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Contaminated Sediments Criteria Report

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ENVIRONMENTAL SERVICES

CONTAMINATED SEDIMENTS CRITERIA REPORT

For

Washington Department of Ecology
Olympia, Washington

Ecology Contract No. C0088171
PTI Contract No. C809

April 1989

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CONTAMINATED SEDIMENTS CRITERIA REPORT

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EXECUTIVE SUMMARY

The Puget Sound Water Quality Authority recently specified a series of activities to control sediment contamination in Puget Sound. These activities were set forth in the 1987 Puget Sound Water Quality Management Plan (the Plan, revised in 1989). In response to Element P-2 of the Plan, the Washington Department of Ecology (Ecology) is developing regulatory standards for identifying and designating sediments that have observable acute or chronic effects on biological resources or pose a significant health risk to humans.

The present report describes issues, alternatives, and recommendations relative to the adoption of criteria for classifying sediments in Puget Sound. Technical analyses conducted for this report included review of issues concerning application of chemical-specific sediment standards and recommended approaches for biological testing and assessment of human health effects from contaminated sediments.

RECOMMENDED APPROACH FOR SEDIMENT QUALITY VALUES

The Apparent Effects Threshold (AET) approach was selected by Ecology as the currently preferred method for developing sediment quality standards that address adverse biological effects in Puget Sound. Eight alternative approaches were evaluated in the present report. These other approaches should be considered for future use as they are developed or tested, including approaches that directly assess human health concerns. The primary reason for selecting the AET approach was its relatively high reliability in classifying Puget Sound samples as "impacted" or "not impacted." The reliability of the AET has been assessed using a large database comprising samples from 13 Puget Sound embayments (all biological indicators were not available in all embayments). In at least 85 percent of the available samples for each biological indicator, the approach either correctly classifies as "impacted" samples that exhibit adverse biological effects or correctly classifies as "not impacted" samples that do not exhibit adverse biological effects. In addition to its reliability in classifying sediments, the AET approach can be used to provide sediment quality values for the greatest number and widest range of chemicals of concern in Puget Sound. The approach also incorporates the widest range of biological indicators that are directly applicable to sediment conditions.

SUITABILITY OF AET TO INTERTIDAL AND LOW-SALINITY ENVIRONMENTS

Because existing AET are based primarily on samples from marine subtidal areas of Puget Sound, their applicability to other kinds of habitats (i.e., intertidal and low-salinity) found in the Sound were evaluated. AET are recommended for use at intertidal stations in the Sound based on preliminary reliability studies. Ongoing review of any additional verification data is recommended. Development of separate standards for intertidal sediments is not recommended as an efficient regulatory approach based on existing reliability results and the small areas relative to subtidal sediments in Puget Sound. Almost no information exists on the reliability of AET when they are applied to low-salinity environments. Until verification studies have been conducted, application in low-salinity environments is not recommended.

ROLE OF BIOLOGICAL TESTING IN DEVELOPING SEDIMENT QUALITY STANDARDS

Element P-2 of the 1987 Plan specifies that sediment quality standards be developed for identifying and designating sediments having observable acute or chronic adverse effects on biological resources. The Plan also specifies that these standards may use physical, chemical, and biological tests with clearly identified pass/fail criteria. Chemical criteria can identify sediment

chemical concentrations below which adverse biological effects would rarely be expected; however, biological testing is useful for confirming that biological effects are induced by sediments having chemical concentrations above the chemical criteria. Eight sediment bioassays and two measures of *in situ* effects on indigenous biota (i.e., bioaccumulation, reductions in the abundance of benthic macroinvertebrates) were evaluated for use in confirming the toxicity of Puget Sound sediments with concentrations exceeding chemical criteria.

To develop a classification scheme for identifying and designating sediments having observable acute or chronic adverse biological effects, it is recommended that biological testing be integrated with chemical criteria. The key elements of the classification scheme are:

- Chemical criteria to identify sediments having a very low potential for causing adverse acute and chronic biological effects

- Biological testing using acute and chronic sediment bioassays and effects on indigenous organisms to confirm that sediments with chemical concentrations above chemical criteria have acute and/or chronic effects, and to classify the sediments in accordance with the observed biological results.

It is recommended that chemical criteria be based on Puget Sound AET, which have been tested in this region. Because there is little information regarding chronic effects in Puget Sound, it is recommended that chemical criteria initially be developed using the Puget Sound AET based on benthic macroinvertebrate assemblages (i.e., an indicator that includes both chronic and acute effects on indigenous organisms), and an acute-to-chronic ratio of 10 relative to the acute bioassays used to develop Puget Sound AET (i.e., one-tenth of the highest AET for amphipod mortality, oyster larvae abnormality, and Microtox tests). The lower limit for chemical criteria should be set at the 90th percentile of chemical concentrations from Puget Sound reference areas. Confirmation of adverse effects should be conducted using the recommended sediment bioassays and effects on benthic macroinvertebrate assemblages.

The proposed classification scheme will classify sediments in Puget Sound as having or not having adverse effects on biological resources, and identify those sediments with adverse effects according to whether the effects are chronic or acute. In doing so, the classification scheme will facilitate sediment management activities in the sound. The use of chemical criteria as an integral part of the scheme will focus biological testing on the sediments having the greatest potential for causing adverse effects.

DEVELOPMENT OF HUMAN HEALTH CRITERIA

The recommended approach for human health assessment consists of a combination of:

- **Qualitative Hazard Assessment**—Evaluation of the potential toxicity of chemicals measured in sediments based on available data and literature reviews of health effects

- **Quantitative Risk Assessment**—Modeling of potential human health risks resulting from consumption of fish or shellfish associated with contaminated sediments to derive sediment standards

- **Sediment-Specific Hazard Assessment**—Short-term bioassays (e.g., Ames test) of sediments or, preferably, tissues of organisms exposed to specific sediments.

Implementation of human health criteria as part of the sediment quality standards requires further development of available hazard and risk assessment techniques. Thus, implementation of human health guidelines is not recommended at present. The developmental status and needs for each element of the recommended approach are summarized below.

A quantitative risk assessment model based on food-chain bioaccumulation of contaminants is recommended to address the question of how much risk may be associated with contaminated sediments. For example, by assuming that humans consume selected indicator species of seafood from Puget Sound at specified rates, the U.S. Environmental Protection Agency (EPA) risk assessment approach can be used to develop guidelines for chemical concentrations in edible tissues of fish or shellfish. Using a model of interactions between sediments and fish/shellfish, the tissue quality guidelines would then be extrapolated to develop sediment quality standards (i.e., screening values). These standards would represent the concentrations corresponding to tolerable levels of risk (or exposure) selected by regulatory management policy. In the near term, available data and equilibrium partitioning models may be used to develop very conservative (i.e., protective) screening values for concentrations of contaminants in sediments. When the sediment criteria are exceeded, a potential for adverse human health effects should be confirmed by using a laboratory test of bioaccumulation of contaminants in clams. Protocols for bioaccumulation tests using sediments are being developed by EPA. The concentrations of contaminants in clam tissue would be compared directly with the tissue quality criteria developed earlier. Exceedance of the tissue quality criteria would lead to classification of the sediment as potentially having adverse human health effects.

