited to a maximum elevation of –30 ft MLLW to allow for shallow draft vessel traffic, and the potential need for maintenance dredging in certain areas would have to be carefully considered. Even at this elevation, there is concern that vessel movement could damage the integrity of the cap, which may require armoring to ensure there are no vessel prop wash impacts. (It may be more appropriate to construct this site to a maximum elevation of –40 ft MLLW depending on proposed future use.)

The shallow water CAD in the vicinity of the KPC dock (Site 3) would also be constructed with a berm to an elevation of –30 ft MLLW. The berm construction and dredging would be in the same manner as described for the NCDF sites. Cap material would be placed over the problem sediments using a submerged diffuser (assuming the sediment can be capped). The capping material will consist of 2 ft of sand with 1 ft of gravel armoring. The conceptual level cost for this disposal option, including dredging, constructing the berm, and capping, is $9.6 million.

10.4.2.2 Deep Water CAD (Site 4)

A deep water CAD site (Site 4) was considered for the relatively flat area between elevations of –130 and –150 ft MLLW that could potentially accommodate a substantial amount of problem sediment. This site was selected for evaluation because it is outside the medium- and high-density log area and is in an area with relatively gentle slopes. There are significant design, construction, and disposal costs that make this deep water CAD site unrealistic. For example, it would be necessary to construct a downslope containment berm to keep sediments placed on the site from migrating downslope from the site. If the problem sediments have a high water content (such as the Ward Cove organic material), then it will be necessary to use a tremie tube and/or diffuser to place material in the CAD site. Furthermore, placement of a cap is questionable and at best time consuming, especially when the dredged problem sediment has a very low sediment density and is relatively unstable for cap placement. All of these considerations affect constructability and cost as well as the likelihood of agency approval. At this time, no CAD site deeper than 100 ft has been permitted or constructed. The deep water CAD site is not technically or economically feasible and is eliminated from further consideration.

10.4.2.3 Summary Evaluation of Confined Aquatic Disposal Sites

The deep water CAD site has a number of construction difficulties resulting from water depth. Controlling the placement of material, both problem sediments and capping material, is difficult in deep water. The cost is also very high, and the low chemical concentrations and limited toxicity of problem sediments in Ward Cove do not warrant a high cost disposal option. The fact that deep water CAD development with effective confinement of high water content, organic sediments has not yet been accomplished identifies the deep water CAD at Ward Cove as a research and development project. The deep water CAD is therefore not feasible and is eliminated from further consideration.

Shallow water CAD is retained as a technology for development of the remedial action alternatives. Handling and capping of problem sediment are expected to pose a significant challenge. However, special construction methods could be used to construct a CAD along the shoreline. The development of a shallow surface CAD site could be accomplished using the proposed near-shore site at the mouth of Ward Creek (Site 2). Special construction methods would involve disposal of organic sediments into the near-shore dikes (i.e., the same type of dikes that would be constructed for an NCDF) and construction of a cap to a surface elevation of approximately –5 to –10 ft MLLW. After cap placement, the confinement dikes would be lowered to the elevation of the cap surface. Suitable cap armoring would be included in the design to address wave or navigation impacts.

10.5 SUMMARY OF TECHNOLOGY SCREENING

Table 10-2 provides a summary description of confinement technologies, and Table 10-3 summarizes screening considerations. The technologies and process options that were eliminated are shaded (Table 10-3).

In summary, the following technologies and process options are retained for the development of remedial alternatives in Section 11:

Technology	Process Options
Dredging	Mechanical
Containment facility	Upland disposal in KPC ash landfill
	Upland disposal in Washington State landfill
	Shallow CAD
	NCDF
In-place remediation	Thin cap enhanced recovery
Log removal	Mechanical removal of surface logs

TABLE 10-3. SUMMARY EVALUATION OF CONTAINMENT TECHNOLOGIES

Technology	Process Option	Conceptual Cost Estimate	Construction Issues	Effectiveness	Advantages and Disadvantages
In-Place Containment	Capping (thick cap)[a]		Thickness of very soft sediment, presence of logs, and steep slope would cause construction problems.	Reliable containment of sediment if cap could be properly constructed, which is unlikely for reasons previously stated under construction issues.	Capping in shallow areas restricts future navigation. Difficult to construct in Ward Cove.
	Enhanced recovery[a] (thin cap)		Use conventional equipment. Thin layer (e.g., 6–12 in.) may sink in soft sediments. Mounds could be constructed but large quantities and high cost where sediment is greater than 5 ft thick.	Would enhance recovery.	Lowest water quality impact. Some areas would not be completely capped (e.g., around logs) but there would be no significant adverse environmental impacts in exposed areas.
Containment Facility	Sawmill dock NCDF at Site 1	$27 million	Build in shallow water with conventional equipment. Use proven methods.	Reliable containment of sediments. Low short-term release of sediment.	Adds upland acreage adjacent to mill. Lowest capacity for disposal alternatives. High cost for little or no environmental gain.
	Ward Creek NCDF at Site 2	$30 million	Build in moderate depth water with conventional equipment. Use proven methods.	Reliable containment of sediments. Low short-term release of sediment.	Adds upland acreage away from mill; more capacity than Site 1. High cost for little or no environmental gain.
	Ward Creek CAD at Site 2	$14 million	Build in moderate depth water with conventional equipment. Would need to build up dikes and then partially remove them. Use proven methods.	Reliable containment of sediments. Low short-term release of sediment.	Lower capacities than NCDF sites. Significantly less expensive to construct than NCDF at this site.

TABLE 10-3. (cont.)

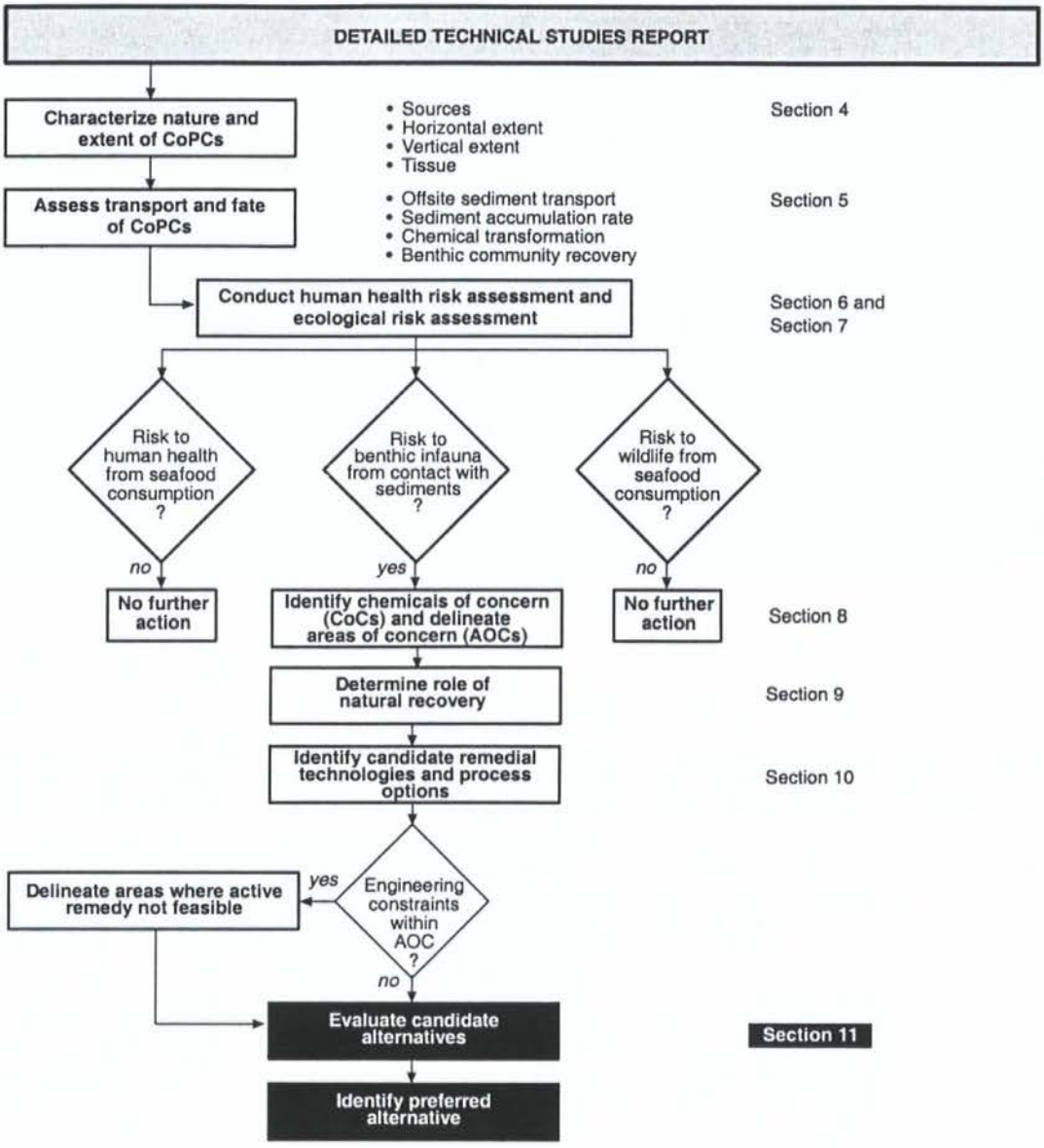
Technology	Process Option	Conceptual Cost Estimate	Construction Issues	Effectiveness	Advantages and Disadvantages
	KPC dock CAD at Site 3	$10 million	Use proven methods; however, water depth and slope steepness could present construction difficulties. Capping would be very difficult or not possible.	Reliable containment of sediments (if could be capped). Some short-term release to water.	CAD to –30 ft could restrict future navigation for deep-draft vessels. Greater capacity than Site 2.
	Deep CAD at Site 4[a]	Prohibitively expensive	Difficult to place sediment and cap in deep water. CAD construction has not been performed in water deeper than 100 ft.	Problem sediment likely to spread over large area. Cap would have varying thickness and discontinuities if it could be constructed at all.	Would not restrict future navigation. Very difficult to construct and obtain permit.
	Upland disposal[a]	Relatively low cost	Conventional equipment and landfill techniques. Possible handling problems with soft sediment.	Reliable containment of sediment. Limited landfill capacity.	Low water quality impact. Low cost if sediment can be placed into existing KPC landfill.

Note: Process options removed from further consideration are shaded.

CAD - confined aquatic disposal
KPC - Ketchikan Pulp Company
NCDF - near-shore confined disposal facility

[a] Costs for this option were not developed during technology screening.

11. ASSEMBLY OF ALTERNATIVES AND DETAILED EVALUATION



The technologies and process options retained for the development of remedial action alternatives for the Ward Cove sediment remediation project were identified in the previous section. In this section, the technologies and process options are assembled into alternatives that meet the RAOs and are evaluated in detail. This evaluation was conducted by Hartman Consulting Corporation and Exponent.

11.1 BASIS FOR DEVELOPING ALTERNATIVES

Several alternatives have been developed that range from no action to an alternative that includes dredging and containing a large volume of the problem sediments. These alternatives are compared against one another to allow selection of an appropriate remedy for Ward Cove. EPA guidance states "in developing alternatives, the range of options will vary depending on site-specific conditions" (U.S. EPA 1988). Although site-specific conditions are a factor at all sites that are investigated and remediated, those at the Ward Cove site present a more complex challenge in the development of the alternatives. The key site-specific conditions that affect development of the alternatives are summarized as follows:

- The sediments are within acceptable limits for human health and wildlife and are of limited toxicity to the benthos.
- The sediment toxicity appears to be related to nonpersistent by-products of the decomposition of organic matter and wood debris (i.e., sulfide, ammonia, and 4-methylphenol).
- There are no "hot spots" of contamination (i.e., there is **not** a small portion of the AOC that contains most of the mass of CoCs). The size of the AOC (87 acres) and the total volume of organic-rich sediment (approximately 840,000 cy, assuming an average thickness of 6 ft) poses unique challenges for balancing benefits and costs.
- Recovery of benthic communities has been occurring and should proceed more rapidly, because pulp-related discharges from the KPC facility have been eliminated and clean sediment is actively accumulating.
- The affected sediments contain wood debris, are fine-grained, and have high water and organic content. From an engineering and remediation perspective, these sediments have very limited structural strength or have essentially no strength, depending on the water content. Difficulties would be encountered in dredging, transporting, disposing, or capping these extremely soft, organic, fine-grained sediments.
- There are significant areas of the bottom covered by sunken logs. In some areas, the sunken logs are several layers thick.
- Ward Cove is in an isolated location with limited road access (i.e., the Ketchikan area is accessible only by air or water and all material transport to and from Ward Cove must be conducted by vessels), the steep surrounding terrain results in limited suitable upland area for disposal or treatment, and there is a lack of a local source of clean capping material.

- There are few potential disposal sites in Ward Cove for dredged sediment because of the bathymetry and limited size of the Cove.
- It is believed that capping or dredging steep slopes (steeper than 4H:1V) in the AOC would not be successful. Capping or dredging in water depths greater than 100 ft would be difficult to achieve. Capping can be performed in deeper water (e.g., 120 ft) when the criteria for capping effectiveness are relaxed (e.g., partial coverage or mounding is acceptable). For the purpose of this report, −120 ft MLLW is used for the maximum depth for thin capping.

The remedy must also be compatible with ongoing and future business operations. Current operations consist of activities related to the Ketchikan sawmill and include towing and storing log rafts, dewatering log bundles at the LTF, sawing and chipping logs, hogging bark, and transferring wood products to barges. The City of Ketchikan has expressed interest in developing a portion of the south shore into a marina.

The alternatives were developed to include a combination of different response actions to accommodate the site-specific conditions discussed above. Given the large area and volume of the AOC (87 acres and 840,000 cy, respectively) and the limited human and environmental risks, thin capping and natural recovery are critical elements of most alternatives. Alternatives include dredging to either address future operational needs (i.e., navigational dredging) or illustrate the costs associated with the most feasible disposal options (shallow CAD and NCDF). For those alternatives that include CAD or NCDF, the volume of material dredged is based entirely on the capacity of the disposal site.

The ability of an alternative to meet RAOs was also an important consideration during development of the alternatives. As presented in Section 8, the RAOs are as follows:

- Reduce sediment toxicity
- Enhance recolonization of surface sediments to support a healthy benthic infaunal community with multiple taxonomic groups
- Provide a benthic macroinvertebrate community that constitutes an abundant food source to larger invertebrates and fishes.

Benthic infauna data for the lowest practicable taxon (usually species) will be collected following remediation to evaluate RAOs.

The alternatives have been developed at a conceptual level for evaluation and comparison purposes. After the remedy is selected, the specific details of the remedial action will be determined during a remedial design so that the remedial action can be implemented by a remediation contractor.

11.2 DESCRIPTION OF ALTERNATIVES

11.2.1 Alternative A1—No Action

The no action alternative is carried forward as a baseline alternative for comparison purposes. No active remediation would occur at the site. Although natural recovery would be expected to occur, no monitoring would be conducted.

11.2.2 Alternative A2—Natural Recovery

Ward Cove sediments are within acceptable limits for human health and wildlife and are of limited toxicity to the benthos. The potential for benthic macroinvertebrate communities in Ward Cove to recover naturally is relatively high, now that the source of fine-grained organic matter has been eliminated. The potential for natural recovery is facilitated by the fact that most toxicity throughout the Cove appears to be related to non-persistent by-products of the decomposition of organic matter (i.e., sulfide, ammonia, 4-methylphenol), rather than persistent chemicals such as metals and organic compounds (e.g., PAHs). The results of the specialized toxicity tests conducted as part of the ecological evaluation (Section 7) further support the potential for natural recovery, because they indicate that sulfide appears to be the major cause of sediment toxicity in sediment samples from most areas of the Cove. Because sulfide appears to be the main toxic component of the sediments, simple aeration of pore water eliminated most toxicity, as was demonstrated in the specialized toxicity test on pore water using *Rhepoxynius abronius* (Section 7.1.4). Aeration of sediments by benthic organisms will occur through irrigation and physical mixing. Physical processes (i.e., the slow burial of surface sediments with clean material) also contribute to natural recovery (Section 9).