Further refinement of quantitative approaches for setting human health criteria is recommended. For example, models for relating sediment contamination to fish or shellfish contamination need to be developed further. The preferred approach involves development of empirical relationships between chemical concentrations in sediments and in fish/shellfish tissue. Extensive sampling and analysis may be required to obtain sufficiently precise relationships for a variety of chemicals in different sediment matrices (e.g., sediments differing in total organic carbon concentration and grain size). Development of equilibrium partitioning models to predict tissue concentrations of contaminants from corresponding concentrations in sediments is possible, but laboratory and field validation of models is needed for at least key organic contaminants (e.g., PAH, PCBs, DDT and related metabolites).

Finally, short-term bioassays (e.g., Ames test) of tissue extracts are recommended to evaluate the mutagenic and carcinogenic potential of chemical mixtures and single chemicals that are not analyzed for. These short-term tests complement the quantitative risk assessment modeling approach. Methods are available for analysis of tissue extracts using the Ames test, but problems in measuring the mutagenic potential of complex mixtures of contaminants require further research. Calibration of short-term bioassays to various concentrations of known mutagens in tissue samples is also needed. Consequently, the Ames test or another short-term bioassay will likely require substantial development before possibly being applied to the sediment quality standards. When data on the performance of the forthcoming Microtox mutagenicity bioassay become available, this mutagenicity test should be evaluated for use as part of the sediment quality standards.

INTEGRATION OF BIOLOGICAL RESOURCES AND HUMAN HEALTH ASSESSMENTS

Several options were considered for integrating the assessments of biological resources and human health in the development of sediment quality standards. It is recommended that sediments be evaluated separately with respect to each category of adverse effects. The evaluations should be parallel but independent, so that results from one category do not influence results from the other category. The independence of the two evaluations recognizes their uniqueness.

TECHNICAL SUMMARY

Sediment in several areas of Puget Sound is contaminated with potentially toxic substances such as petroleum-derived compounds, chlorinated organic compounds, and metals. Sources of these contaminants include runoff from urban streets, industrial discharges, and municipal sewage treatment plants. Many contaminants are present at much higher concentrations in sediment than in water because the contaminants are not readily soluble in water and they tend to adhere to organic and inorganic particulate matter. Sediment contamination has been associated with a variety of impacts on the biota of Puget Sound, and local health departments have advised local residents to limit their consumption of recreationally harvested seafood.

Pollution control programs in Puget Sound have traditionally focused on protecting water quality through effluent discharge limits and water quality standards. Such controls have generally not been effective at preventing sediment contamination. The control of sediment contamination is currently limited because no guidelines or standards have been available to assess adverse biological impacts of contaminated sediments. Such tools are needed primarily to 1) prevent sediment contamination by controlling contaminant inputs, 2) determine when and how to perform remedial operations in areas with contaminated sediment, and 3) determine how to manage the transfer and disposal of contaminated sediment.

The Puget Sound Water Quality Authority (the Authority) recently specified a series of activities to control sediment contamination in Puget Sound. These activities were set forth in the 1987 Puget Sound Water Quality Management Plan (the Plan, revised in 1989). In response to Element P-2 of the Plan, the Washington Department of Ecology (Ecology) is developing regulatory standards for identifying and designating sediments that have observable acute or chronic effects on biological resources or pose a significant health risk to humans. As part of the standards development process, Ecology convened four public workshops that were attended by representatives of environmental and public interest groups, ports, industry, state and federal agencies, local governments, and Indian tribes.

The present report describes issues, alternatives, and recommendations relative to the adoption of criteria for classifying sediments in Puget Sound. Technical analyses conducted for this report included review of specific issues concerning application of chemical-specific sediment standards and recommended approaches for biological testing and assessment of human health effects from contaminated sediments. The report is divided into five major sections, with discussion and recommendations organized to present a range of options for consideration. The advantages and disadvantages of nine available approaches for developing sediment quality values are reviewed in Section 2. Several specific issues with respect to the use of the Apparent Effects Threshold (AET) approach as one tool for developing state sediment quality standards are discussed in Section 3. In Section 4, discussion is presented on how biological testing can be integrated with chemical criteria to classify sediments on the basis of chronic and acute adverse biological effects. Several options for developing human health guidelines with respect to contaminated sediment are presented in Section 5. Finally, Section 6 provides a summary of the information presented in earlier sections with respect to the recommended approaches for developing state standards for contaminated sediment.

RECOMMENDED APPROACH FOR SEDIMENT QUALITY VALUES

In the past two decades, several federal, regional, and state agencies have developed numerical criteria or assessment methods for evaluating contamination in sediments and dredged material. Most early efforts for developing criteria were based on comparing chemical concentrations in

contaminated areas to those in reference areas, and did not consider biological effects. More recently, approaches for evaluating sediment quality have focused on determining relationships between sediment contaminant levels and adverse biological impacts. The following nine approaches were reviewed and summarized in this report:

- Field-based approaches
  - Reference area
  - Field-collected sediment bioassay
  - Screening level concentration (SLC)
  - Sediment quality triad (Triad)
  - Apparent effects threshold (AET)
- Laboratory/theoretically-based approaches
  - Water quality criteria (WQC)
  - Equilibrium partitioning (sediment-water)
  - Equilibrium partitioning (sediment-biota)
  - Spiked sediment bioassay.

Field-based approaches rely on empirical chemical and/or biological measurements of sediments to establish sediment quality values. Some of these approaches are purely chemical (e.g., reference area approach) or biological (e.g., field-collected sediment bioassay approach). Other approaches such as SLC, Triad, and AET correlate biological responses (e.g., bioassays on field-collected sediment, *in situ* biological effects observed in organisms associated with sediments) and chemical concentrations measured in sediments to develop sediment quality values. Laboratory/-theoretically-based approaches rely on extrapolation of water quality criteria to sediments, models of environmental fate of chemicals (e.g., equilibrium partitioning), or extrapolation of laboratory cause-effect studies to develop sediment quality values (e.g., spiked sediment bioassays).

The AET approach was selected by Ecology as the currently preferred method for developing sediment quality standards that address adverse biological effects in Puget Sound. Other approaches should still be considered as they are developed or tested, including approaches that directly assess human health concerns. The primary reason for selecting AET was its relatively high reliability in classifying Puget Sound samples as "impacted" or "not impacted." The reliability of the AET has been assessed using a large database comprising samples from 13 Puget Sound embayments (all biological indicators were not available in all embayments). In at least 85 percent of the available samples for each biological indicator, the approach either correctly classifies as "impacted" samples that exhibit adverse biological effects or correctly classifies as "not impacted" samples that do not exhibit adverse biological effects. In addition to its reliability in classifying sediments, the AET approach can be used to provide sediment quality values for the greatest number and widest range of chemicals of concern in Puget Sound. The approach also incorporates the widest range of biological indicators that are directly applicable to sediment conditions.

IDENTIFICATION AND EVALUATION OF CRITERIA DEVELOPMENT ISSUES

In this section, three issues are evaluated with respect to the use of AET values as a basis for setting sediment quality criteria for Puget Sound. The first two issues relate to the suitability of existing AET to intertidal and low-salinity areas. Uncertainty exists because existing AET have been developed primarily using data from marine subtidal areas. The third issue is based on the implications of excluding Microtox data from compilation of the lowest AET (LAET) for a range of biological indicators, and the influence of selected chemicals of concern from the Puget Sound Dredged Disposal Analysis (PSDDA) and the Puget Sound Estuary Program (PSEP) on the reliability of LAET.

Reliability is defined in terms of the following measures, which are evaluated with actual field data:

- **Sensitivity** in detecting environmental problems (i.e., are all biologically impacted sediments identified by the predictions of the chemical sediment criteria?)
- **Efficiency** in screening environmental problems (i.e., are only biologically impacted sediments identified by the predictions of the chemical sediment criteria?).

Suitability of AET Values in Intertidal Areas

At present, little information exists with respect to the biological effects of sediment chemical contamination in intertidal areas of Puget Sound. In addition, there are no readily available data on tidal elevations at the stations sampled in the intertidal areas. Because intertidal areas have a number of physical/chemical and biological characteristics that differ from subtidal areas, the suitability of existing AET (developed primarily in subtidal areas of Puget Sound) was examined for application in the intertidal zone. The primary purpose of this section is to identify the major factors that could limit the application of Puget Sound AET to intertidal areas and to qualitatively evaluate the implications of this application. The major conclusions of this section are:

- Different physical and chemical processes operating in the intertidal environment are predicted to both increase (e.g, potentially greater influence of microlayer contaminants) and decrease (e.g., generally increased particle size) the labile fraction of particle bound contamination relative to conditions in the subtidal.
- It is likely that the different contaminated matrices to which AET have been applied in the subtidal environment represent a broader range in matrix types, and associated variations in bioavailability, than do differences between subtidal and intertidal environments.
- Physical and chemical conditions in the intertidal environment suggest that natural recovery will be enhanced relative to the subtidal environment by a number of factors that are generally unique to the intertidal, including direct exposure to solar radiation (i.e., increased potential for photodegradation reactions), potentially enhanced groundwater flushing, and enhanced sediment reworking.
- Intertidal biological assemblages (i.e., groups of species) generally differ markedly from subtidal assemblages.
- In general, intertidal assemblages are comprised of relatively hardy species that can tolerate the large variations in environmental conditions often experienced in the intertidal zone.
- Although little is known of the effects of toxic chemicals on intertidal organisms, they may be more tolerant than subtidal organisms because they are adapted to a more variable (i.e., stressful) environment.
- If intertidal organisms are more tolerant of chemical toxicity than are subtidal organisms, use of AET developed using subtidal organisms may be a conservative approach to identifying potential problems.
- Because two of the test species used to develop existing AET occur in intertidal areas (i.e., *Rhepoxynius abronius*, *Crassostrea gigas*), it may be appropriate to use these species as representatives of intertidal organisms in general.

- Existing AET are recommended for use at intertidal stations in Puget Sound based on preliminary reliability studies. Ongoing review of any additional verification data is recommended. Based on existing reliability results and the small areas relative to subtidal sediments in Puget Sound, development of separate standards for intertidal sediments is not recommended as an efficient regulatory approach.

Suitability of AET Values to Low-Salinity Environments

Little information was found with respect to the biological effects of sediment chemical contamination in low-salinity areas of Puget Sound. Because these areas have a number of physical/chemical and biological characteristics that differ from marine areas, the suitability of existing AET (developed primarily in marine areas of Puget Sound) was examined for application to low-salinity environments. The primary purpose of this section is to identify the major factors that could limit the application of Puget Sound AET to low-salinity areas and to qualitatively evaluate the implications of this application. The major conclusions of this section are:

- The physical/chemical changes that accompany decreases in salinity include the following:
  - Decreased ionic strength
  - Altered dissolved metal speciation
  - Decreased particle affinity of neutral organic compounds
  - Decreased buffering capacity (and thus greater susceptibility to pH shifts)
  - Increased flocculation (at the freshwater/seawater contact zone).
- The net effect of decreases in salinity on contaminant bioavailability cannot be predicted because the physical/chemical processes that accompany these changes are complex and in some cases poorly understood.
- It is likely that the different contaminated matrices to which AET have been applied in the subtidal, marine environment represent a broader range in matrix types, and associated variations in bioavailability, than do differences between marine and low-salinity environments.
- Biological assemblages in low-salinity habitats generally differ markedly from assemblages in marine habitats.
- In general, assemblages in low-salinity habitats are comprised of relatively hardy species that can tolerate the large variations in environmental conditions often experienced in those environments.
- Although little is known of the effects of toxic chemicals on organisms from low-salinity habitats, it might be surmised that they are more tolerant than marine organisms because they are adapted to a more variable (i.e., stressful) environment.
- If organisms from low-salinity habitats are more tolerant of chemical toxicity than are marine organisms, then use of AET developed using marine organisms may be a conservative approach to identifying potential problems.
- Because two of the test species used to develop existing AET occur in low-salinity areas (i.e., *Photobacterium phosphoreum*, *Crassostrea gigas*), it may be appropriate to use these species as representatives of organisms from low-salinity habitats in general.

- Almost no information exists on the reliability of AET (which are derived in saline environments) when they are applied to low-salinity environments. Until verification studies have been conducted, application in low-salinity environments is not recommended.

The available field data in low-salinity environments consists exclusively of stations that are low in chemical contamination and do not exhibit biological effects. Before further application, it is recommended that the AET approach be field verified in low-salinity environments, especially in areas approaching freshwater conditions and in areas in which chemical contamination is of concern. A field verification program to characterize the applicability of AET in low-salinity environments is described in the report.

Reliability Tests for Alternate AET

Using the LAET for available biological indicators on a chemical-by-chemical basis is a means for generating a set of AET that should be protective for that range of indicators. Reliability tests were performed to examine 1) the influence of Microtox AET values on LAET reliability, 2) the influence of PSDDA chemicals of concern that do not have established AET, and 3) the influence of chemicals observed in Puget Sound that are not included on the PSDDA list.

Microtox Bioassay Effect on LAET Reliability--The Microtox bioassay has been considered for exclusion from LAET generation. The effect of the exclusion of these data is evaluated to determine the contributions of the Microtox bioassay to LAET reliability for other indicators, and to determine if the resulting LAET would still be predictive of Microtox results. Excluding Microtox data from the LAET resulted in relatively small decreases in sensitivity and increases in efficiency for all but the Microtox bioassay stations, at which the opposite trend was observed. Overall reliability increased by up to 11 to 15 percent for the amphipod bioassay, oyster larvae bioassay, and benthic infauna stations. The overall reliability for the Microtox bioassay stations was relatively unchanged (3 percent decrease). Based on decreases in sensitivity, and small increases in efficiency and overall reliability, the exclusion of Microtox results from LAET generation is not strongly warranted.

Chemicals of Concern Lacking AET Values--The influence of PSDDA chemicals of concern on the reliability of sediment quality values was evaluated. In this evaluation, LAET were used whenever available, and PSDDA screening levels were used for chemicals without established AET. The contribution of non-AET PSDDA chemicals of concern to reliability was evaluated by comparing the sensitivity and efficiency of two sets of sediment quality values:

- LAET for the 51 PSDDA chemicals of concern that have established AET
- LAET and screening level values for all 58 PSDDA chemicals of concern, using PSDDA screening level values for those chemicals without established AET.

Of the 58 PSDDA chemicals of concern, aldrin, chlordane, dieldrin, heptachlor, lindane (gamma-HCH), trichloroethene and hexachloroethane are all without defined AET. Screening level values for trichloroethene and hexachloroethane were calculated as 10 percent of the maximum level value set by the sediment-water equilibrium partitioning approach, and the pesticide screening levels were set at an estimated limit of quantitation (i.e., 5 times an assumed analytical detection limit of 2 ug/kg dry weight sediment). The stations with concentrations exceeding these chemical screening levels also exceeded at least one LAET value. Therefore, there was no difference between the reliabilities of the two chemical sets identified above. For this reason, 51 rather than 58 chemicals are recommended for the sediment quality standards.