As described in PTI (1996), historical studies of benthic macroinvertebrate communities in Ward Cove suggest that recovery has been occurring slowly over the past 20 years. In 1968–1969, FWQA (1970) conducted macroscopic evaluations of benthic communities in the Cove and found few benthic invertebrates. Following installation of the primary treatment system for wastewater at the KPC facility, U.S. EPA (1975) conducted macroscopic evaluations of sediment samples in Ward Cove and found that polychaetes were common at all locations except immediately offshore from the KPC facility. Finally, EVS (1992) evaluated benthic macroinvertebrate communities in Ward Cove in 1992 and found that most communities were dominated by the opportunistic polychaete *Capitella capitata*. Because *C. capitata* is a well-documented indicator species for organic enrichment and one of the first species to colonize organically enriched sediments, its dominance of benthic communities in Ward Cove in 1992 supports the suggestion that recovery was occurring.

Because most discharges from the KPC facility have been eliminated, recovery of benthic macroinvertebrate communities should proceed more rapidly than in the past and should follow the classical patterns of recolonization and recovery documented for organically enriched areas (Pearson and Rosenberg 1978) and dredged material disposal areas

(Rhoads et al. 1977, 1978; Rhoads and Boyer 1982). Those patterns include initial colonization by "pioneering" species, subsequent modification of physical/chemical characteristics, and final colonization by deeper dwelling "equilibrium" species. In general, equilibrium species are associated with a well-oxygenated sediment surface where the redox potential discontinuity commonly reaches depths of over 10 cm (Rhoads and Boyer 1982). The earliest benthic communities in the recovery process tend to consist of large numbers of a few species, whereas the equilibrium communities are characterized by a greater number of species and a more even distribution of individuals among species.

The first organisms to colonize a disturbed area generally are small, opportunistic, tube-dwelling polychaetes, followed by tube-dwelling amphipods (Rhoads and Boyer 1982). Most pioneering species feed near the sediment surface or from the water column and are thereby largely isolated from potentially toxic conditions in deeper sediments. The tube walls isolate the colonizing organisms from ambient surface sediments by controlling the diffusion rate of ambient porewater solutes into the tube environment (Aller 1982). In addition, by aerating the water in their tubes, organisms can effectively isolate themselves from oxidizable porewater constituents such as sulfide. In this manner, they can inhabit sediments that are toxic to free-burrowing organisms. The activities of the pioneering species modify the physical/chemical properties of the sediments so that additional species can colonize them. Such activities include bioturbation, irrigation, particle reworking, and progressively deeper penetration of subsurface sediments (Aller 1982).

Several aspects of the results of the Ward Cove sediment toxicity studies suggest that recovery of benthic macroinvertebrate communities in the Cove can occur more rapidly, now that most KPC discharges have been eliminated. Because the results of the specialized toxicity tests indicate that sulfide may be the primary cause of toxicity in most sediments from the Cove, its rapid oxidation and subsequent toxicity reduction in the presence of oxygen suggest that pioneering tube-dwelling polychaetes and amphipods will be able to successfully colonize the surface sediments and isolate themselves from elevated sulfide concentrations in pore water by irrigating their burrows. Once these pioneering species have colonized the surface sediments, the classical patterns of benthic recolonization and recovery should occur.

The time to achieve an abundant and diverse benthic community is expected to vary for different chemicals and for different locations within Ward Cove. It is likely that recovery times will range from 8 years to more than 20 years (Section 9). Monitoring of the recovery rate of the benthic community will be an important component of the natural recovery alternative.

11.2.3 Alternative B—Thin Capping with Navigational Dredging and Upland Disposal

11.2.3.1 Overview of Alternative

This section presents an overview of an alternative that includes thin capping and navigational dredging with upland disposal of dredged material. Portions of the AOC where thin capping is not practicable would undergo natural recovery. The overview is followed by more detailed information and evaluations regarding the basis for developing the alternative.

The goal for thin capping in Ward Cove is to amend surface sediments (i.e., the biologically active zone) through partial surface cover or dilution. With thin capping, surface coverage is expected to vary spatially, providing variable areas of capped surface sediments and amended surface sediment (i.e., where mixing between capping material and problem sediment occurs) as well as areas where no cap is evident. Thin capping would be accomplished by distributing a thin layer (e.g., 6–12 in.) of cap material to amend the surface sediments or by placing larger amounts of capping material at separate locations to create island mounds of clean sediment. A pilot study would be required to determine whether it would be possible to place clean material as a thin cap rather than a mound and to determine the maximum thickness of the existing soft sediments that could be capped and still result in the desired mixing of surficial sediment and sand for satisfactory enhanced recovery. The water depth at which thin capping could be conducted could also be examined during the pilot study, as well as other design issues such as placement method. For the purpose of alternative development in this report, both methods of thin capping are considered. The two thin capping options for this alternative are each discussed below. Portions of the AOC where thin capping is not practicable would undergo natural recovery.

For the purpose of developing this option in this report, it is assumed that an average of 1 ft of capping material would be distributed over an area of 34–40 acres (Figure 11-1), depending on the post-dredging area requiring capping if native sediments are not reached during dredging. The water depth for this area is less than 120 ft, and the slopes are less than 25 percent. Thin capping would not be conducted in the very high-density log area of the AOC (Figure 11-1). The rationale for not removing the logs and an evaluation of the cost-effectiveness regarding the removal of logs in the very high-density log area are presented later in this section. The specific area that could be capped would need to be examined in more detail during remedial design (i.e., to further evaluate the thickness of the soft sediment, water depths, and other design parameters). A pilot study would be conducted prior to or during the remedial design phase of the project to further evaluate thin capping. The specific objectives and details of the pilot study would be determined further during remedial design. Several approaches for a pilot study could be considered. For the purpose of alternative development in this report, it is assumed that the pilot study would involve pilot-scale laboratory tests conducted on samples of organic sediment from Ward Cove. The tests would be used to determine the degree of cap

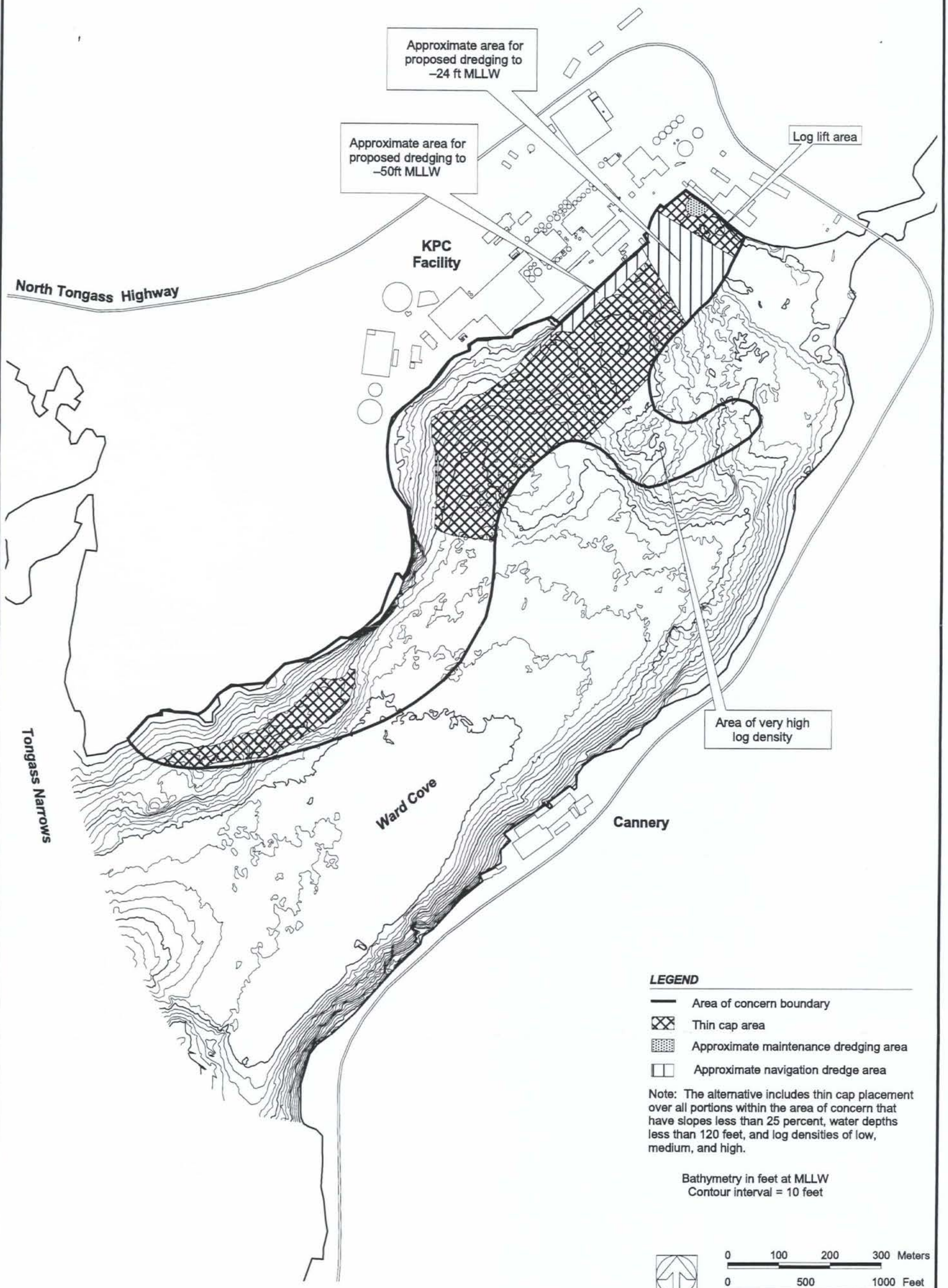


Figure 11-1. Alternative B: Thin capping with navigational dredging and upland disposal.

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formation (i.e., surface sediment amendment) when capping material is released into the water at different rates and with different methods of placement. The pilot-scale testing equipment would be of an appropriate size to minimize the physical and geometric effects of the test equipment on the capping performance to simulate full-scale performance (U.S. EPA 1988). The tests would be used to further define the capping approach:

- If the laboratory tests are successful and a cap is formed (i.e., adequate surface sediment amendment) at a specific release rate, that method would be used to place the first quantity of capping material during of the actual remedial action. Subsequently, the deposition area would be monitored to determine whether the initial release successfully formed a thin cap. If the release proved successful in forming a thin cap, the same method would be used to release the remainder of the material over the thin cap area. If the release proved unsuccessful, then the remainder of the cap material would be placed in mounds.
- If the laboratory tests show that an adequate cap cannot be formed under any of the release scenarios, then the capping material would be placed in mounds in the area that was to be thin capped. The isolated mounds of capping material would be placed throughout an area of 21 acres. This area includes the portion of the AOC where the soft organic material is less than 5 ft thick. The water depth for this area is less than 120 ft.

The source and volume of the capping material needed would be determined during the remedial design and remedial action phase of the project. For the purposes of estimating costs, it is assumed that the source of capping material will come from southeastern Alaska or northern British Columbia, and the volume of capping material will be approximately 65,000 cy.

In addition, approximately 12,300 cy of sediment would be dredged from the area of about 3–6 acres in front of the main dock using a clamshell dredge, because it is assumed that a cap cannot be placed in this portion of the site without affecting navigation. Logs would also be removed in the areas where navigational dredging would occur. The volume of dredged sediment includes approximately 3,300 cy at the western end of the dock to potentially provide a water depth of –50 ft MLLW and approximately 9,000 cy at the eastern end to potentially provide a water depth of –24 ft MLLW. The water depth of –24 ft MLLW is required to ensure limited or no prop wash of the organic-rich material by working vessels at the eastern end and future use of the western end by 40,000-ton cargo vessels or by cruise ships. The areas to be dredged (and the configuration of the navigational dredging area) were preliminarily identified in this report to address those two uses. In addition to navigational dredging, some ongoing maintenance dredging will be conducted near the sawmill log lift. The costs for maintenance dredging are not included in this report. Because the maintenance dredging area is generally dredged on an annual basis, it will not be capped.

The areas preliminarily identified to be dredged for this alternative are shown in Figure 11-1. The approximate area within which navigational dredging may occur and the dredging depths in this area are also shown on the figure. The rationale for dredging to −50 ft MLLW at the western end of the dock and to −24 ft at the eastern end is presented later in this section. The area proposed for thin capping east of the navigational dredge area (the trapezoidal cross-hatched area in the vicinity of the log lift) is used only by shallow draft vessels, and at this time, KPC has no plans to use this area for deeper draft vessels. The current depth is adequate for the small boats currently used, and no dredging is anticipated in this area. Estimated costs for dredging this area are presented later in this section.

After navigational dredging is conducted, the bed surface may still have some residual loose, organic sediment. The actual depth of sediments that can be dredged in each location will depend on site-specific conditions in those locations such as rock formations or other hard subsurface materials. Only sediment and logs would be removed from these areas; dredging would stop if large rocks or bedrock were encountered before the desired depth was reached. At this time, it is not known if the navigational dredging will reach native sediments. The rationale for not dredging to native sediments, including estimated costs to dredge to native sediments, is presented later in this section. If native sediments are not reached during navigational dredging, thin capping would be conducted in that area after the dredging is completed. For the purpose of developing the thin capping alternative, it is assumed that thin capping would be conducted in the navigational dredge area (i.e., 6 acres) after dredging.

Dredged material would be disposed at either the KPC landfill or at an approved offsite landfill. For cost estimating purposes, an offsite landfill in Washington State has been identified. The sediments could be loaded onto barges and transferred to trucks on the pier to haul to the KPC flyash landfill. If the sediments are to be disposed of at an approved landfill in Washington, they would be loaded onto barges with some settlement dewatering in Ward Cove prior to shipping. The barges would be watertight and would transport the sediment to Puget Sound where the sediment would be off-loaded into watertight containers. The containers would be loaded onto railcars and transported to the landfill. At the landfill, the sediment would be further dewatered, as needed, to meet any additional disposal standards at the landfill (e.g., the material may have to meet free liquid criteria such as passing the paint filter test). Dewatering at the landfill may include adding a stabilizing agent and would be conducted in an area lined with a geomembrane (or similar contained area) to prevent leaching of any liquid into the groundwater. After the sediment meets the landfill's disposal standards, it would be disposed in the lined landfill.

In summary, the elements of Alternative B are as follows:

- A pilot study to determine areas and methods for thin capping.
- Placement of a thin cap of fine sand over that portion of the AOC that can be successfully capped. The portion of the AOC that would be

capped and the method of placement would be determined during pilot testing; for the purpose of this report, two scenarios were considered:

  - An average of 1 ft of cap material would be distributed over 34–40 acres, depending on the post-dredging area requiring capping if native sediments are not reached during dredging. Cost estimates are based on capping 40 acres.

  - Isolated mounds of capping material would be placed throughout an area of approximately 21 acres.

- Limited dredging near the existing dock (approximately 12,300 cy over 3–6 acres); thin capping after navigational dredging unless native sediments are reached during dredging.

- Removal of logs in areas to be dredged.

- Settlement dewatering of dredged material in the haul barge.

- Disposal of dredged sediment and logs in an upland landfill.

- Natural recovery as described in Alternative A2.

The time to achieve an abundant and diverse benthic community is expected to vary for different chemicals and for different locations within Ward Cove. The benthic communities in areas that have been capped or amended will be initially eliminated through burial. The time for benthic organisms to recolonize capped or amended areas is expected to take between 3 and 5 years. The time to achieve abundant and diverse benthic communities in areas targeted for natural recovery is uncertain, but expected to range from 8 years to more than 20 years.