Other Potential Chemicals of Concern--An additional test was conducted to evaluate the effect on reliability of augmenting the PSDDA chemicals with PSEP chemicals that were detected in greater than 5 percent of the Puget Sound sediment samples in the existing AET database, and for which AET have been established. A combined list of the PSDDA chemicals of concern and the PSEP chemicals of concern yielded the same results as the PSDDA chemicals alone. The only chemical added from the PSEP list, according to the considerations listed above, was 2-methoxyphenol (guaiacol). An additional evaluation was conducted by adding chemicals that are not on the PSDDA or PSEP lists but for which 1988 AET values exist. These additions resulted in an increase by 0 to 7 percent in sensitivity and decrease of 0 to 6 percent in efficiency, depending on the biological indicator used in the reliability analyses.

Although additional chemicals did not greatly improve the performance of LAET, it is recommended that chemicals that may be important near certain kinds of sources (e.g., guaiacols and dehydroabietic acid near pulp mill discharges) be considered for sediment quality management in localized areas. However, use of tentatively identified organic compounds in developing sediment quality standards is not recommended until data with more rigorous quantification are available.

ROLE OF BIOLOGICAL TESTING IN DEVELOPING SEDIMENT QUALITY STANDARDS

Element P-2 of the Plan specifies that sediment quality standards be developed for identifying and designating sediments having observable acute or chronic adverse effects on biological resources. The Plan also specifies that these standards may use physical, chemical, and biological tests with clearly identified pass/fail criteria. AET can be modified to identify chemical concentrations below which adverse biological effects would rarely be expected [i.e., No Effects Concentration (NEC)]. Biological testing is useful for confirming that biological effects are induced by sediments having chemical concentrations above NEC.

In this section, the potential role of biological testing in meeting the specifications of the Plan is presented. Discussions include 1) the general characteristics of the two major kinds of biological tests commonly used to assess adverse effects in sediments (i.e., sediment bioassays and *in situ* evaluations of indigenous organisms, primarily benthic macroinvertebrate assemblages); 2) the characteristics of the specific tests considered available for immediate use in Puget Sound; and 3) a scheme for classifying sediments in Puget Sound with respect to the presence or absence of adverse biological effects.

Characteristics of Available Biological Tests

The characteristics of eight sediment bioassays and two measures of *in situ* effects on indigenous organisms are evaluated for use in Puget Sound. These tests include:

- Acute (i.e., short-term) sediment bioassays
  - Amphipod (*Rhepoxynius abronius*) test
  - Amphipod (*Eohaustorius estuarius*) test
  - Juvenile bivalve (*Panope generosa*) test
  - Bivalve (*Crassostrea gigas*) larvae test
  - Echinoderm (*Dendraster excentricus*) embryo test
  - Microtox (*Photobacterium phosphoreum*) saline extract test
  - Microtox (*Photobacterium phosphoreum*) organic extract test

- Chronic (i.e., long-term) sediment bioassays
  - Juvenile polychaete (*Neanthes arenaceodentata*) test

- *In situ* effects on indigenous organisms
  - Characteristics of benthic macroinvertebrate assemblages
  - Bioaccumulation

Based on the results of the evaluations, the major conclusions are:

- Of the candidate bioassays evaluated in this report, the amphipod reburial test (both test species), the juvenile bivalve test, and the echinoderm embryo chromosomal abnormality tests are not presently recommended for use in confirming the toxicity of Puget Sound sediments with chemical concentrations exceeding NEC.

- Both amphipod mortality tests are recommended for immediate use. However, the availability of *E. estuarius* in the Puget Sound region should be determined. If it is found that local collection of this species is impractical, then the use of an alternate estuarine amphipod (e.g., *E. washingtonianis*) should be evaluated for testing sediments with interstitial salinities lower than 25 ppt.

- The juvenile polychaete test is recommended for use, pending future test development and standardization of the test protocols. This bioassay should be developed further primarily because it is the only candidate bioassay that directly addresses chronic effects.

- The bivalve larvae and echinoderm embryo developmental abnormality tests are recommended for use with minor revisions to the test protocols. To enhance the precision of the abnormality endpoint, a minimum number of larvae (e.g., 40-100) should be evaluated for each replicate analysis. In addition, the validity of using alternate test species should be determined.

- Both Microtox tests are recommended for use only as screening tools to determine which samples require further analysis using the other bioassays.

- Bioaccumulation is not presently recommended for evaluation of effects on indigenous organisms, but may be appropriate for human health risk assessment.

- Evaluation of effects on benthic macroinvertebrate assemblages is recommended as a measure of effects on indigenous organisms. However, a list should be developed of the kinds of habitats that are appropriate for evaluation using this test. In addition, minimal requirements for reference conditions and acceptable reference areas in Puget Sound should be specified.

Classification of Sediment Quality Based on Biological Effects

A scheme to classify sediments with respect to the presence or absence of adverse biological effects is developed in this section. The key elements of this classification scheme involve the integration of chemical criteria and biological testing. As much as possible, the classification scheme is consistent with similar schemes developed already by regional agencies. In this manner, decisions made using all schemes will be relatively consistent.

One major enhancement to existing classification schemes is the inclusion of chronic effects as an integral component. The importance of chronic effects in the scheme considered here derives from the Authority's goal of "no adverse effects" in Puget Sound. Because chronic effects generally occur at lower levels of chemical toxicity than do acute tests, it is anticipated that the classification scheme recommended here may identify more stations as exceeding criteria than would other schemes based largely on acute effects. However, the recommended scheme may be similar to approaches that incorporate a "safety factor" to account for potential chronic effects.

The general elements of the recommended classification scheme are presented in Figure TS1. At all stages, various options exist to modify or refine the scheme. Many of these options are discussed in the report.

The basic goal of the recommended scheme is to classify sediments as having or not having adverse effects, and to identify those sediments having adverse effects with respect to whether the effects are chronic or acute. It is expected that sediments having no adverse effects would pose little or no risk to aquatic biota, whereas sediments having acute effects would pose a substantial risk and possibly warrant some kind of management action. The environmental risks associated with sediments having only chronic effects are relatively less certain, but warrant consideration to attain the Authority's goal of protecting against adverse acute and chronic effects.

The first step in the recommended classification scheme involves the determination of whether any chemical in a particular sediment sample exceeds its NEC. The NEC is the concentration below which adverse effects are not expected to occur. If it is assumed that chronic effects generally occur at lower chemical concentrations than acute effects, NEC based on chronic effects should be environmentally protective and account for both chronic and acute effects. Because there is little information regarding chronic effects in Puget Sound, it is recommended that NEC be developed initially using chemical AET values based on *in situ* evaluations of benthic macroinvertebrate assemblages (which incorporate chronic effects) and an acute-to-chronic ratio (ACR) of 10 applied to chemical AET values based on existing acute bioassays.

At least three options are available for using an ACR approach to develop NEC. These options include the following:

1. Application of the ACR to the LAET for the acute bioassays and use of that value or the benthic AET, whichever is lower, as the NEC. Although this approach is very protective, it would probably generate many NEC with values comparable to or below those commonly found in reference areas.

2. Application of the ACR to the highest AET (HAET) for the acute bioassays and use of that value or the benthic AET, whichever is lower, as the NEC. Although this approach is less protective than Option 1, it would more frequently result in NEC that exceed reference area concentrations than would Option 1. In addition, this approach is consistent with the screening level (SL) approach used by PSDDA when evaluating options for dredged material disposal in Puget Sound.

3. Application of the ACR to the amphipod mortality AET and use of that value or the benthic AET, whichever is lower, as the NEC. This approach would account primarily for chronic mortality, as estimated from the ACR applied to the acute mortality evaluated in the amphipod bioassay.

To help ensure that NEC are not unreasonably sensitive or inefficient, it is recommended that they be no lower than the 90th percentile for chemical concentrations measured in all Puget Sound reference areas. Reference areas are defined as areas removed from major contaminant sources that have relatively low observed levels of both sediment contamination and adverse biological effects. The use of reference conditions to set lower limits for NEC will help prevent test sediments similar to Puget Sound reference sediments from being classified as having adverse effects, and thereby ensure that management activities are focused on sediments having the highest priority (i.e., sediments having chemical concentrations that exceed those in most Puget Sound reference areas).

Reliability tests are performed to compare the reliability of the three options for using an ACR approach to develop NEC. The reliability of existing PSDDA SL values is also compared to

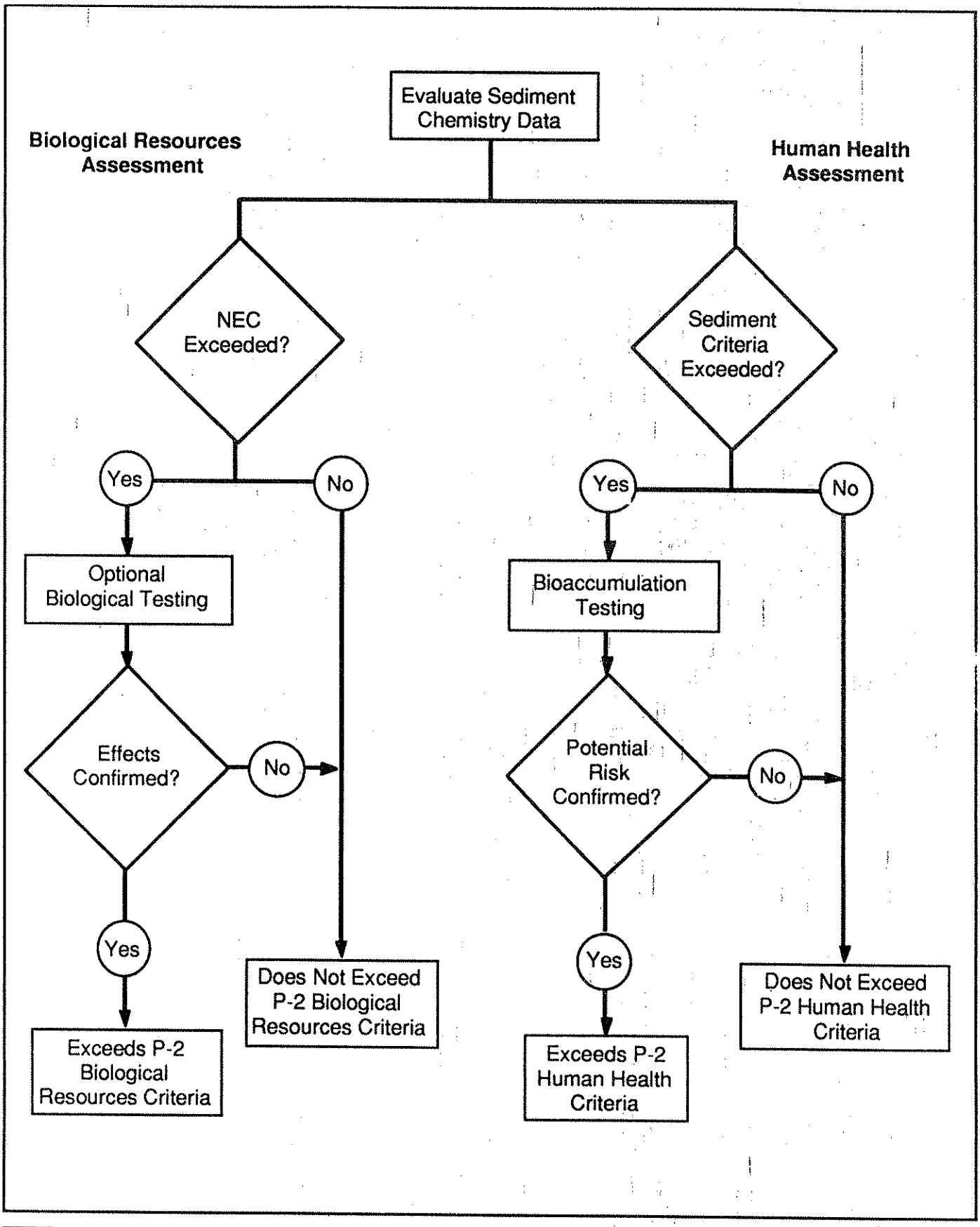


Figure TS-1. Integrated approach to biological resources and human health assessments

that of the alternative NEC values. All of these approaches result in values that are highly sensitive (i.e., >90 percent sensitivity in identifying impacted stations). Their efficiency in only identifying impacted stations (according to available acute bioassays and analyses of benthic macro-invertebrates) is relatively low (i.e., typically 30-60 percent). Of the various options, including PSDDA SL values, Option 2 (application of the ACR to the HAET for the acute bioassays and use of that value or the benthic AET, whichever is lower, as the NEC) is the most efficient.

Because NEC may not be available for all chemicals that may sometimes be present in samples, chemical screening alone may sometimes not be sufficiently protective. Therefore, if there is reason to suspect that a sample may contain potential problem chemicals that are not accounted for using NEC, biological testing could be required on a case-by-case basis despite the fact that no chemicals in the sample exceed existing NEC.

If a sample is classified as potentially having adverse effects based on NEC comparisons, and there is reason to believe the sample may not have acute or chronic effects, it could be subjected to biological testing to confirm its designation based on NEC. The test series could consist of sequential analysis using acute and chronic biological tests. If the sample fails the acute test, it would be identified as having acute effects. If it passes the acute test, it would be subjected to a chronic bioassay and identified as having chronic effects or classified as not having adverse effects based on the results of the latter test.

The proposed classification scheme will classify sediments in Puget Sound as having or not having adverse biological effects and identify those sediments with adverse effects according to whether the effects are chronic or acute. In doing so, the classification scheme will facilitate sediment management activities in the sound. The use of chemical screening criteria as an integral part of the proposed classification scheme will focus biological testing on the sediments having the greatest potential for causing adverse effects.

DEVELOPMENT OF HUMAN HEALTH GUIDELINES

Element P-2 of the Plan specifies that criteria for maximum allowable concentrations of contaminants in sediments be developed to protect public health. Sediments that fail the criteria would be classified as possibly posing a significant health risk. Options for development of sediment contamination guidelines based on consideration of potential human health risks are evaluated in this section of the report. The alternatives for use of human health guidelines in conjunction with other sediment quality standards are also presented.

In the preferred option for development of human health standards, a procedure called risk assessment would be used to develop numerical sediment quality criteria. Models would be developed to extrapolate data on sediment contamination to predict contamination of fish and shellfish consumed by humans and to extrapolate the results of laboratory animal bioassays involving chemical exposure to estimate potential health effects in humans. Because of the high degree of uncertainty in risk assessment models, it is recommended that the approach to human health guidelines for sediments be developed further before being incorporated into the regulations for sediment quality standards. Developmental needs for implementation of sediment quality standards based on human health risk assessment are summarized below.

Definitions and Overview of Risk Analysis

Risk is defined here as the probability that potential exposure of humans to toxic chemicals in contaminated sediments of Puget Sound will result in adverse health effects. Potential health effects of concern include, for example, cancer and birth defects from chronic exposure to chemicals such as polychlorinated biphenols (PCBs) and polycyclic aromatic hydrocarbons (PAH).