11.2.3.2 Rationale for Navigational Dredging Depth

The maximum dredging depth of approximately –50 ft MLLW used in this report is based on the largest vessel that may use the facility. For instance, very large container ships (fourth and fifth generation) would have drafts approaching 40–44 ft. Bulk ore vessels of 100,000–250,000 deadweight tons would have drafts approaching 45–48 ft. However, based on the past port design experience of Hartman Consulting, the population of the local community, and the potential for future industrial development in the borough, it is anticipated that second or third generation container ships, 40,000–80,000 deadweight tons bulk ships, or large cruise ships would be the deepest draft vessels for final design consideration. The design vessel would probably have loaded drafts varying from 27 to 36 ft.

It is likely that future industrial activity in Ward Cove will rely on a combination of bulk container vessels and shallow draft haul barges and tugs. Large oceangoing barge and tug operations that carry bulk ore, containers, or other general cargo and equipment are

commonly used in Alaska by companies such as Crowley Maritime. These barges would require drafts of 15–20 ft, whereas the tugs would have slightly lesser drafts.

A 36-ft draft vessel, with tug assist or with bow thruster, could disturb fine-grained, organic surface sediment at a depth of several feet below the vessel bottom. The computer model Propwash (Hartman 1995) and data from other studies (Blaauw and van de Kaa 1978; Verhey 1983) were used for this report to provide a preliminary estimate of the water depth needed below a 36-ft draft vessel. Propeller sizes of 12 and 15 ft in diameter were considered. The propeller shaft is always located on a vessel such that the tip of the blades are well above the bottom of the vessel (keel). The maximum revolutions per minute (rpms) of a propeller are typically 100–150 rpms. For this report, an rpm value of about one-third of the maximum propeller rpm (i.e., 50 rpms) was assumed for a vessel while berthing. Based on the results of the computer model Propwash and data from other studies, a value of 10 ft below the vessel hull is used in this report. Allowing for a maximum low tide elevation approaching –3 ft MLLW, the maximum depth of dredging would need to be –49 to –50 ft MLLW to ensure no resuspension of the bed sediment. A tug with a 16-ft draft would disturb sediment at 4–5 ft below the hull bottom and, at an extreme low tide, would require a dredge depth of –23 to –24 ft MLLW. Further evaluation of the effects of propeller wash would be conducted during the remedial design. Also, an analysis of potential erosion of subsurface sediments from maneuvering vessels may be conducted during the remedial design.

A distance of 10 ft between the vessel bottom and the Cove sediments is consistent with a recent dredging project proposed by the Corps for Cook Inlet (near Fire Island). For that project, it was noted that cargo ships that visit Anchorage have a draft of 27–29 ft. Because vessel owners and insurers prefer to have at least 10 ft between their ships and the sediments, the project calls for the channel to be dredged to 39 ft (U.S. EPA 1999a).

An existing dock at the KPC facility runs parallel with the shoreline in an approximate direction of southwest to northeast. In general, the depth of water at the western end of the dock is greater than at the eastern end of the dock. Also, the area of the Cove in the vicinity of the western end of the dock (i.e., the approach a vessel would take to reach the dock) has deeper water than the area in the vicinity of the eastern end of the dock. Therefore, less dredging would be required to accommodate deeper draft vessels at the western end of the dock. For this report, it is assumed that the area along the western end of the dock would be dredged to –50 ft MLLW. The dredging would proceed eastward along the dock, as needed, to accommodate the design vessel. The shallower area immediately east of the dock would be dredged to –24 ft MLLW to accommodate shallow draft haul barges and tugs in that area.

The resulting natural slope of the sediment "connecting" the two dredging areas (i.e., the –50 ft MLLW deeper area to the west and the –24 ft MLLW shallower area to the east) may present a problem for final design because the strength of the fine-grained organic sediment is low. As a result, the slope between the –24 ft and the –50 ft bed could result in significant sloughing of sediment from the shallower area to the deeper area. This material would then have to be removed to ensure the deep berth area is adequate for the

proposed vessel operation. These design issues and parameters will be evaluated during the remedial design. In addition, the slope between the two dredging areas and the adjacent thin capping areas will need to be evaluated during the remedial design.

11.2.3.3 Rationale for Not Dredging to Native Sediments in Areas Targeted for Navigational Dredging

Navigational dredging is a component of the remedy that supports an important beneficial use of Ward Cove, industrial activity, and related economic benefits to the local community. Navigational dredging will remove a portion of the problem sediments; however, it is not intended to be a stand-alone cleanup action. Dredging to native sediments in areas targeted for navigational dredging is not currently included in the remedy because the added costs are not believed to be warranted by the limited toxicity of problem sediments.

The limited toxicity of sediments in Ward Cove is a very important consideration in remedial planning. Only the upper 10 cm of sediments is associated with toxicity to selected organisms. Thus, the upper 10 cm is the focus of the recommended remedial method, thin capping. Removal of the 6–12 ft (180–360 cm) of sediments underlying this 10-cm surface was demonstrated to be extremely costly because of the large volume of dredged material produced and the high costs of transport and disposal.

Costs for dredging to native sediment and thin capping in areas targeted for navigation are described in the next section.

11.2.3.4 Costs for Dredging to Native Sediment

The depth of non-native material remaining after dredging cannot be estimated with certainty based on available information. Only one deep core station (Station 5) was located in the area considered for navigational dredging. The core collected from this station indicates a thickness of organic-rich sediments of 7 ft. The design vessel, the specific areas that will actually be dredged for navigational purposes, and the dredging depth are yet to be determined during remedial design. Even if native material were encountered and the area were overexcavated, some residual organic-rich material is likely to remain (the dredge bucket may not be able to remove all of the soft organic material), or organic material from the side slopes may slump onto the bottom of the dredged area.

For the purpose of this preliminary evaluation, costs for dredging to native sediment are estimated for two scenarios: 1) a small volume, small area scenario and 2) a larger volume, larger area scenario. Each scenario is briefly discussed below.

Small Volume/Area Scenario—In the small volume/area scenario, it is assumed that 2 acres will be dredged for navigational purposes and the dredge volume will be 12,300 cy. The average dredging depth would be about 4 ft. To reach native

sediments, it is assumed that an additional 3 ft of organic-rich material would need to be removed (approximately 9,700 cy). Of this material, it is assumed that 3,700 cy could be disposed in the KPC landfill, which has a capacity of 16,000 cy. The remaining 6,000 cy would be transported to Puget Sound for disposal in Washington State. The total capital cost for the additional 3 ft of dredging, including disposal, is approximately $880,000. This total does not include the costs for navigational dredging or post-dredging capping.

Larger Volume/Area Scenario—In the larger volume/area scenario, it is assumed that 6 acres will be dredged for navigational purposes. It is also assumed that an additional 3 ft of organic-rich material would need to be removed throughout the 6 acres (approximately 29,000 cy). All of this material would be transported to Puget Sound for disposal in Washington State. The total capital cost for the additional 3 ft of dredging, including disposal, is approximately $3.7 million. This total does not include the costs for navigational dredging or post-dredging capping.

11.2.3.5 Costs for Dredging Sediment Adjacent to the Log Lift Area

Only shallow draft vessels such as the small boats (i.e., "log broncos") that are used to work with the floating logs use the area near the log lift area. At this time, KPC has no plans to use this area for deeper draft vessels. The current depth is adequate for the small boats currently used, and no dredging is anticipated in this area.

As shown on Figure 11-1, there is a small maintenance dredge area in the vicinity of the log lift where very small volumes of sediments are occasionally removed, as needed. The removed sediments are currently disposed in the KPC landfill. Floating logs are handled in the log lift area, including the area designated for maintenance dredging. The KPC sawmill has an LTF permit to manage and transfer the logs for the sawmill. No specific zone of deposit is associated with the LTF permit for the sawmill.

The trapezoidal cross-hatched area adjacent to the log lift area (exclusive of the maintenance dredge area) is approximately 2 acres. If 6 ft of material is removed from this entire area, including side slopes, the volume of material removed would be approximately 20,000 cy.

The costs for dredging the sediments adjacent to the log lift area will vary according to the amount of navigational dredging conducted. However, because the KPC landfill has limited capacity, most if not all of the sediment dredged from that area would likely need to be disposed in an offsite landfill such as the one in Washington State. For the purposes of this cost estimate, it was assumed that 3,700 cy of the 20,000 cy of sediment could be disposed in the KPC landfill and the remaining dredged sediment would be transported to Puget Sound for disposal in Washington State. The total capital cost for dredging and disposing of 20,000 cy is approximately $2.1 million. This total does not include post-dredging capping or removal/disposal of logs that may be present.

11.2.3.6 Rationale for Not Including Removal of Sunken Logs as an Element of Remedial Alternatives

Within the AOC, the toxic effects identified by sediment toxicity tests are believed to be attributable to substances (i.e., sulfide, ammonia, and 4-methylphenol) that are generated *in situ* as a result of degradation of organic matter in the soft sediments found among the sunken logs. There is no reason to believe that the sunken logs are the source of the toxicity; in fact, sunken logs do not pose a toxic risk to human health or the environment (U.S. EPA 1999b). Sunken logs have been observed at 15 of 24 stations within the AOC and at 10 of 18 stations outside the AOC (U.S. EPA 1999b). The lack of significant sediment toxicity at any of the stations outside the AOC suggests that there is no link between the presence of sunken logs and the observed toxicity. It has been suggested that organic compounds leaching from logs submerged in seawater could have adverse effects on marine organisms. However, research indicates that leaching of water-soluble substances from a log begins immediately after the log enters the water, and, as time passes, these substances are depleted and no further leaching takes place (Tetra Tech 1996). The available information suggests that most of the logs on the bottom of Ward Cove have been there for 30 or more years, making it unlikely that there is any ongoing leaching of such substances from those logs. Furthermore, it is unlikely that there will be a significant ongoing source of new logs sinking to the bottom of the Cove (U.S. EPA 1999b).

Portions of the AOC have been proposed for sediment remediation through a combination of dredging, thin capping, and natural recovery. Sunken logs will be removed from the area proposed for dredging, but not from areas proposed for thin-layer capping. Further, in an 8-acre area of the AOC, the presence of a very high density of sunken logs (i.e., >500 logs/10,000 m^2) would tend to compromise the effectiveness of thin capping, and it is not practicable to remove the logs and then place a thin-layer cap (see next section).

11.2.3.7 Evaluation of the Cost-Effectiveness of Removing Sunken Logs from the Area of Very High Log Density, Prior to Thin Capping

An elongate lobe of the AOC partially overlaps an area with very high log density (i.e., >500 logs/10,000 m^2) located near the center of the Cove (see Figures 10-5 and 11-1). The area of overlap has been estimated to be approximately 8 acres. This elongate lobe is included as part of the AOC on the basis of exceedances of the MCUL values for two sediment toxicity tests (i.e., *Rhepoxynius abronius* and *Dendraster excentricus*) at two stations (i.e., Stations 16 and 17; see Figures 8-3 and 8-5).

It has been suggested that, within the approximately 8-acre portion of the AOC that also has a very high density of sunken logs, soft-bottom sediments could be remediated with placement of a thin cap if the logs were first removed. It should be noted that the 8-acre area of overlap represents less than 10 percent of the total area of the proposed AOC (i.e., 87 acres), and only about 3 percent of the total area of Ward Cove (approximately 247 acres). The cost for removing the sunken logs from that area was evaluated by KPC and Hartman Consulting in 1998. The total cost to remove the sunken logs from the

bottom, transport the logs to Washington State for disposal, and place a thin (i.e., 1-ft) sand cap over the soft-bottom substrate within the approximately 8-acre area was estimated to be approximately $1.5 million.

To justify such an expenditure, a demonstration of its cost-effectiveness is warranted. Under the National Contingency Plan (NCP), a demonstration of cost-effectiveness is generally conducted as one element of the selection of the most appropriate remedial alternative. Strictly speaking, the cost-effectiveness evaluation under the NCP is conducted to select a remedial alternative from among a suite of such alternatives, all of which have already been shown to provide adequate protection of the public health and welfare and the environment. If the remedial alternatives are not equally feasible, reliable, and capable of providing the same level of protection, the cost, level of protection, and reliability of each alternative must be considered in selecting the most appropriate alternative among them. As discussed above, attainment of the RAOs for Ward Cove does not require that all of the AOC be actively remediated (i.e., dredged or capped). The 8-acre area could be omitted from active remediation without compromising achievement of the overall RAOs for Ward Cove. Therefore, a range of remedial alternatives, each of which was capable of providing adequate protection of the public health and welfare and the environment, was not developed for the 8-acre area. Nevertheless, an evaluation of the cost-effectiveness of removing the logs from that area and then placing a thin sand cap can be made by comparing the total cost per acre for that scenario with the total cost per acre for thin capping alone in other portions of the AOC. If the two scenarios are approximately equal in their effectiveness but the cost of one scenario is much greater, a proportional relationship does not exist between cost and effectiveness. Hence, the more expensive scenario may not be cost-effective.

The estimated total cost for placing a thin sand cap over 40 acres of the AOC is $2,058,613 (see Tables 11-1 through 11-4), or $51,465 per acre. As indicated above, the total cost to remove the sunken logs from the bottom, transport the logs to Washington State for disposal, and place a thin sand cap over the approximately 8-acre area was estimated to be approximately $1.5 million, or $187,500 per acre. Hence, the added cost of having to remove and dispose of the logs prior to thin capping increased the cost per acre for the 8-acre area by a factor in excess of 3.6-fold. This difference in cost suggests that the latter scenario is not cost-effective, considering that it would only achieve a similar condition on the bottom as the thin capping over the 40-acre area. At least two factors could potentially increase the costs of the latter scenario even further. First, the maximum depth of the 8-acre area approaches the depth at which it may not be considered practicable to place a thin cap. A pilot study may have to be conducted to demonstrate whether it is practicable to place a thin cap at such depths, potentially adding to the cost of this scenario. Second, the process of removing sunken logs from the bottom has the potential to cause releases of suspended sediments to the water column. These resuspended sediments may cause short-term impacts to the environment. Thus, log removal efforts may require consideration of use of silt curtains, including any engineering limitations on their use (e.g., depth, currents). Such factors may further increase the cost of this scenario. Furthermore, the disturbance of the sediments inherent in removing the logs from the bottom would, as a result of the release of sediments to the water column,

TABLE 11-1. COST ESTIMATE FOR ALTERNATIVE B, OPTION 1

Item	Quantity	Unit Cost	Cost
Construction Costs			
Placement of cap sand (40 acres[a])	64,533 cy	$6.90	$445,280
Delivery of sand to dockside	64,533 cy	$25.00	$1,613,333
Dredging	12,300 cy	$13.83	$170,150
Placement in KPC Landfill	12,300 cy	$7.13	$87,638
Off-loading of logs	1,400 tons	$15.76	$22,064
Chipping of logs at KPC	1,400 tons	$15.00	$21,000
Mobilization	1 lump sum	$100,000	$100,000
Field overhead	2 months	$15,000	$30,000
Water quality monitoring	30 days	$1,500	$45,000
Construction Cost			$2,534,465
Contingency	30 percent		$760,339
Construction Estimate			$3,294,804

Summary	Direct Costs	Percentage	Cost			
Cap 40 acres[a]	$2,058,613	88.9	$2,928,132	**Cap Unit Cost**	$ 73,203	per acre
Dredge 12,300 cy	$257,788	11.1	$336,672	**Upland Unit Cost**	$ 30	per cy
Sum	$2,316,401	100.0	$3,294,804			

Non-Construction Costs			
Capping pilot study	1 lump sum	$200,000	$200,000
Design	8 percent of construction		$263,584
Capping/dredging monitoring	40 days	$3,000	$120,000
Construction management	4 percent of construction		$131,792
Non-Construction Estimate			$715,377
Total Estimated Capital Costs			$4,010,181
Periodic Monitoring Costs			
Monitoring every other year for 10 years	5 events	$120,000	
Present worth of 10 years monitoring			$450,000
Total Estimated Costs			$4,500,000

Note: cy - cubic yard
KPC - Ketchikan Pulp Company

[a] Costs are based on 40 acres, which is the maximum area that would require capping.