Risk assessment is a scientific procedure used to estimate the probability of adverse health effects that may result from exposure to toxic chemicals. Assessing risks of human exposure to toxic chemicals consists of the following four major steps:

- **Hazard Identification** - Qualitative evaluation of the potential for a substance to cause adverse health effects (e.g., birth defects, cancer) in animals or in humans

- **Dose-response assessment** - Quantitative estimation of the relationship between the dose of a substance and the probability of an adverse health effect

- **Exposure assessment** - Characterization of the populations exposed to the toxic chemicals of concern; the environmental transport and fate pathways; and the magnitude, frequency, and duration of exposure

- **Risk characterization** - Estimation of risk for the health effect of concern based on information from the first three steps.

Estimates of excess cancer risk in humans associated with chemical exposure are typically expressed as the probability that each exposed individual will experience cancer within his or her lifetime (usually assumed to be 70 years). For noncarcinogenic effects, there is usually a threshold dose (i.e., Reference Dose) below which no adverse health effects are expected over a lifetime of exposure. Risk estimates are determined for specific exposure pathways and routes. In the Puget Sound region, the most important way in which humans are exposed to toxic chemicals associated with sediments is through consumption of seafood organisms living in areas with contaminated sediments (e.g., bottomfish, crabs, clams).

Options for Developing Human Health Guidelines

The options outlined below for developing human health guidelines are general approaches that incorporate some or all of the steps of risk assessment defined earlier:

- **Option 1: Hazard Assessment Literature Review** - Option 1 involves using available information on the toxicity of chemicals detected in sediments to qualitatively evaluate the potential hazards to human health. Because Option 1 only provides information on relative hazards of chemicals, it can not be used to establish numerical sediment quality standards.

- **Option 2: Sediment-Specific Hazard Assessment** - Option 2 involves directly measuring genotoxic effects of contaminated sediments or tissues in short-term bioassays (e.g., the Ames mutagenicity bioassay), and using the results as a relative indicator of human cancer hazard associated with exposure to contaminated sediments. Although short-term genotoxicity tests do not provide estimates of carcinogenic potency of chemicals in humans, they could be an important supplement to risk assessment models because they account for effects of multiple chemicals and unanalyzed chemicals.

- **Option 3: Site-Specific Hazard Assessment** - Under Option 3, indirect evidence of potential human health hazards related to sediments would be obtained by measurement of liver disease in English sole associated with contaminated sediments. Although Option 3 would provide direct field assessment of multiple chemical effects in a vertebrate species, the different exposure regimes for fish and humans confounds the interpretation of fish pathology data relative to human health concerns.

- **Option 4: Quantitative Risk Assessment to Develop Sediment Quality Standards with Confirming Bioaccumulation Test** - Under Option 4, the EPA risk assessment approach would be applied to develop guidelines for chemical concentrations in edible tissues of fish or shellfish. These tissue quality guidelines would be extrapolated to establish numerical sediment quality standards. A bioaccumulation test involving exposure of clams to test sediments would be used to confirm biological uptake of chemicals sufficient to pose a potential health risk to humans. Although Option 4 provides numerical criteria based on the exposure pathway of primary concern (i.e., food-chain contamination), the unknown uncertainty levels associated with sediment-to-tissue extrapolations and the limited information available for many chemicals precludes immediate application of Option 4.

- **Option 5: Quantitative Risk Assessment to Develop Sediment Quality Standards** - Option 5 is similar to Option 4 except that the confirmatory laboratory test for bioaccumulation of contaminants is eliminated. Thus, there is additional uncertainty associated with the lack of a confirmatory test.

- **Option 6: No Action** - Under Option 6, sediment quality standards would be developed without guidelines derived from human health hazard or risk assessment. Although this would simplify the sediment quality standards, it is counter to the mandate specified in the Plan.

Because most of the options are not mutually exclusive, more than one option may be integrated into the sediment quality standards.

Recommended Approach to Human Health Guidelines

The recommended approach for human health guidelines (Figure TS1) consists of a combination of:

- **Qualitative Hazard Assessment (Option 1)** - Evaluation of the potential toxicity of chemicals measured in sediments based on available data and literature reviews of health effects

- **Quantitative Risk Assessment (Option 4)** - Modeling of potential human health risks resulting from consumption of fish or shellfish associated with contaminated sediments to derive sediment standards

- **Sediment-Specific Hazard Assessment (Option 2)** - Short-term bioassays (e.g., Ames test) of sediments or, preferably, tissues of organisms exposed to specific sediments.

Implementation of human health guidelines as part of the sediment quality standards requires further development of available hazard and risk assessment techniques. The developmental status and needs for each element of the recommended approach are summarized below.

Qualitative hazard assessment should follow procedures described by U.S. EPA (1986f) and Pastorok (1988). A state registry (either electronic or paper) of toxicity profiles should be established for Puget Sound chemicals of concern as defined under the proposed Element P-2 sediment standards. For many chemicals, the chemical file from EPA's Integrated Risk Information System (IRIS) will probably be sufficient to serve as the toxicity profile for the state registry.

A quantitative risk assessment model based on food-chain bioaccumulation of contaminants is recommended to address the question of how much human health risk may be associated with contaminated sediments. In the near term, available data and equilibrium partitioning models may be used to develop very conservative (i.e., protective) screening values for concentrations of

contaminants in sediments. Further refinement of quantitative approaches for setting human health guidelines is recommended as a long-term developmental effort. For example, models for relating sediment contamination to fish or shellfish contamination need to be developed further. Long-term development of equilibrium partitioning models to predict tissue concentrations of contaminants from corresponding concentrations in sediments is possible, but laboratory and field validation of models is needed for at least key organic contaminants (e.g., PAH, PCBs, DDT and related metabolites).

Finally, short-term mutagenicity bioassays (e.g., Ames test) of tissue extracts are recommended to complement the quantitative risk assessment modeling approach. Methods are available for analysis of tissue extracts using the Ames test, but problems in measuring the mutagenic potential of complex mixtures of contaminants require further research. Calibration of short-term bioassays to various concentrations of known mutagens in tissue samples is also needed.

Relationship of Human Health Guidelines to Other Sediment Criteria

Options for incorporation of human health guidelines into the proposed sediment quality standards are evaluated in the text. It is recommended that sediments be evaluated relative to the biological resources and the human health assessments independently. The results of both evaluations would be used to rank sites on the inventory. This approach would consistently provide the most information for classifying sites in Puget Sound. At the same time, it recognizes the uniqueness of potential effects on biological resources and human health.

1. INTRODUCTION

The following report describes issues, alternatives, and recommendations relative to the adoption of criteria for classifying sediments within Puget Sound. Technical analyses conducted for this report included review of specific issues concerning application of chemical-specific sediment standards and recommended approaches for biological testing and assessment of human health effects from contaminated sediments.