TABLE 11-2. COST ESTIMATE FOR ALTERNATIVE B, OPTION 2

Item	Quantity	Unit Cost	Cost
Construction Costs			
Placement of cap sand (40 acres[a])	64,533 cy	$6.90	$445,280
Delivery of sand to dockside	64,533 cy	$25.00	$1,613,333
Dredging	12,300 cy	$13.83	$170,150
Transport to Puget Sound	12,500 tons	$17.28	$216,000
Disposal to landfill	12,500 tons	$50.00	$625,000
Stabilizing agent to dewater	1,200 tons	$40.00	$48,000
Off-loading of logs	1,400 tons	$15.76	$22,064
Chipping of logs at KPC	1,400 tons	$15.00	$21,000
Mobilization	1 lump sum	$100,000	$100,000
Field overhead	2 months	$15,000	$30,000
Water quality monitoring	30 days	$1,500	$45,000
Construction Cost			$3,335,827
Contingency	30 percent		$1,000,748
Construction Estimate			$4,336,576

Summary	Direct Costs	Percentage	Cost			
Cap 40 acres[a]	$2,058,613	66.0	$2,863,377	**Cap Unit Cost**	$ 71,584	per acre
Dredge 12,300 cy	$1,059,150	34.0	$1,473,198	**Upland Unit Cost**	$ 120	per cy
Sum	$3,117,763	100.0	$4,336,576			

Non-Construction Costs			
Capping pilot study	1 lump sum	$200,000	$200,000
Design	8 percent of construction		$346,926
Capping/dredging monitoring	40 days	$3,000	$120,000
Construction management	4 percent of construction		$173,463
Non-Construction Estimate			$840,389
Total Estimated Capital Costs			$5,176,965
Periodic Monitoring Costs			
Monitoring every other year for 10 years	5 events	$120,000	
Present worth of 10 years monitoring			$450,000
Total Estimated Costs			$5,600,000

Note: cy - cubic yard
KPC - Ketchikan Pulp Company

[a] Costs are based on 40 acres, which is the maximum area that would require capping.

TABLE 11-3. COST ESTIMATE FOR ALTERNATIVE C

Item	Quantity	Unit Cost	Cost
Construction Costs			
Placement of cap sand (34 acres)	54,853 cy	$6.90	$378,488
Purchase and delivery of sand	54,853 cy	$25.00	$1,371,333
Dredging	80,000 cy	$13.83	$1,106,667
Placement of sediment in CAD	80,000 cy	$6.90	$552,000
Placement of berm gravel	135,000 cy	$13.83	$1,867,500
Purchase of berm gravel	135,000 cy	$8.00	$1,080,000
Delivery of berm gravel	135,000 cy	$29.25	$3,948,750
Placement of CAD cover	22,200 cy	$6.90	$153,180
Purchase of cover spalls	7,326 cy	$20.00	$146,520
Delivery of spalls	7,326 cy	$29.25	$214,286
Purchase and delivery of sand	14,807 cy	$25.00	$370,185
Off-loading of logs	1,600 tons	$15.76	$25,216
Chipping of logs at KPC	1,600 tons	$15.00	$24,000
Mobilization	1 lump sum	$100,000	$100,000
Field overhead	4 months	$15,000	$60,000
Water quality monitoring	90 days	$1,500	$135,000
Construction Cost			$11,533,125
Contingency	30 percent		$3,459,937
Construction Estimate			$14,933,062

Summary	Direct Costs	Percentage	Cost			
Cap 34 acres	$1,749,821	15.6	$2,334,480	**Cap Unit Cost**	$ 68,661	per acre
Dredge and place 80,000 cy	$1,658,667	14.8	$2,212,869			
Construct CAD berm and cover	$7,829,637	69.7	$10,445,713	**CAD Unit Cost**	$ 158	per cy
Sum	$11,238,125	100.0	$14,993,062			

Non-Construction Costs			
Capping pilot study	1 lump sum	$200,000	$200,000
Design	5 percent of construction		$749,653
Capping/dredging monitoring	40 days	$3,000	$120,000
Construction management	2.5 percent of construction		$374,827
Non-Construction Estimate			$1,444,480
Total Estimated Capital Costs			$16,437,541
Periodic Monitoring Costs			
Monitoring every other year for 10 years	5 events	$120,000	
Present worth of 10 years monitoring			$450,000
Total Estimated Costs			$16,900,000

Note CAD - confined aquatic disposal
cy - cubic yard
KPC - Ketchikan Pulp Company

TABLE 11-4. COST ESTIMATE FOR ALTERNATIVE D

Item	Quantity	Unit Cost	Cost
Construction Costs			
Placement of cap sand (34 acres)	54,853 cy	$6.90	$378,488
Delivery of sand to dockside	54,853 cy	$25.00	$1,371,333
Dredging	176,400 cy	$13.83	$2,440,200
Placement of sediment in NCDF	176,400 cy	$11.50	$2,028,600
Placement of berm gravel	202,500 cy	$27.67	$5,602,500
Purchase of berm gravel	202,500 cy	$8.00	$1,620,000
Delivery of berm gravel	202,500 cy	$29.25	$5,923,125
Placement of cover	81,481 cy	$11.50	$937,037
Purchase and delivery of sand	81,481 cy	$25.00	$2,037,037
Off-loading of logs	2,100 tons	$15.76	$33,096
Chipping of logs at KPC	2,100 tons	$15.00	$31,500
Mobilization	1 lump sum	$100,000	$100,000
Field overhead	8 months	$15,000	$120,000
Water quality monitoring	180 days	$1,500	$270,000
Construction Cost			$22,892,916
Contingency	30 percent		$6,867,875
Construction Estimate			$29,760,791

Summary	Direct Costs	Percentage	Cost			
Cap 34 acres	$1,749,821	7.8	$2,324,522	**Cap Unit Cost**	$68,368	per acre
Dredge and place 160,000 cy	$4,468,800	19.9	$5,936,505			
Construct NCDF berm and cover	$16,184,295	72.2	$21,499,765	**NCDF Unit Cost**	$155.53	per cy
Sum	$22,402,916	100.0	$29,760,791			

Non-Construction Costs				
Capping pilot study	1 lump sum		$200,000	$200,000
Design	5 percent of construction			$1,488,040
Capping/dredging monitoring	40 days		$3,000	$120,000
Construction management	2.5 percent of construction			$744,020
Non-Construction Estimate				$2,552,059
Total Estimated Capital Costs				$32,312,851
Periodic Monitoring Costs				
Monitoring every other year for 10 years	5 events		$120,000	
Present worth of 10 years monitoring				$450,000
Total Estimated Costs				$32,800,000

Note: cy - cubic yard
KPC - Ketchikan Pulp Company
NCDF - near-shore confined disposal facility

reduce the short-term effectiveness of this scenario, providing further justification for leaving the logs in place.

In summary, an evaluation was conducted of the cost-effectiveness of removing the logs from the 8-acre area prior to placement of a thin cap over that area. The cost of that scenario was found to be far out of proportion to the questionable benefits of replacing one type of habitat (hard bottom) with another type (soft bottom). The toxicity testing that was applied to sediments that had accumulated in the interstices between logs is not a direct measure of the quality of the hard bottom habitat represented by logs. Furthermore, the logs provide substrate for a variety of marine organisms that would otherwise be unlikely to inhabit this portion of Ward Cove. Finally, the logs are in relatively deep water and would not interfere with the intended uses of Ward Cove.

11.2.4 Alternative C—Thin Capping with Dredging and Disposal in a Shallow, Subtidal CAD at Site 2

In this alternative, a thin layer cap (or mounds) would be placed over portions of the AOC (Figure 11-2). For cost estimating purposes, it is assumed that 34 acres would be thin capped and the volume of capping material would be approximately 55,000 cy. Assumptions regarding the source of the capping material are presented in the development of Alternative B. Approximately 80,000 cy of sediment would be dredged using a clamshell dredge. This material would be taken from the area in front of the main dock and from areas with thick soft sediments that may not be feasible to cap. The approximate area where dredging would occur is shown in Figure 11-2. To remove a majority of the highly organic matter located in front of the main dock, final depths may be achieved that are deeper than the navigation depths required for Alternative B. After dredging, the bed surface will still have loose, organic sediment. Although this alternative includes the option of thin capping after navigational dredging, the cost estimate for this alternative was based on thin capping of only 34 acres and not 40 acres. The dredged material would be transported and placed by bottom dumping into a rectangular area bounded on three sides by constructed berms and on the fourth side by the shoreline. The area for development of Site 2 is limited by a very high density of logs to the northwest, steep slopes to the southwest, and by Ward Creek to the northeast. It is the capacity of the available site area and depth that determines the amount of dredging and method of placement (bottom dumping) for this alternative.

Initially, Site 2 would be designed, constructed, and filled as if it were an NCDF site as described in Alternative D, with the berms constructed to an elevation above high tide. The high berms would allow the sediments to settle and be dewatered. The sediments placed in the CAD would gradually be dewatered until sufficiently stable to be capped (the settled sediments would be at or below an elevation of about –3 ft MLLW). The cover of the CAD site would then be constructed in the intertidal elevation to limit the amount of cover material required to ensure cap integrity from wave and other erosive forces. The cover sand would be placed by clamshell. Some gravel and rock cover is required to armor the intertidal cap against erosion; therefore, the cost estimate includes a minimum of riprap rock 1 ft thick on the surface, as well as a minimum 2-ft thickness of

gravelly sand. The high berms would be removed to the height of the final cover. The berm material that is removed would likely be used as part of the final cover. Log removal activities would occur in the portions of the AOC identified for dredging activities.

In summary, the elements of Alternative C are as follows:

- Thin cap (or mound) on a portion of the AOC. For cost estimating purposes, it is assumed that 34 acres would be thin capped and the volume of capping material would be approximately 55,000 cy.
- Dredging of approximately 80,000 cy of sediment (up to 9 ft over approximately 7–8 acres).
- Thin capping of the dredged area unless native sediments are reached during dredging (not included in cost estimate for this alternative).
- Disposal of dredged sediment by bottom dumping into Site 2 CAD.
- Removal of logs in areas to be dredged.
- Natural recovery for the area that is too steep or deep or has a very high density of logs and for a portion of the area that has a thick layer of soft organic material.

The time to achieve an abundant and diverse benthic community is expected to vary for different chemicals and for different locations within Ward Cove. The benthic communities in areas that have been capped or amended will be initially eliminated through burial. The time for benthic organisms to recolonize capped or amended areas is expected to take between 3 and 5 years. The time to achieve abundant and diverse benthic communities in areas targeted for natural recovery is uncertain, but expected to range from 8 years to more than 20 years. Dredged areas that are not capped or amended are expected to have recovery times similar to those areas targeted for natural recovery because some problem sediments are expected to remain in dredged areas after dredging has occurred (see discussion in Section 10 on problems with effective dredging of Ward Cove problem sediments).

11.2.5 Alternative D—Thin Capping with Dredging and Disposal in a Near-Shore Confined Disposal Facility at Site 2

In this alternative, a thin layer cap (or mounds) would be placed over portions of the AOC (Figure 11-3). For cost estimating purposes, it is assumed that 34 acres would be thin capped and the volume of capping material would be approximately 55,000 cy. Assumptions regarding the source of the capping material are presented in the development of Alternative B. Approximately 175,000 cy of sediment would be dredged using a clamshell dredge. This material would be taken from the area in front of the main dock. The approximate area where dredging would occur is shown in Figure 11-3. To remove

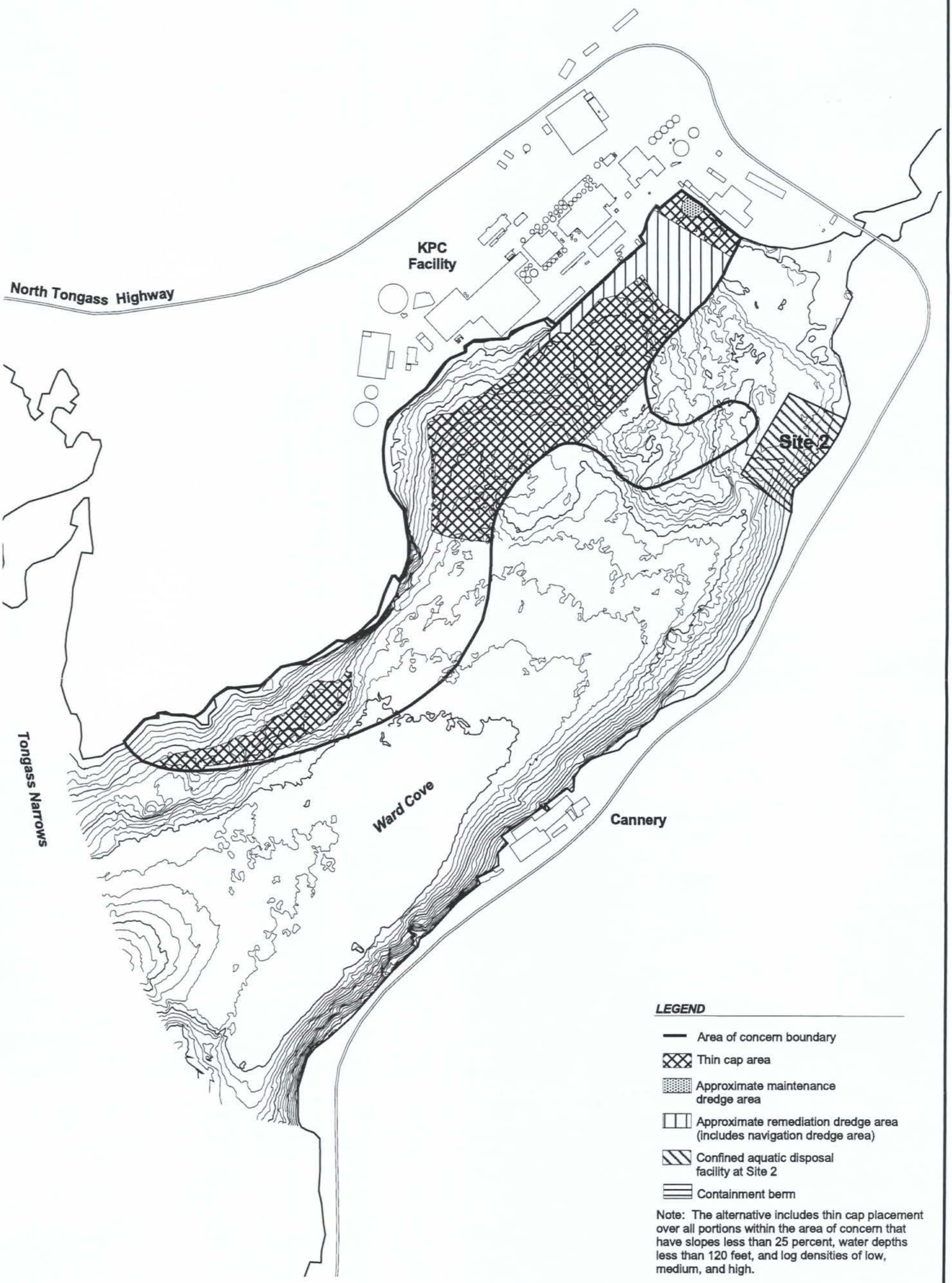

Figure 11-2. Alternative C: Thin capping with dredging and confined aquatic disposal at Site 2.

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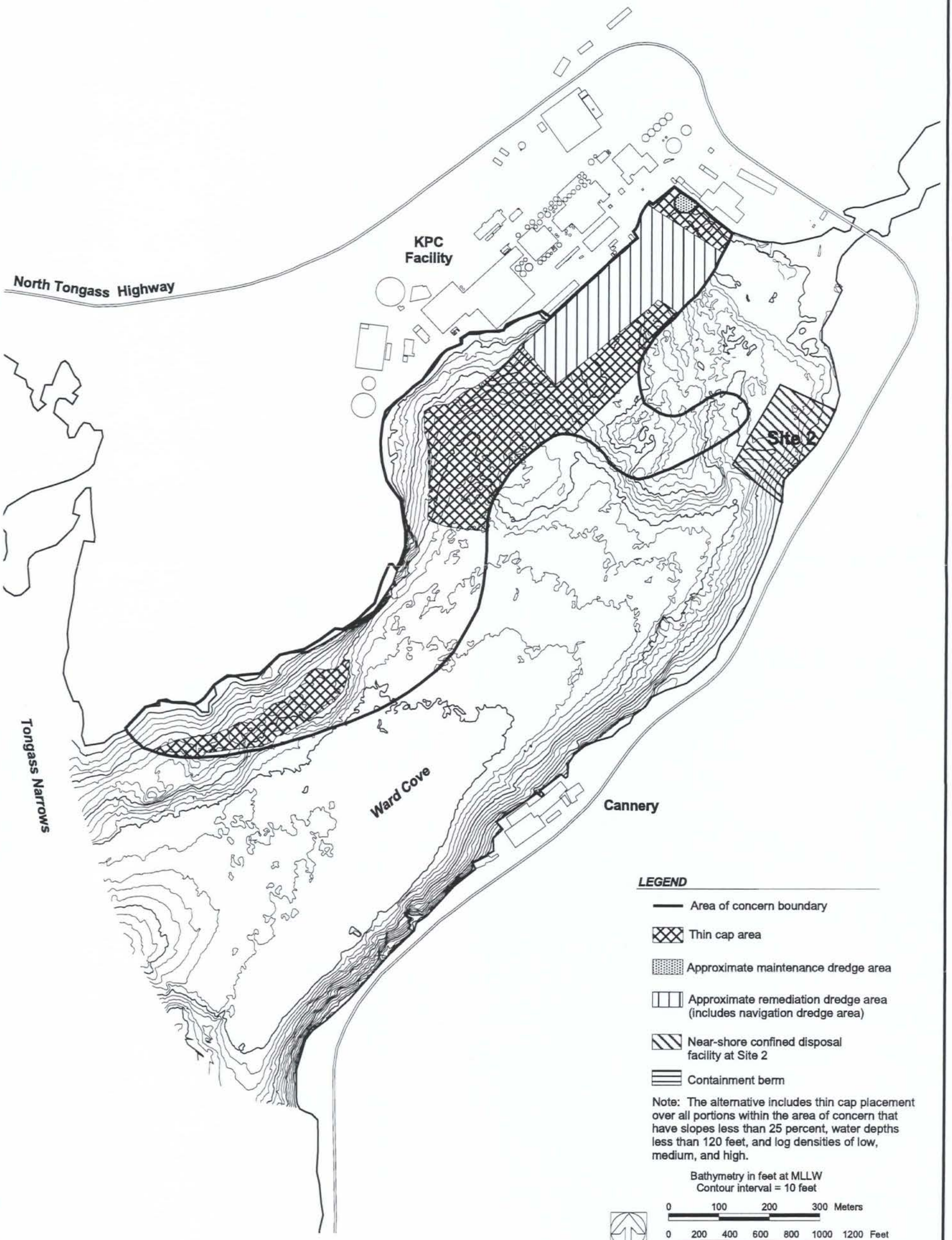

Figure 11-3. Alternative D: Thin capping with dredging and near-shore confined disposal at Site 2.

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the majority of the highly organic matter located in front of the main dock, final depths may be deeper than the navigation depths stated in Alternative B. After dredging, the bed surface will still have loose, organic sediment. Although this alternative includes the option of thin capping after navigational dredging, the cost estimate for this alternative was based on thin capping of only 34 acres and not 40 acres. The dredged material would be placed in a constructed near-shore site (Site 2), which would be engineered to isolate the contaminants from the environment. It is the capacity of the near-shore site that determined the amount of dredging for this alternative. Problem sediments would be placed below the groundwater elevation (+7 ft MLLW) to ensure that they will always remain saturated. The dredged sediments would be placed to an elevation of +2 ft MLLW by bottom dump barge and to an elevation of +7 ft MLLW by subsequent handling. The sand cover from +7 to +18 ft MLLW would be delivered by barge and placed hydraulically or by clamshell and dozers. Limited log removal activities would occur in the portions of the AOC identified for dredging activities.

In summary, the elements of Alternative D are as follows:

- Thin cap (or mound) on a portion of the AOC. For cost estimating purposes, it is assumed that 34 acres would be thin capped and the volume of capping material would be approximately 55,000 cy.
- Dredging of approximately 175,000 cy of sediment (up to 9 ft over approximately 12–14 acres).
- Thin capping of the dredged area unless native sediments are reached during dredging (not included in cost estimate for this alternative).
- Disposal of dredged sediment by bottom dumping and subsequent handling of sediment in NCDF Site 2.
- Removal of logs in areas to be dredged.
- Natural recovery for the area that is too steep or deep or has a very high density of logs and for a portion of the area that has a thick layer of soft organic material.

The time to achieve an abundant and diverse benthic community is expected to vary for different chemicals and for different locations within Ward Cove. The constraints on recovery time for Alternative D are the same as those described for Alternative C.

11.2.6 Alternative E—Thin Capping with Dredging and Disposal in a Near-Shore Confined Disposal Facility at Site 1

This alternative would include dredging and confined disposal of approximately the same volume of sediment as Alternative D. In addition, a thin layer cap (or mounds) would be placed over portions of the AOC (Figure 11-4). For cost estimating purposes, it is assumed that 27 acres would be thin capped and the volume of capping material would be

approximately 44,000 cy. Assumptions regarding the source of the capping material are presented in the development of Alternative B. Approximately 155,000 cy of sediment would be dredged using a clamshell dredge. The approximate area where dredging would occur is shown in Figure 11-4. It is the capacity of the NCDF site that determined the amount of dredging for this alternative. This alternative is similar to Alternative D except for the location of the NCDF and the difference in the capacity of the NCDF. In addition, the area that would be capped is somewhat reduced from that in Alternative D. Because Site 1 is located within the AOC, it encompasses a portion of the area that would otherwise be capped.

In summary, the elements of Alternative E are as follows:

- Thin cap (or mound) on a portion of the AOC. For cost estimating purposes, it is assumed that 27 acres would be thin capped and the volume of capping material would be approximately 44,000 cy.
- Dredging of approximately 155,000 cy of sediment (up to 9 ft over approximately 10–12 acres).
- Thin capping of the dredged area unless native sediments are reached during dredging (not included in cost estimate for this alternative).
- Disposal of dredged sediment by bottom dumping and subsequent handling of sediment in NCDF Site 1.
- Removal of logs in areas to be dredged.
- Natural recovery for the area that is too steep or deep or has a very high density of logs and for a portion of the area that has a thick layer of soft organic material.

The time to achieve an abundant and diverse benthic community is expected to vary for different chemicals and for different locations within Ward Cove. The constraints on recovery time for Alternative E are the same as those described for Alternative C.

11.3 DESCRIPTION OF EVALUATION CRITERIA

CERCLA and the regulations developed thereunder are helpful in evaluating remedial alternatives. The NCP (40 CFR 300) requires that each alternative be evaluated according to nine specific criteria. The purpose of the evaluation is to identify the advantages and disadvantages of each alternative and thereby assist in the decision making process. The nine specific criteria are all important, but they are grouped into three sets of criteria that are weighted differently in the decision process. These nine criteria are presented below:

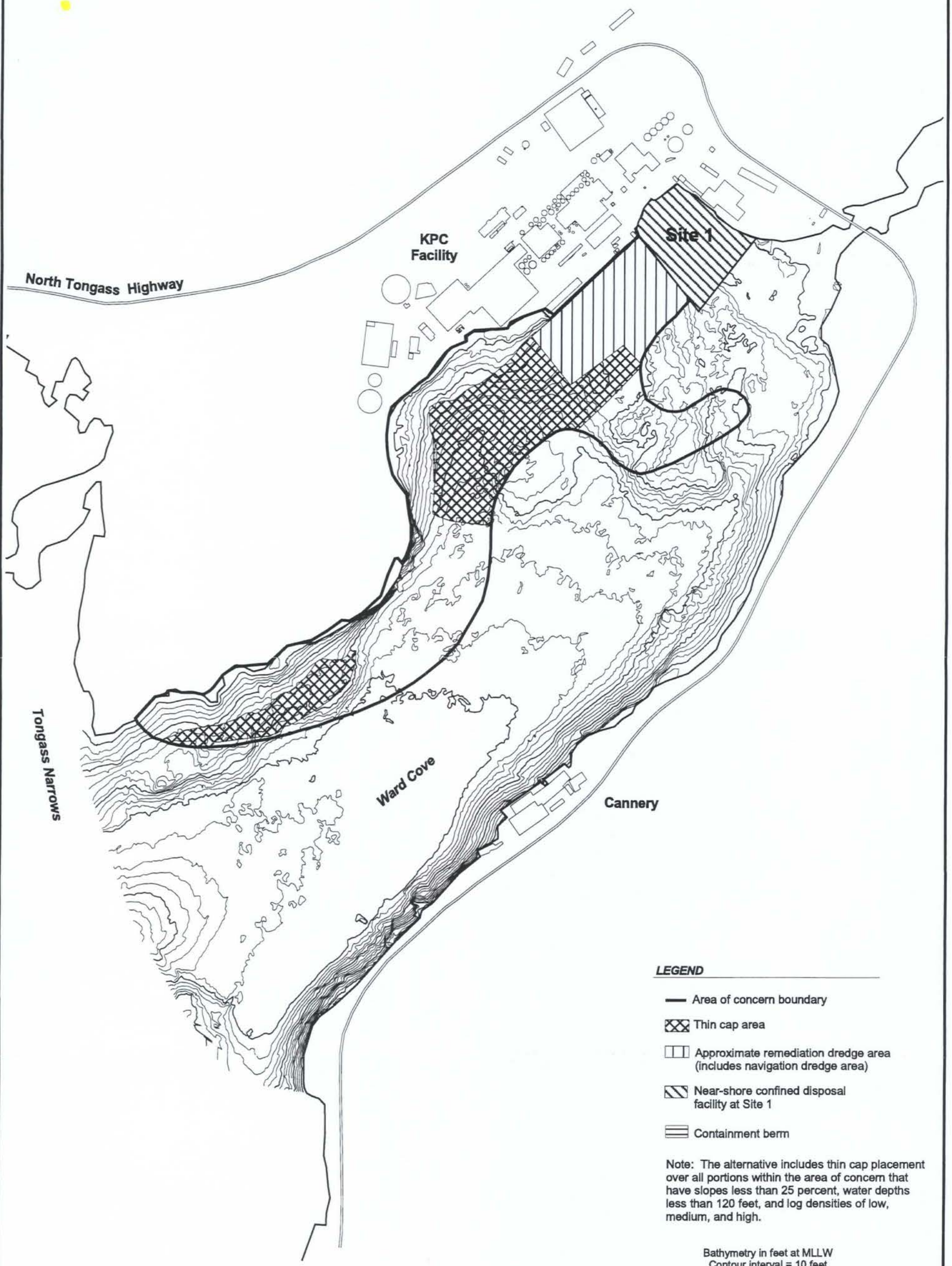

Figure 11-4. Alternative E: Thin capping with dredging and near-shore confined disposal at Site 1.

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- Threshold Criteria
  - Overall protection of human health and the environment
  - Compliance with ARARs
- Primary Balancing Criteria
  - Long-term effectiveness and permanence
  - Reduction of toxicity, mobility, or volume through treatment
  - Short-term effectiveness
  - Implementability
  - Cost
- Modifying Criteria
  - State acceptance
  - Community acceptance.

An alternative must meet the threshold criteria to be eligible for selection as the remedy. The primary balancing criteria represent the primary criteria upon which the analysis of alternatives is based taking into account technical, cost, institutional, and risk concerns. The preferred alternative may then be modified based upon the results of the state and community comments received during the public comment period on the DTSR. Table 11-5 presents a summary of the alternatives with respect to how they meet the threshold and primary balancing criteria.

11.3.1 Overall Protection of Human Health and the Environment

This evaluation criterion is used to measure how an alternative will achieve and maintain human health and environmental protectiveness. It assesses whether the risk posed to potential receptors is eliminated, reduced, or controlled through each pathway by natural recovery, treatment, engineering, or institutional controls.

The overall protectiveness of a candidate remedy must be considered in light of the results of the human health and environmental assessments (Sections 6 and 7). Risks associated with human and wildlife (bird and mammal) exposure to CoCs through sea-food consumption were determined to fall within acceptable limits when considered in the context of the conservative modeling assumptions. Thus, risk to humans and fish-eating birds and mammals is not a concern now or in the future. The primary hazard associated with the sediments in Ward Cove is from direct exposure of benthic organisms.

TABLE 11-5. SUMMARY OF DETAILED ANALYSIS OF ALTERNATIVES

	Alternative A1	Alternative A2	Alternative B	Alternative C	Alternative D	Alternative E
	No Action	Natural Recovery	Thin Cap, Dredge 12,300 cy, Dispose Upland[a]	Thin Cap, Dredge 80,000 cy, Dispose in Site 2 CAD[a]	Thin Cap, Dredge 175,000 cy, Dispose in Site 2 NCDF[a]	Thin Cap, Dredge 155,000 cy, Dispose in Site 1 NCDF[a]
Threshold Criteria						
Overall Protection of Human Health[b] and Environment	Sediments pose very limited hazard to environment. Natural recovery would likely occur to meet RAOs. Ability of tube-dwelling organisms to successfully colonize Ward Cove has been demonstrated by sediment toxicity tests. However, no monitoring would be conducted to verify recovery.	Same as Alternative A1 except monitoring would be conducted to verify recovery and ability to meet RAOs.	Sediments pose very limited hazard to environment. Natural recovery would likely occur to meet RAOs in uncapped areas. Ability of tube-dwelling organisms to successfully colonize Ward Cove has been demonstrated by sediment toxicity tests. Thin cap over portion of AOC would accelerate natural recovery. Dredging of small area would have minimal adverse impacts on environment, workers, and public.	Same as Alternative B except dredging larger volume, CAD construction, and log removal in areas to be dredged would have greater potential short-term adverse impacts on environment, workers, and public.	Same as Alternative C except dredging larger volume would have greater potential short-term adverse impacts on environment, workers, and public.	Same as Alternative D.
Compliance with ARARs	Will comply with ARARs.	Will comply with ARARs.	Will comply with ARARs.	Will comply with ARARs.	Will comply with ARARs.	Will comply with ARARs.
Primary Balancing Criteria						
Long-Term Effectiveness and Permanence	Would likely provide long-term protectiveness, but no monitoring would be conducted to verify it.	Would likely provide long-term protectiveness; monitoring would be conducted to verify it.	Same as Alternative A2.	Same as Alternative A2.	Same as Alternative A2.	Same as Alternative A2.
Reduction of Toxicity, Mobility, or Volume through Treatment	No treatment would occur.	Same as Alternative A1.	Same as Alternative A1.	Same as Alternative A1.	Same as Alternative A1.	Same as Alternative A1.
Short-Term Effectiveness	No additional risks to environment, workers, or public.	Minor safety hazards to workers during sampling.	Minimal risks to public. Construction related risks for remediation workers associated with working on water and with heavy equipment. Existing benthic communities would be largely eliminated by capping, but would recolonize. Water quality effects would need to be monitored during remediation.	Same as Alternative B except short-term risks would be greater because of larger volume of sediment dredged.	Same as Alternative C except short-term risks would be greater because of larger volume of sediment dredged.	Same as Alternative D. Short-term risks may be less if smaller volume of sediment is dredged.

TABLE 11-5. (cont.)