1.1. BACKGROUND

Sediment in several areas of Puget Sound is contaminated with potentially toxic substances such as petroleum-derived compounds, chlorinated organic compounds, and metals. Sources of these contaminants include runoff from urban streets, industrial discharges, and municipal sewage treatment plants. Many contaminants are present at much higher concentrations in sediment than in water because the contaminants are not readily soluble in water and they tend to adhere to organic and inorganic particulate matter. Sediment contamination in Puget Sound has been associated with impacts to benthic infauna, and development of tumors and other abnormalities in bottom-dwelling fish (Long and Chapman 1985; Barrick et al. 1985, 1986; Malins et al. 1982, 1984; Swartz et al. 1982). In addition, fish, crabs, and bivalves in contaminated areas have been observed to accumulate pollutants in their muscle tissue and organs (Dexter et al. 1981; Gahler et al. 1982; Yake and Norton 1986; Ginn and Barrick 1988; Barrick et al. 1985; Beller et al. 1988; Pastorok et al. 1988). In several of these areas (e.g., Elliott Bay, Commencement Bay, and Eagle Harbor), local health departments have advised local residents to limit their consumption of recreationally harvested seafood.

Pollution control programs in Puget Sound have traditionally focused on protecting water quality through effluent discharge limits and water quality standards. Such controls have generally not been effective at preventing sediment contamination. The control of sediment contamination is currently limited because no guidelines or standards have been available to assess adverse biological impacts of contaminated sediments. Such tools are needed primarily to 1) prevent sediment contamination by controlling contaminant inputs, 2) determine when and how to perform remedial operations in areas with contaminated sediment, and 3) determine how to manage the transfer and disposal of contaminated sediment.

In the early and mid 1980s several studies and programs were initiated to develop tools for evaluating and managing contaminated sediment:

- **Commencement Bay Superfund Investigations**—During the course of these investigations, the Apparent Effects Threshold (AET) method was developed for assigning values of sediment quality. AET are based on observed relationships between sediment contamination and a number of biological effects, including benthic infauna depressions, fish histopathology, and several bioassays (e.g., amphipod, oyster larvae, and Microtox).

- **Puget Sound Dredged Disposal Analysis (PSDDA)**—PSDDA is a cooperative program of the U.S. Army Corps of Engineers (Corps), U.S. Environmental Protection Agency (EPA), Washington Department of Ecology (Ecology), and Washington Department of Natural Resources. The primary objective of PSDDA is to develop environmentally safe and publicly acceptable options for unconfined, open water disposal of dredged material. PSDDA has developed procedures and guidelines (based in part on AET and a biological testing strategy) for evaluating dredged material, and has recommended disposal sites in central Puget Sound. Final evaluation guidelines and recommendation of disposal sites for north and south Puget Sound are under development.

- **Urban Bay Action Program**—The Urban Bay Action Program is a major component of the Puget Sound Estuary Program (PSEP) and was begun in 1984-1985 by EPA's Office of Puget Sound with substantial participation by Ecology, Puget Sound Water Quality Authority (Authority), and other state agencies, and local governments. Urban Bay Action Programs consist of the identification of contaminated problem areas (predominantly using site-specific biological tests and AET values to assess sediment contamination), identification of potential contaminant sources, development of an action plan for source control, and formation of an action team for action plan implementation.

Actions resulting from the above programs were recently focused by the Authority's Puget Sound Water Quality Management Plan (hereafter referred to as the Plan; PSWQA 1987). The Authority adopted a revised plan in 1989 which did not affect the technical criteria development issues considered in this report. Ecology is currently developing and implementing regulatory standards and guidelines for five areas dealing with sediments:

1. General sediment quality standards
2. Guidelines for limitations on the discharge of contaminants and contaminated particulate material
3. Standards for unconfined open water disposal of dredged material
4. Standards for confined disposal of dredged material
5. Guidelines for cleanup of contaminated sediments (i.e., remedial action).

As part of the development process for sediment quality standards, Ecology convened four public workshops that were attended by representatives from environmental and public interest groups, ports, industries, state and federal agencies, local governments, and Indian tribes.

The remainder of the introduction presents discussions of key elements of the Plan, a summary of public advisory committee meetings on development of sediment quality standards, and an overview of this report.

1.2. PUGET SOUND WATER QUALITY MANAGEMENT PLAN ELEMENTS RELATED TO SEDIMENT STANDARDS DEVELOPMENT

The primary program elements of the Plan that relate to the development of sediment standards are:

- **Element P-2** - Criteria for Classifying Sediments as Having Adverse Effects
- **Element P-7** - Effluent Limits in Permits—Particulates (and related Element P-3 addressing sediment dilution zones)
- **Element P-8** - Effluent/Sediment Monitoring in Permits
- **Element S-2** - Program for Unconfined Open Water Disposal, including PSDDA
- **Element S-4** - Confined Disposal Standards for Sediments
- **Element S-7** - Guidelines for Remedial Actions
- **Element S-8** - Investigations of Contaminated Sediment Sites (Urban Bay Action Programs)
- **Element SW-4** - Guidelines and Ordinances for Stormwater Programs Elements SW-1 and SW-2.

Element P-2 calls for Ecology to develop and adopt by regulation, criteria for identifying and designating sediments that have observable acute or chronic adverse effects on biological resources or pose a significant health risk to humans. The criteria for defining "sediments having adverse effects" may use chemical, physical, and biological tests and shall clearly define pass/fail standards for the tests. Initial criteria may deal exclusively with biological effects, but shall be revised to include human health concerns as this information becomes available. The criteria are to be used to limit discharges through the National Pollutant Discharge Elimination System (NPDES) (Element P-7) and the stormwater (Elements SW-2 and SW-4) and nonpoint source programs, to identify sites with sediment contamination (Element S-8), and to limit the disposal of dredged material (Element S-4). Sediments having adverse effects shall not be used as capping material for either dredged material disposal actions or remedial actions. A remedial action trigger and cost-modified cleanup levels higher than the "adverse-effect" levels may be established by Ecology for conducting cleanup actions (Element S-7).

The criteria that are to be developed by Ecology will include review and comment by an advisory committee composed of representatives from environmental and public interest groups, ports, industries, state and federal agencies, local governments, and Indian tribes. The criteria shall be reviewed at least every 3 years, and if laboratory-or field-based sediment quality values are used as the basis for developing the criteria, then these values shall be recomputed periodically to incorporate new data. The 1987 Plan calls for Ecology to develop interim sediment criteria by 30 June 1988, and final criteria by 30 June 1989.

Element P-7 requires Ecology to include specific conditions in NPDES permits to limit particulate contamination and ensure that the ambient sediment criteria (developed as part of Element P-2) will not be violated. Conditions may include source control measures, best management practices, numerical limits on the toxicity of the particulate fraction of the effluent, numeric limits on the mass or concentration of chemicals discharged, or other appropriate conditions. The 1987 Plan calls for Ecology to develop procedures for satisfying Element P-7 by 30 June 1989. Element P-3 is closely related to Element P-7 and requires Ecology to adopt administrative rules specifying criteria for the establishment of dilution zones in wastewater discharge permits, including sediment dilution zones. Sediment dilution zones are areas near a discharge point in which violation of sediment standards could be allowed. Final rules for Element P-3 are to be adopted by 30 June 1989.

Element P-8 calls for Ecology to require all appropriate types of monitoring during NPDES permit issuance or modification, including sediment monitoring in the vicinity of outfalls, analysis of the particulate fraction of effluents, sediment bioassays, and analyses of benthic infauna.

Element S-2 calls for the Authority to review PSDDA draft reports and environmental impact statements, and specify how state agencies and local governments are to conform their programs to PSDDA recommendations. If necessary, state agencies and local government are to revise their regulations and programs to conform to recommendations of PSDDA within 1 year after adoption of PSDDA by the Authority.

Element S-4 calls for Ecology to develop standards for the confined disposal of sediment that is classified as "having adverse effects" (Element P-2) and is not suitable for unconfined, open water disposal. The standards are to be used by Ecology and local government in the permit process for dredged material disposal. The standards shall address treatment and disposal options in water and on land. In setting the standards, Ecology is to consider solid and hazardous waste programs, and sludge management programs.