	Alternative A1	Alternative A2	Alternative B	Alternative C	Alternative D	Alternative E
	No Action	Natural Recovery	Thin Cap, Dredge 12,300 cy, Dispose Upland[a]	Thin Cap, Dredge 80,000 cy, Dispose in Site 2 CAD[a]	Thin Cap, Dredge 175,000 cy, Dispose in Site 2 NCDF[a]	Thin Cap, Dredge 155,000 cy, Dispose in Site 1 NCDF[a]
Implementability	No technologies are to be implemented.	No technologies are to be implemented.	Technically feasible to implement, but not for slopes steeper than 4H:1V and very high-density log area. A pilot study would be conducted to determine capping approach (thin layer vs. mounding), placement methods, and other implementability issues.	Same as Alternative B except removing larger quantity of sediment and constructing CAD would be more difficult to implement. Capping the CAD would be difficult. A special construction approach (building up the dikes to allow settling/dewatering, then partially removing the dikes) would be needed to facilitate capping CAD. Implementation would need to be coordinated with potential future development (e.g., a marina).	Appears to be technically feasible. Implementability same as Alternative B. Implementation would need to be coordinated with potential future development (e.g., a marina). After construction, NCDF could be used for storage or parking, but use for buildings would require pilings.	Appears to be technically feasible. Implementability same as Alternative B. Implementation would need to be coordinated with future use of KPC facility. After construction, NCDF could be used for storage or parking, but use for buildings would require pilings. Use of NCDF for log storage would require additional evaluation during design and could affect capacity of NCDF.
Cost (total present worth)	Minimal or none	$0.5 million	$4.5 million (KPC landfill) $5.6 million (Washington landfill)	$17 million	$33 million	$30 million

Note: AOC - area of concern
ARAR - applicable or relevant and appropriate requirement
CAD - confined aquatic disposal
cy - cubic yard
KPC - Ketchikan Pulp Company
NCDF - near-shore confined disposal facility
RAO - remedial action objective

[a] Alternative includes removal of logs in areas to be dredged.

[b] Sediments are within acceptable limits for human health.

Sediment toxicity to benthic organisms was rigorously evaluated during the ecological assessment (Section 7). Even within the boundary of the AOC, sediment toxicity was limited. The ability of tube-dwelling organisms to thrive in Ward Cove sediments was demonstrated by results of the sediment toxicity tests conducted in 1996 using the tube-dwelling polychaete *Neanthes* sp. and the tube-dwelling amphipod *Leptocheirus plumulosus*. Tests based on both of those species showed that none of the sediments from the 28 stations sampled throughout the Cove were toxic, including all stations within the AOC. Furthermore, the abundance of the opportunistic polychaete *Capitella capitata* in Ward Cove sediments (EVS 1992) indicates that a potentially important food source to certain benthic fishes was present in Ward Cove even in 1992, when the pulp mill was still active.

The RAOs for Ward Cove sediments are to reduce sediment toxicity, enhance recolonization of surface sediments, and provide an abundant and functioning benthic community that provides food to larger invertebrates and fish. In the following discussions detailing the evaluation of candidate alternatives, overall protection of human health and the environment is evaluated in terms of the ability of the candidate alternative to achieve RAOs.

11.3.2 Compliance with ARARs

Alternatives are assessed to determine whether they attain ARARs under federal and state environmental laws. A detailed description of ARARs and TBC criteria is provided in Appendix L.

11.3.3 Long-Term Effectiveness and Permanence

Alternatives are assessed for the long-term effectiveness and permanence they afford, along with the degree of certainty that the alternative will prove successful. The assessment includes the consideration of the magnitude of the residual risk remaining at the conclusion of the remedial activities and the adequacy and reliability of controls such as containment systems and institutional controls.

11.3.4 Reduction of Toxicity, Mobility, or Volume through Treatment

The degree to which alternatives employ treatment that reduces toxicity, mobility, or volume is assessed, including consideration of the amount of CoCs treated, the degree of expected reduction, the degree to which the treatment is irreversible, and the type and quantity of residuals that will remain following treatment.

11.3.5 Short-Term Effectiveness

This criterion addresses the short-term risks posed to the community during implementation of an alternative, the potential effects on workers during remedial action, the potential environment effects of the remedial action, and the time until protection is achieved.

11.3.6 Implementability

The ease or difficulty of implementing the alternatives is assessed by considering the technical and administrative feasibility and the availability of the materials and services required to implement the alternative. Administrative feasibility includes the ability and time required to obtain any necessary approvals from agencies.

11.3.7 Cost

This criterion addresses the costs associated with the alternative including direct capital costs (i.e., construction, equipment, land, services), indirect capital costs (i.e., engineering, contingency), long-term monitoring costs, operation and maintenance costs, and total net present value.

11.3.8 State Acceptance

This criterion addresses the state's concerns, if any, of the preferred alternative. The state's input is solicited during the comment period of the DTSR.

11.3.9 Community Acceptance

This criterion addresses the public's concerns, if any, of the preferred alternative. It will be addressed during the comment period of the DTSR and will not be further addressed in this report.

11.4 ALTERNATIVE A1—NO ACTION

The no action alternative would not implement any remedial actions or institutional controls. The site would remain as is and no environmental monitoring would be performed to monitor the natural recovery of the site. Accretion and natural recovery would continue at the site at present rates. It is included as a baseline to which the other alternatives can be compared.

11.4.1 Overall Protection of Human Health and the Environment (Alternative A1)

The no action alternative does not provide any additional protection for benthic organisms from sediment toxicity other than natural recovery. The natural recovery discussion in Section 11.5.1 is relevant to the no action alternative; however, natural recovery is not a specific objective of the no action alternative and its progress would not be monitored.

11.4.2 Compliance with ARARs (Alternative A1)

Alternative A1 will achieve compliance with ARARs. There are no chemical-specific ARARs for sediments. The provisions in the Alaska water quality standards that relate to sediment toxicity are very broad and, accordingly, they are not legally ARARs in establishing cleanup levels as defined in Section 121 of CERCLA (U.S. EPA 1998c). Washington State SQSs (which are TBC criteria) were considered during screening level evaluation of chemicals found in Ward Cove sediments to evaluate potential ecological effects. Other potential ARARs or TBC criteria that have been considered and will continue to be considered for Ward Cove sediments include EPA's contaminated sediment strategy and Ward Cove site-specific sediment quality values (U.S. EPA 1998c).

11.4.3 Long-Term Effectiveness and Permanence (Alternative A1)

This alternative would provide long-term reduction of contaminant levels by the process of natural recovery; however, natural recovery is not a specific objective of the no action alternative and would not be monitored.

11.4.4 Reduction of Toxicity, Mobility, or Volume through Treatment (Alternative A1)

No treatment would be performed under this alternative. The natural recovery of the site will provide a degree of reduction in toxicity and mobility in the biologically active zone (see discussion in Section 11.5.1).

11.4.5 Short-Term Effectiveness (Alternative A1)

Because no new construction activities would occur under this alternative, there would be no additional risks to workers or the public. Also, no additional risks to the environment would occur if this alternative were selected.

11.4.6 Implementability (Alternative A1)

There are no remedial actions to implement under the no action alternative.

11.4.7 Cost (Alternative A1)

There are only minimal costs, if any, associated with the no action alternative.

11.5 ALTERNATIVE A2—NATURAL RECOVERY

The natural recovery alternative depends on natural processes (e.g., sediment accumulation, mixing, chemical degradation and diffusion, benthic community succession) to achieve RAOs. Monitoring to confirm recovery is an important component of this alternative.

11.5.1 Overall Protection of Human Health and the Environment (Alternative A2)

Sediments in Ward Cove are currently within acceptable limits for human health and wildlife and are of limited toxicity to benthos. As discussed in Sections 6 and 7, risks to humans and fish-eating birds and mammals are not a concern now or in the foreseeable future. The primary hazard in Ward Cove is to benthic organisms from direct toxicity of sediments; however, even within the boundary of the AOC, sediment toxicity is limited. The ability of tube-dwelling organisms to thrive in Ward Cove sediments was demonstrated by results of the sediment toxicity tests conducted in 1996 using the tube-dwelling polychaete *Neanthes* sp. and the tube-dwelling amphipod *Leptocheirus plumulosus*, which exhibited no toxicity at any of the 28 stations sampled throughout the Cove. Furthermore, the abundance of the opportunistic polychaete *Capitella capitata* in sediment throughout Ward Cove (EVS 1992) indicates that a potentially important food source to certain benthic fishes was present in Ward Cove even when the pulp mill was active.

Natural recovery is a benign and effective way to achieve RAOs that has none of the disadvantages of sediment removal or capping. It must be recognized that a recovery period will be necessary to achieve an abundant and functioning benthic community that provides food to invertebrates and fish. The time period over which benthic macroinvertebrates in Ward Cove can be expected to recover naturally can be estimated from historical studies that have monitored benthic recovery in areas affected by dredged material disposal, deposition of sewage sludge, and deposition of pulp mill material. Those studies indicate that natural recovery of surface sediments may occur as quickly as 2–3 years or as slowly as 20 years or more (Section 9).

11.5.2 Compliance with ARARs (Alternative A2)

Alternative A2 will achieve compliance with ARARs. There are no chemical-specific ARARs for sediments. The provisions in the Alaska water quality standards that relate to sediment toxicity are very broad and, accordingly, they are not legally ARARs in establishing cleanup levels as defined in Section 121 of CERCLA (U.S. EPA 1998c). Washington State SQSs (which are TBC criteria) were considered during screening level

evaluation of chemicals found in Ward Cove sediments to evaluate potential ecological effects. (See discussion in Section 11.5.1 concerning protection of the environment.) Other potential ARARs or TBC criteria that have been considered and will continue to be considered for Ward Cove sediments include EPA's contaminated sediment strategy and Ward Cove site-specific sediment quality values (U.S. EPA 1998c).

11.5.3 Long-Term Effectiveness and Permanence (Alternative A2)

This alternative would likely provide long-term effectiveness. As previously discussed, the ability of tube-dwelling organisms to successfully colonize Ward Cove has been demonstrated by results of sediment toxicity tests. Monitoring would be conducted to verify long-term effectiveness and future protection of the environment.

11.5.4 Reduction of Toxicity, Mobility, or Volume through Treatment (Alternative A2)

No treatment would be performed under this alternative. The natural recovery of the site will provide a degree of reduction in toxicity and mobility in the biologically active zone.

11.5.5 Short-Term Effectiveness (Alternative A2)

Because no new construction activities would occur under this alternative, there would be no additional risks to workers or the public. Also, no additional risks to the environment would occur if this alternative were selected.

11.5.6 Implementability (Alternative A2)

There are no remedial actions to implement under the natural recovery alternative.

11.5.7 Cost (Alternative A2)

There are no capital costs associated with the natural recovery alternative. Costs for monitoring the recovery of the benthic community are approximately $120,000 per monitoring event. Assuming five events over 10 years and a 5 percent discount rate (after inflation), a present worth of $450,000 is calculated.

11.6 ALTERNATIVE B—THIN CAPPING WITH NAVIGATIONAL DREDGING AND UPLAND DISPOSAL

This alternative incorporates a thin cap (amend surface sediments) or creation of clean island surfaces over the portions of the AOC where thin capping (or mounding) is

practicable to enhance recovery. Areas with slopes steeper than 4H:1V and very high-density log areas would not be capped. A pilot study would be conducted to evaluate the other portions of the AOC where capping could be implemented, methods of placement, and other design considerations. For the purpose of this discussion, it is assumed that an average of 1 ft of capping material would be distributed over an area of 34–40 acres, depending on the post-dredging area requiring capping if native sediments are not reached during dredging. In addition, approximately 12,300 cy of sediment from a 3–6 acre area near the dock would be dredged. Logs located in the area to be dredged would be removed. Two options have been identified for disposal of the dredged material and logs. They would either be disposed in the landfill at the plant currently used for flyash disposal (Option 1) or at an alternative upland disposal site (Option 2).

11.6.1 Overall Protection of Human Health and the Environment (Alternative B)

Sediments in Ward Cove are currently within acceptable limits for human health and wildlife and are of limited toxicity to benthos. As discussed in Sections 6 and 7, risks to humans and fish-eating birds and mammals are not a concern now or in the foreseeable future. Toxicity to benthic organisms is the primary concern for Ward Cove sediment; however, even within the boundary of the AOC, sediment toxicity is limited. As discussed in Section 11.5.1, natural recovery is a benign and effective way to achieve RAOs over much of the AOC in a reasonable time frame. Natural recovery would be the preferred remedy for that portion of the AOC where thin capping was not practicable.

Application of a thin cap or clean island creation over a portion of the AOC would likely accelerate natural recovery processes for that portion of the AOC, reducing the concentrations of surface CoCs potentially associated with the limited sediment toxicity observed for Ward Cove sediments. Existing benthic communities would probably be largely eliminated in the capped areas immediately after the application of the cap or sand mounds, but would likely recolonize the capped or mounded area over a relatively short time period. If thin capping is feasible, it would effect accelerated natural recovery over a larger area (40 acres) than the mounding approach, which would necessarily have to be applied to a more limited area (21 acres in the scenario developed for this report).

Dredging would have a similar effect on benthic communities in the area targeted for sediment removal. Existing benthic communities would be eliminated immediately after the dredging, but would likely recolonize the area over a relatively short time period. The surface material in the dredged area would still be organic-rich sediment, but at a deeper elevation. Dredging and log removal would also have short-term adverse impacts on the water column, potentially resulting in brief and localized areas of oxygen depletion due to the resuspension and dispersion of fine-grained sediments with elevated BOD and COD.

11.6.2 Compliance with ARARs (Alternative B)

Alternative B will achieve compliance with ARARs, except for potential short-term water quality impacts. Measures would be taken during remediation to minimize water quality effects during log removal and dredging operations. Construction would likely be conducted within a designated limited time frame to minimize impacts on migrating fish. Alaska water quality standards may be ARARs for dredging or capping activities to ensure that those activities do not contribute to the long-term exceedance of water quality standards in the water column (U.S. EPA 1998c). The State of Alaska has identified the Alaska water quality standard for turbidity for marine waters as the only ARAR for cleanup actions in Ward Cove (Reges 1999, pers. comm.).

There are no chemical-specific ARARs for sediments. The provisions in the Alaska water quality standards that relate to sediment toxicity are very broad and, accordingly, they are not legally ARARs in establishing cleanup levels as defined in Section 121 of CERCLA (U.S. EPA 1998c). Washington State SQSs (which are TBC criteria) were considered during screening level evaluation of chemicals found in Ward Cove sediments to evaluate potential ecological effects. (See discussion in Section 11.5.1 concerning protection of the environment.) Other potential ARARs or TBC criteria that have been considered and will continue to be considered for Ward Cove sediments include EPA's contaminated sediment strategy and Ward Cove site-specific sediment quality values (U.S. EPA 1998c).

Measures would be taken to prevent spills or runoff associated with dewatering dredged sediments. Compliance with ARARs associated with uplands disposal in a landfill would be achieved. Workers who handle the contaminated dredged sediments would comply with all Occupational Safety and Health Administration (OSHA) health and safety requirements.

11.6.3 Long-Term Effectiveness and Permanence (Alternative B)

This alternative would provide long-term effectiveness. As previously discussed, the ability of tube-dwelling organisms to thrive in existing sediments in Ward Cove has been demonstrated by results of sediment toxicity tests. Therefore, colonization for the natural recovery areas would likely occur. Natural recovery is discussed and evaluated in more detail as a separate alternative (Alternative A2). For the portions of the AOC that receive clean surface sediment, natural recovery processes would be accelerated to achieve an abundant and functioning benthic community that provides food to invertebrates and fish. Under Option 1, logs and dredged material would be placed in the KPC landfill, which is equipped with a liner and leachate collection system. Use of this engineered landfill would ensure the long-term effectiveness and permanence of the remedy. Under Option 2, dredged material would be dewatered and taken to an approved offsite landfill for disposal. Monitoring of the AOC in the Cove would be conducted to verify long-term effectiveness and future protection of the environment.

11.6.4 Reduction of Toxicity, Mobility, or Volume through Treatment (Alternative B)

The dredged sediments would likely be dewatered prior to disposal in a landfill. Under Option 1, the sediments would be placed in the KPC flyash landfill. Under Option 2, the sediments could be mixed with a stabilizing agent prior to disposal in an approved offsite upland landfill, which would reduce the mobility of chemicals in the sediment.

11.6.5 Short-Term Effectiveness (Alternative B)

There are minimal risks to the public during the implementation of this alternative. Capping or dredging activities would not likely affect the public. There would be construction-related risks for remediation workers (e.g., potential safety hazards associated with working on the water and with heavy equipment). All remediation workers involved with activities associated with the log removal and dredged sediments would need to comply with OSHA health and safety regulations. Existing benthic communities would be affected by capping and by dredging. However the sand/organic material mixture at the sediment surface (i.e., mixture of capping material and existing organic sediment) or the mounds of capping material would allow for recolonization of opportunistic and secondary benthic communities that would likely occur over a relatively short time period. A laboratory and/or field study would be performed during the design phase to determine the most appropriate method of placing the cap material. Dredging, and to a more limited extent capping, would release some organic-rich sediment to the water column. However, as previously discussed, even within the boundary of the AOC, the sediment toxicity is limited. Because of the limited toxicity and because of the large volume of water in the Cove available to assimilate the released sediment, the short-term effects in the water column should be minimal. Some very short-term oxygen depletion may occur in the water column due to oxidation of reduced compounds. Water quality protection measures and monitoring would need to be implemented during remediation to ensure that the potential effects to the environment would be minimal.

11.6.6 Implementability (Alternative B)

In general, the technologies employed are commonly used and proven to be reliable and the required equipment is readily available. Thin layer caps have been successfully constructed in the past (i.e., Pier 64/65 in Seattle, Washington, and Eagle Harbor, Washington). However, it is uncertain as to whether the soft organic-rich sediments in Ward Cove can support a thin cap or amendment to the surface layer. A laboratory and/or field pilot test would need to be performed during the design phase to determine if thin capping would effectively amend the surface sediments and to evaluate the thickness of organic-rich sediments that could support the cap. The water depth at which thin capping could be conducted could also be examined during the pilot study (i.e., whether thin capping could be conducted below water depths of 120 ft), as well as other design issues such as placement method.

Mounding of capping material to create a discontinuous, island-like cover would be technically implementable. This type of enhanced recovery would be more readily implementable in areas where the soft sediment is less than 5 ft thick, where 5–6 ft mounds could displace the organic sediment. Mounding of cap material in areas where the organic-rich sediment is greater than 5 ft thick would be technically feasible; however, it would be very costly because of the amount of cap material needed to create emergent mounds.

11.6.7 Cost (Alternative B)

The estimated total cost for Alternative B is approximately $4.5 million for Option 1 and approximately $5.6 million for Option 2, as shown in Tables 11-1 and 11-2, respectively. These costs are based on 40 acres, which is the maximum area that would required capping. Monitoring costs during construction activities are included in these amounts. Long-term monitoring costs are estimated at approximately $120,000 per event. Assuming five events over 10 years and a discount rate of 5 percent (after inflation), a present worth of $450,000 is calculated. Monitoring costs at the landfills are not included, because these costs are typically included in the tipping fee, and are incurred by the landfill operator.

11.7 ALTERNATIVE C—THIN CAPPING WITH DREDGING AND DISPOSAL IN A SHALLOW SUBTIDAL CAD AT SITE 2

This alternative is similar to Alternative B, except that a greater amount of sediments (approximately 80,000 cy, or approximately 7.5 acres) would be dredged. Logs would be removed from the dredged area and disposed. Dredged sediments would be disposed in an engineered CAD site constructed at Site 2 (Figure 11-3). No sediments would be disposed in the upland landfill discussed in Alternative B. The top of the CAD site would have an elevation of approximately –2 to –5 ft MLLW after capping and would be covered with rocks for armor, which would allow for shallow draft barges and small boat traffic.

11.7.1 Overall Protection of Human Health and the Environment (Alternative C)

Sediments in Ward Cove are currently within acceptable limits for human health and wildlife and are of limited toxicity to benthos. As discussed in Sections 6 and 7, risks to humans and fish-eating birds and mammals are not a concern now or in the foreseeable future. Toxicity to benthic organisms is the primary concern for Ward Cove sediment; however, even within the boundary of the AOC, sediment toxicity is limited. As discussed in Section 11.5.1, natural recovery is a benign and effective way to achieve RAOs in a reasonable time frame. Natural recovery would be the preferred remedy for that portion of the AOC where capping is not anticipated to be practicable.

Application of a thin cap or mounds interspersed over a portion of the AOC would accelerate natural recovery processes for that portion of the AOC, reducing the concentrations of CoCs potentially associated with the limited sediment toxicity observed for Ward Cove sediments. Existing benthic communities would be largely eliminated immediately after the application of the cap, but would likely recolonize the capped or mounded area over a relatively short time period.

Approximately 80,000 cy (the capacity of CAD Site 2) would be dredged under this alternative. Dredging would be conducted to leave minimal residual problem sediment where performed. Total removal is not expected because of the limitation of dredge equipment and the characteristics of the problem sediment. Dredging would also have short-term adverse impacts on the water column and associated organisms, potentially resulting in brief and localized areas of oxygen depletion due to the resuspension and dispersion of fine-grained sediments with elevated BOD and COD.

Disposal of sediment at Site 2 would eliminate the local benthic community in that area. Site 2 is outside of the AOC; therefore, an unaffected benthic community would be eliminated immediately following disposal and capping at Site 2. It is likely that benthic organisms would recolonize the disposal area over a relatively short time period.

11.7.2 Compliance with ARARs (Alternative C)

Alternative C will achieve compliance with ARARs. No long-term exceedances of water quality standards are anticipated; however, short-term localized exceedances are possible with this alternative during dredging and log removal activities. Measures would be taken during remediation to minimize water quality effects. Construction would be conducted within a designated time frame to minimize impacts on migrating fish. Alaska water quality standards may be ARARs for dredging or capping activities to ensure that those activities do not contribute to the long-term exceedance of water quality standards in the water column (U.S. EPA 1998c). The State of Alaska has identified the Alaska water quality standard for turbidity for marine waters as the only ARAR for cleanup actions in Ward Cove (Reges 1999, pers. comm.).

There are no chemical-specific ARARs for sediments. The provisions in the Alaska water quality standards that relate to sediment toxicity are very broad and, accordingly, they are not legally ARARs in establishing cleanup levels as defined in Section 121 of CERCLA (U.S. EPA 1998c). Washington State SQSs (which are TBC criteria) were considered during screening level evaluation of chemicals found in Ward Cove sediments to evaluate potential ecological effects (Section 11.5.1). Other potential ARARs or TBC criteria that have been considered and will continue to be considered for Ward Cove sediments include EPA's contaminated sediment strategy and Ward Cove site-specific sediment quality values (U.S. EPA 1998c).

11.7.3 Long-Term Effectiveness (Alternative C)

This alternative would provide long-term effectiveness. As previously discussed, the ability of tube-dwelling organisms to thrive in the existing sediments in Ward Cove has been demonstrated by results of sediment toxicity tests. Therefore, colonization for the natural recovery areas would likely occur. Natural recovery is discussed and evaluated in more detail as a separate alternative (Alternative A2). For the portions of the AOC that receive a thin cap (or mounds), natural recovery processes would be accelerated to achieve an abundant and functioning benthic community that provides food to invertebrates and fish. The sediment that is dredged would be disposed of in an engineered CAD, where it would be permanently retained. The type and extent of projected benthic recolonization of the dredged area would need to be examined further during design. Monitoring of the AOC in the Cove would be conducted to verify long-term effectiveness and future protection of the environment.

11.7.4 Reduction of Toxicity, Mobility, or Volume through Treatment (Alternative C)

No treatment would be performed under this alternative.

11.7.5 Short-Term Effectiveness (Alternative C)

Under this alternative, there would be construction-related risks for remediation workers (e.g., potential safety hazards associated with working on the water and with heavy equipment). All remediation workers involved with activities associated with handling sediments would need to comply with OSHA health and safety regulations. Potential effects of the dredged sediments during remediation on the public would need to be evaluated further during design. Existing benthic communities would be affected by capping, dredging, and construction of the CAD but would likely recolonize the area over a relatively short period. A laboratory and/or field study would be performed during the design phase to determine the most appropriate method of placing the cap material. The type and extent of projected benthic recolonization of the dredged area would need to be examined further during design. Dredging, and to a more limited extent capping, would release some organic-rich sediment to the water column. However, as previously discussed, even within the boundary of the AOC, the sediment toxicity is limited. Because of the limited toxicity and because of the large volume of water in the Cove available to assimilate the released sediment, the short-term effects in the water column should be minimal. Some very short-term oxygen depletion may occur in the water column due to oxidation of reduced compounds. Water quality protection measures and monitoring would need to be implemented during remediation to ensure that the potential effects to the environment would be minimal. Special procedures and equipment may need to be used to reduce the resuspension of sediments (i.e., slower production rates, removal of only those logs with no portion buried in the sediments). Log removal operations may disturb and resuspend sediments.

11.7.6 Implementability (Alternative C)

This alternative is technically implementable, given that the technologies employed are commonly used and proven to be reliable and the required equipment is readily available. Capping of the CAD site would be difficult but possible. After sediments are placed in the CAD, it is expected that they will initially have even less strength than the *in situ* sediments. A special construction approach (building up the dikes to allow settling and dewatering of the dredged sediment, then partially removing the dikes) would have to be used to facilitate capping the CAD. Construction and filling of a CAD site could occur at all stages of tide. However, use of bottom dump barges may not be feasible during periods of high currents.

As discussed for Alternative B, thin layer caps have been successfully constructed in the past (i.e., Pier 64/65 in Seattle, Washington, and Eagle Harbor, Washington). However, the degree to which the soft organic-rich sediments in Ward Cove can support a cap or amendment to the surface layer is uncertain. A pilot study would need to be performed during the design phase to determine if thin capping would effectively amend the surface sediments, to evaluate the thickness of organic-rich sediments that could support the cap, and to evaluate other design parameters. Mounding of capping material to create a discontinuous, island-like cover would be technically implementable.

11.7.7 Cost (Alternative C)

This alternative is expected to cost approximately $17 million to construct, as shown in Table 11-3. Monitoring costs during construction activities are included in this amount. Long-term monitoring costs are estimated at approximately $120,000 per event. Assuming five events in 10 years and a discount rate of 5 percent (after inflation), a present worth of $450,000 is calculated.

11.8 ALTERNATIVE D—THIN CAPPING WITH DREDGING AND DISPOSAL IN A NEAR-SHORE CONFINED DISPOSAL FACILITY AT SITE 2

This alternative is similar to Alternative C, except that a greater amount of sediments (approximately 175,000 cy or approximately 13 acres) would be dredged and the disposal site would be constructed to an elevation of approximately +18 ft MLLW. Logs would be removed from the dredged area and disposed. Dredged sediments would be placed to an elevation of +7 ft MLLW in an engineered NCDF site constructed at Site 2 (Figure 11-3). The sediments in the NCDF site would be covered with sand, and the top of the sand cap would have an elevation of approximately +18 ft (the cost estimate does not include costs for any gravel or asphalt for the final cover). The site could be used for open storage or parking after it stabilized. Buildings could be constructed over the fill, but pile support would be required because the underlying organic sediment would be expected to continue to settle for several years.

11.8.1 Overall Protection of Human Health and the Environment (Alternative D)

Sediments in Ward Cove are currently within acceptable limits for human health and wildlife and are of limited toxicity to benthos. As discussed in Sections 6 and 7, risks to humans and fish-eating birds and mammals are not a concern now or in the foreseeable future. Toxicity to benthic organisms is the primary concern for Ward Cove sediment; however, even within the boundary of the AOC, sediment toxicity is limited. As discussed in Section 11.5.1, natural recovery is a benign and effective way to achieve RAOs in a reasonable time frame. Natural recovery would be the preferred remedy for that portion of the AOC where capping is not anticipated to be practicable.

Application of a thin cap or mounds interspersed over a portion of the AOC would accelerate natural recovery processes for that portion of the AOC, reducing the concentrations of CoCs potentially associated with the limited sediment toxicity observed for Ward Cove sediments. Existing benthic communities would be largely eliminated immediately after the application of the cap, but would likely recolonize the capped area over a relatively short time period.

Approximately 175,000 cy (the capacity of NCDF Site 2) would be dredged under this alternative. Dredging would be conducted to leave minimal residual problem sediment where performed. Total removal is not expected because of the limitation of dredge equipment and the characteristics of the problem sediment. Dredging would have short-term adverse impacts on the water column and associated organisms, potentially resulting in brief and localized areas of oxygen depletion due to the resuspension and dispersion of fine-grained sediments with elevated BOD and COD.

Disposal of sediment at Site 2 would eliminate the local benthic community in that area. Site 2 is outside of the AOC; therefore, an unaffected benthic community would be eliminated immediately following disposal and capping at Site 2.

11.8.2 Compliance with ARARs (Alternative D)

Alternative D will achieve compliance with ARARs. No exceedances of water quality standards are anticipated; however, localized exceedances are possible with this alternative during dredging and log removal activities. Measures would be taken during remediation to minimize water quality effects. Construction would be conducted within a designated time frame to minimize impacts on migrating fish. Alaska water quality standards may be ARARs for dredging or capping activities to ensure that those activities do not contribute to the long-term exceedance of water quality standards in the water column (U.S. EPA 1998c). The State of Alaska has identified the Alaska water quality standard for turbidity for marine waters as the only ARAR for cleanup actions in Ward Cove (Reges 1999, pers. comm.).

There are no chemical-specific ARARs for sediments. The provisions in the Alaska water quality standards that relate to sediment toxicity are very broad and, accordingly,

they are not legally ARARs in establishing cleanup levels as defined in Section 121 of CERCLA (U.S. EPA 1998c). Washington State SQSs (which are TBC criteria) were considered during screening level evaluation of chemicals found in Ward Cove sediments to evaluate potential ecological effects (Section 11.5.1). Other potential ARARs or TBC criteria that have been considered and will continue to be considered for Ward Cove sediments include EPA's contaminated sediment strategy and Ward Cove site-specific sediment quality values (U.S. EPA 1998c).

11.8.3 Long-Term Effectiveness (Alternative D)

This alternative would provide long-term effectiveness. As previously discussed, the ability of tube-dwelling organisms to thrive in the existing sediments in Ward Cove has been demonstrated by results of sediment toxicity tests. Therefore, colonization for the natural recovery areas would likely occur. Natural recovery is discussed and evaluated in more detail as a separate alternative (Alternative A2). For the portions of the AOC that receive a thin cap (or mounding), natural recovery processes would be accelerated to achieve an abundant and functioning benthic community that provides food to invertebrates and fish. The sediment that is dredged would be disposed of in an engineered NCDF, where it would be permanently retained. The type and extent of projected benthic recolonization of the dredged area would need to be examined further during design. Monitoring of the AOC in the Cove would be conducted to verify long-term effectiveness and future protection of the environment.

11.8.4 Reduction of Toxicity, Mobility, or Volume through Treatment (Alternative D)

No treatment would be performed under this alternative.

11.8.5 Short-Term Effectiveness (Alternative D)

Under this alternative, there would be construction-related risks for remediation workers (e.g., potential safety hazards associated with working on the water and with heavy equipment). All remediation workers involved with activities associated with handling sediments would need to comply with OSHA health and safety regulations. Potential effects of the dredged sediments during remediation on the public would need to be evaluated further during design. Existing benthic communities would be affected by capping, dredging, and construction of the NCDF but would likely recolonize the dredged area over a relatively short time period. A laboratory and/or field study would be performed during the design phase to determine the most appropriate method of placing the cap material. The type and extent of projected benthic recolonization of the dredged area would need to be examined further during design. Dredging, and to a more limited extent capping, would release some organic-rich sediment to the water column. However, as previously discussed, even within the boundary of the AOC, the sediment toxicity is limited. Because of the limited toxicity and because of the large volume of water in the

Cove available to assimilate the released sediment, the short-term effects in the water column should be minimal. Some very short-term oxygen depletion may occur in the water column due to oxidation of reduced compounds. Water quality protection measures and monitoring would need to be implemented during remediation to ensure that the potential effects to the environment would be minimal. Special procedures and equipment may need to be used to reduce the resuspension of sediments (i.e., slower production rates, removal of only those logs with no portion buried in the sediments). Log removal operations may disturb and resuspend sediments.

11.8.6 Implementability (Alternative D)

This alternative is technically implementable, given that the technologies employed are commonly used and proven to be reliable and the required equipment is readily available. NCDFs have been successfully constructed at several sites in the Pacific Northwest (i.e., Terminal 91, Seattle, Milwaukee Waterway, Tacoma and Port of Everett). The dredged sediment in the disposal site would be very soft and difficult to cover. The cover sand would be placed using techniques for building over very soft peat or very soft soil. After the disposal site is dewatered, sand would be placed over the sediment slowly and carefully, starting at the perimeter berms. One capping method for the NCDF would be to place individual clamshell buckets of sand in a row away from the berm and then add sand in the space between the berm and initial row of sand. This procedure would be repeated to work gradually toward the center of the site. Once the initial layer was placed, additional material could be placed with "low-ground pressure" dozers. Geotextile fabric could also be used to increase the bearing capacity of the very soft organic sediment.

As discussed for Alternative B, thin layer caps have been constructed in the past (i.e., Pier 64/65 in Seattle, Washington, and Eagle Harbor, Washington). However, it is uncertain as to whether the soft organic-rich sediments in Ward Cove can support any type of cap or amendment in the surface layer. A pilot study would need to be performed during the design phase to indicate whether thin capping may effectively amend the surface sediments, to evaluate the thickness of organic-rich sediments that could support the cap, and to evaluate other design parameters. Mounding of capping material to create a discontinuous, island-like cover would be technically implementable.

11.8.7 Cost (Alternative D)

The estimated total cost for Alternative D is approximately $33 million, as shown in Table 11-4. Monitoring costs during construction activities are included in this amount. Long-term monitoring costs are estimated at approximately $120,000 per event. Assuming five events in 10 years and a discount rate of 5 percent (after inflation), a present worth of $450,000 is calculated.

11.9 ALTERNATIVE E—THIN CAPPING WITH DREDGING AND DISPOSAL IN A NEAR-SHORE CONFINED DISPOSAL FACILITY AT SITE 1

This alternative is very similar to Alternative D, except the location of the NCDF will be at Site 1. The bottom of the NCDF would be located between –20 and –30 ft MLLW.

The evaluation for this alternative is essentially the same as for Alternative D. The end use of the NCDF would affect how it is constructed. Special design considerations would be evaluated further during the design phase if the NCDF is to be used for log storage or heavy industrial purposes. The NCDF may not support log storage or heavy industrial use in the short-term unless only a limited quantity of problem sediment is disposed in it. The cost estimate includes sand as the final cover; costs for a gravel or asphalt cover would be additional. KPC operations (e.g., relocating the sawmill log lift and docking cargo vessels) would also need to be addressed during the design phase.

The estimated total cost for Alternative E is approximately $30 million, as shown in Table 11-6. A pilot study is recommended during design of this alternative to establish preferred construction methods and costs. Monitoring costs during construction activities are included in this amount. Long-term monitoring costs are estimated at approximately $120,000 per event. Assuming five events over a 10-year period and a discount rate of 5 percent (after inflation), a present worth of $450,000 is calculated.

11.10 RECOMMENDED ALTERNATIVE

Both Alternatives A2 and B will achieve RAOs, but over different time periods and at different costs. Natural recovery (Alternative A2) is less expensive, but slower. Thin capping or island mounds (Alternative B; also known as enhanced recovery) is more expensive, but quicker. A conceptual view of the varying effects of the thin capping/ enhanced recovery (Alternative B) and natural recovery (Alternative A2) on the benthic community over time is shown in Figure 11-5. Alternative B is expected to achieve a more advanced stage of benthic recolonization over a shorter period (Stage E); however, the existing benthic community will be impacted upon placement of the thin cap (Stage B).

Both alternatives are particularly suitable for the type of problem sediment present in Ward Cove, which has limited toxicity and does not contain persistent chemicals that are highly toxic or that have the potential to bioaccumulate. The applicability of thin capping, or island mounding to the AOC is limited by physical constraints within Ward Cove (i.e., steep slopes along portions of the shoreline) and by the physical properties of Ward Cove sediments (i.e., where the soft, organic-rich layer is thick).

Alternative B, which involves a combination of thin capping, natural recovery, and limited dredging, is the recommended alternative. The thin cap would be placed on approximately 34–40 acres of the AOC (Figure 11-6), depending on the post-dredging area requiring capping if native sediments are not reached during dredging. If thin capping is proven infeasible or ineffective during pilot-scale laboratory testing, island

TABLE 11-6. COST ESTIMATE FOR ALTERNATIVE E

Item	Quantity	Unit Cost	Cost
Construction Costs			
Placement of cap sand (27 acres)	43,560 cy	$6.90	$300,564
Delivery of sand to dockside	43,560 cy	$25.00	$1,089,000
Dredging	152,475 cy	$13.83	$2,109,238
Placement of sediment in NCDF	152,475 cy	$11.50	$1,753,463
Placement of berm gravel	183,000 cy	$27.67	$5,063,000
Purchase of berm gravel	183,000 cy	$8.00	$1,464,000
Delivery of berm gravel	183,000 cy	$29.25	$5,352,750
Placement of NCDF cover	83,631 cy	$11.50	$961,751
Purchase and delivery of sand	83,631 cy	$25.00	$2,090,764
Off-loading of logs	1,400 tons	$15.76	$22,064
Chipping of logs at KPC	1,400 tons	$15.00	$21,000
Mobilization	1 lump sum	$100,000	$100,000
Field overhead	8 months	$15,000	$120,000
Water quality monitoring	180 days	$1,500	$270,000
Construction Cost			$20,717,593
Contingency	30 percent		$6,215,278
Construction Estimate			$26,932,871

Summary	Direct Costs	Percentage	Cost		
Cap 27 acres	$1,389,564	6.9	$1,850,193	**Cap Unit Cost**	$ 54,417 per acre
Dredge and place 160,000 cy	$3,862,700	19.1	$5,143,153		
Construct NCDF berm and cover	$14,975,329	74.0	$19,939,526	**NCDF Unit Cost**	$ 164.50 per cy
Sum	$20,227,593	100.0	$26,932,871		

Non-Construction Costs			
Capping pilot study	1 lump sum	$200,000	$200,000
Design	5 percent of construction		$1,346,644
Capping/dredging monitoring	40 days	$3,000	$120,000
Construction management	2.5 percent of construction		$673,322
Non-Construction Estimate			$2,339,965
Total Estimated Capital Costs			$29,272,837
Periodic Monitoring Costs			
Monitoring every other year for 10 years	5 events	$120,000	
Present worth of 10 years monitoring			$450,000
Total Estimated Costs			$29,700,000

Note: cy - cubic yard
KPC - Ketchikan Pulp Company
NCDF - near-shore confined disposal facility

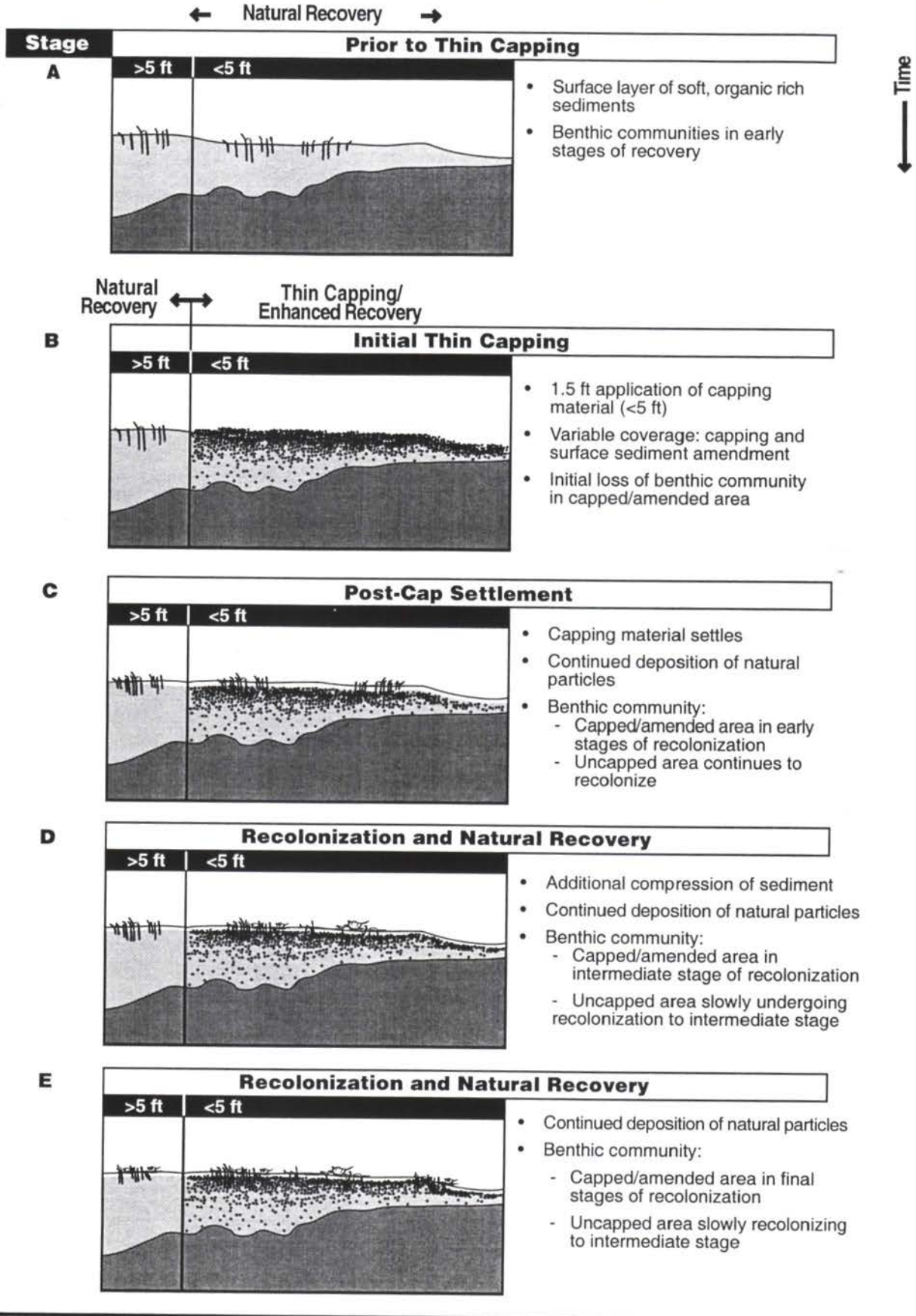

Figure 11-5. Enhanced recovery processes.

mounding would be applied to approximately 21 acres of the AOC. A pilot study would be required to determine the best placement method for the cap material and to determine the maximum thickness of the existing soft sediments that could be capped and still result in the desired results of a surface clean cover for satisfactory recover or whether mounding is the appropriate capping approach. Natural recovery would be in effect for the remainder of the AOC. Limited dredging of the sediments in the vicinity of KPC's main dock would also be conducted under this alternative because a cap could not be placed in this portion of the AOC without affecting navigation. The dredged sediments would be disposed at an upland landfill that is authorized to accept the material. Thin capping would be conducted after navigational dredging unless native sediments are reached during dredging. Recolonization of the benthic community would occur in those areas that are amended (thin capped). For those areas where thin capping is found to be unsuccessful in Ward Cove, it is anticipated that the remedy will be natural recovery.

The alternatives that involve extensive dredging (Alternatives C, D, and E) would also likely meet RAOs, but would be difficult to implement because of the high water content and very soft, fine-grained nature of the sediments. In addition, the incremental costs for Alternatives C, D, and E (compared to Alternative B) are disproportionate to their incremental benefits. There would be little or no gain in overall environmental benefits to the Cove for the additional actions and costs incurred.

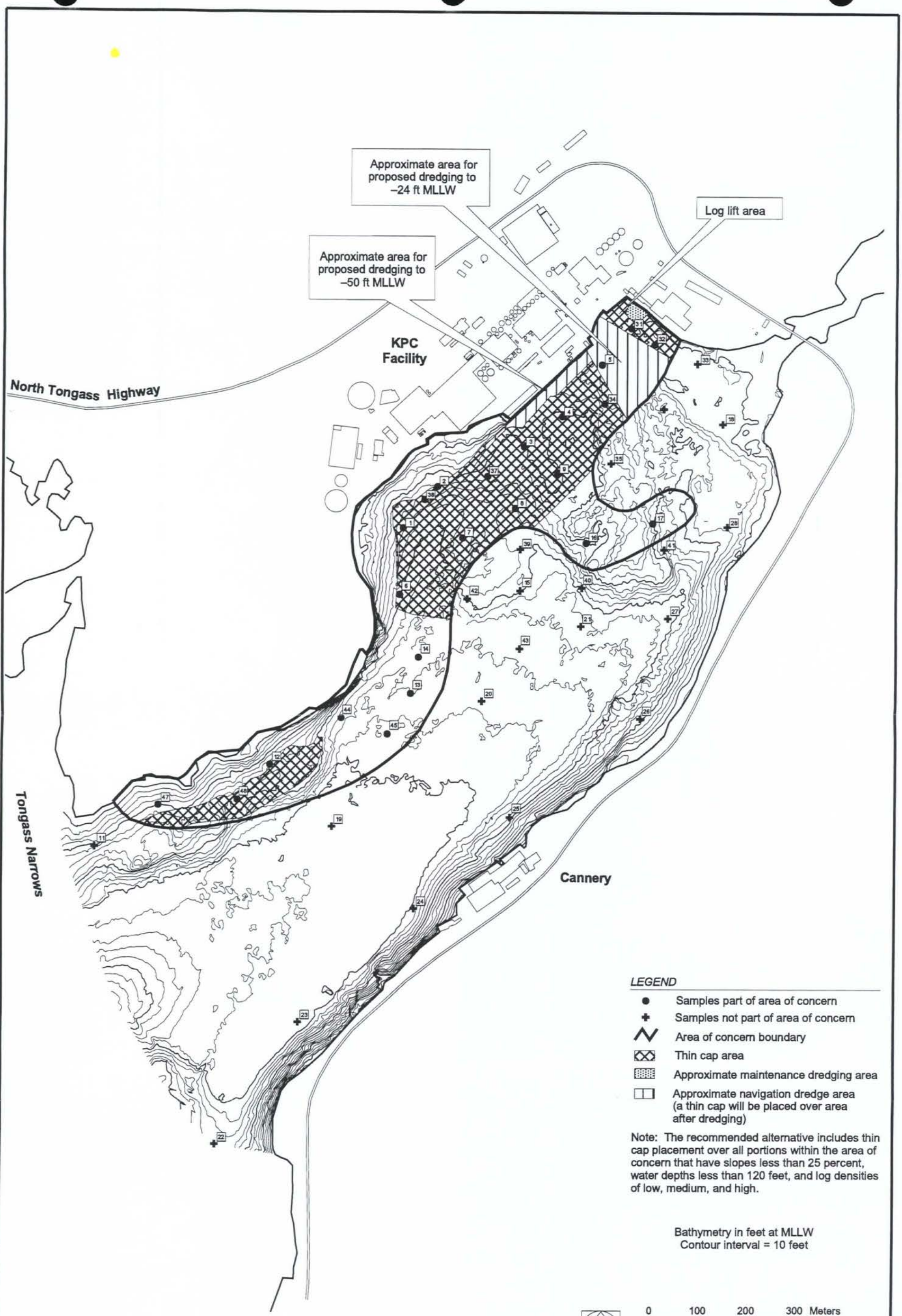

Figure 11-6. Recommended alternative: Alternative B, thin capping with navigational dredging and upland disposal.

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