Element S-7 calls for Ecology to develop standards for deciding when to implement sediment remedial actions. Remedial actions can involve a variety of *in situ* treatment options (e.g., capping), or removal and treatment options. The guidelines are to consider deadlines for making decisions, natural recovery of sediment, procedures for determining priorities for action that include consideration of cost, and trigger levels for defining sediments that require expedited action.

Element S-8 calls for an expansion of the existing Urban Bay Action Program, and states specific recommendations for 1) developing an inventory of sites with adverse effects, 2) developing a priority list and investigation schedule for contaminated sites, 3) developing a ranking method that is applicable to contaminated sites in urban bays, 4) continuing investigations of contaminated sites, 5) continuing development of action teams for source control in contaminated areas, 6) carrying out remedial actions at contaminated sites (pursuant to guidance developed under Element S-7), and 7) continuing efforts at identifying responsible parties for assuming or sharing cleanup costs.

Element SW-4 requires Ecology to prepare and update guidelines and develop model ordinances for initial stormwater programs (Element SW-1) and for long-term urban stormwater programs (Element SW-2). Element SW-4 is relevant to the development of sediment standards to the extent that guidelines and ordinances incorporate measures to prevent violation of sediment standards. Final Element SW-4 rules, guidelines, and model ordinances are to be completed by 31 December 1989.

1.3. SUMMARY OF PUBLIC ADVISORY COMMITTEE MEETINGS

Ecology has conducted four public meetings on sediment quality during development of the draft sediment quality standards, Washington Administrative Code (WAC) 173-204. The purpose of the meetings was to receive comments and generate discussion on policy, procedural, and technical issues concerning the proposed standards. Table 1 presents an outline of the September 1988 draft of WAC 173-204. The meetings were attended by representatives from environmental and public interest groups, ports, industries, state and federal agencies, local governments, and Indian tribes.

The first public meeting, held on 20 April 1988, addressed a number of procedural, and policy, and technical issues. Major procedural and policy issues included questions regarding involvement of key participants and dissemination of information, jurisdictional scope of the regulations, and the need for conducting an analysis of the impacts of implementing the regulations. Major technical issues included the appropriateness of AET values, the need to carefully define important terms such as "sediment in effluent" and "impact zone," and questions regarding approaches to addressing human health concerns in the evaluation process.

On 11 May 1988, Ecology held the second public meeting and discussed procedural and policy issues. A number of specific issues were raised, including the following: 1) how sampling and analysis responsibility would be allocated (i.e., private sector or state) for evaluating sediment contamination, 2) whether standards should be set before determining the effectiveness of "all known available and reasonable methods of treatment" (AKARTs), 3) what type of tests may be required and how the cost of performing such tests will be taken into consideration, 4) concerns over the liability implications for dischargers who have contributed to sediment contamination that currently may exceed criteria, and 5) whether Ecology will consider the regulations as "applicable or relevant and appropriate requirements" (ARARs) for consideration at Superfund sites.

On 10 June 1988, Ecology held the third public meeting to discuss technical issues concerning the proposed regulations. In addition to regulatory and administrative issues, many specific issues were raised including those pursuant to the following technical areas: 1) the technical applicability and development of AET, 2) approaches to incorporating human health concerns into the evaluation and regulation process, 3) uses of No Effects Concentrations (NEC) for criteria development, 4) the roles of biological and chemical testing in evaluating sediments, 5) issues regarding discharges and definition of a sediment impact zone, and 6) approaches to and uses of biological testing.

On 21 September 1988, Ecology held the fourth public meeting to further discuss technical issues concerning the proposed regulations. Topics included 1) the options for setting No Observable Effects Concentrations, 2) the overall sediment designation process, 3) the options for addressing chronic biological effects, and 4) the options for considering human health criteria. Comments were also received on the draft version of the present report. This fourth meeting was

TABLE 1. OUTLINE OF WAC CHAPTER 173-204: DRAFT SEDIMENT QUALITY STANDARDS[a]

WAC 173-204-010 AUTHORITY AND PURPOSE

(1) Promulgation authority
(2) Purpose of standards
(3) Stringency of local ordinances

WAC 173-204-020 APPLICABILITY AND DISCHARGE PERMIT STANDARDS

(1) Applicability
(2) Discharge permits
(3) Sediment impact zone

WAC 173-204-100 DEFINITIONS

WAC 173-204-150 GENERAL CONSIDERATIONS

(1) Waste water dilution zones
(2) Conditions for review and revisions

WAC 173-204-200 PUGET SOUND MARINE SEDIMENT QUALITY STANDARDS

(1) Scope
(2) Puget Sound marine sediment chemical concentration criteria
(3) Sediment toxic/deleterious substances
(4) Confirmatory Puget Sound marine sediment biological tests
(5) Sediment standards designation procedures

WAC 173-204-205 NON-PUGET SOUND MARINE SEDIMENT QUALITY STANDARDS (RESERVED)

WAC 173-204-210 FRESHWATER SEDIMENT QUALITY STANDARDS (RESERVED)

WAC 173-204-260 SAMPLING AND TESTING STANDARDS

(1) Applicability
(2) Requirements for maintaining sampling plans
(3) Sediment sampling locations and procedures and testing protocols
(4) Revisions to protocols

WAC 173-204-400 RECORDS MANAGEMENT

(1) Applicability
(2) Requirements for maintaining records

[a] Draft sediment quality standards dasted 31 July 1988.

the last one scheduled prior to release of the final draft standards. A responsiveness summary is being prepared by Ecology and will be available to the public before adoption of the final sediment quality standards.

Information generated as a result of the public meetings was incorporated into the development of draft regulations. According to the Plan, the regulations are scheduled for final adoption by 30 June 1989. However, a sediment advisory group was established by Ecology in August 1988 to identify key policy issues related to the proposed regulations. As of February 1989, the sediment advisory group had identified several issues that may result in a delay of 6-12 months in adopting the proposed regulations. A decision on any delay is expected from the Authority in April 1989.

1.4. ORGANIZATION OF THE REPORT

The remainder of this report is divided into five major sections, with discussion and recommendations organized to present a range of options for consideration. The advantages and disadvantages of nine available approaches for developing sediment quality values are reviewed in Section 2. Several specific issues with respect to the use of the AET approach as one tool for developing state sediment quality standards are discussed in Section 3. In Section 4, discussion is presented on how biological testing can be integrated with chemical criteria to classify sediments on the basis of chronic and acute adverse biological effects. Several options for developing human health guidelines with respect to contaminated sediment are presented in Section 5. Finally, Section 6 provides a summary of the information presented in earlier sections with respect to the recommended approaches for developing state standards for contaminated sediment. Since the draft contaminated sediments criteria report was completed in August 1988, recommendations have been reevaluated based on new information as it became available (e.g., final reports concerning refinement of AET values; Barrick et al. 1988).

2. RECOMMENDED APPROACH FOR SEDIMENT QUALITY VALUES

Element P-2 of the Authority's 1987 Plan requires Ecology to develop, and adopt by regulation, criteria for identifying and designating sediments that have adverse biological effects or pose a significant human health risk. Ecology has selected the AET approach as the primary method for establishing sediment quality values, principally because of its reliability in predicting adverse biological effects in Puget Sound. The purpose of this section is to summarize advantages and disadvantages of alternative sediment quality approaches that address adverse biological effects, including the AET approach (human health concerns are addressed more explicitly in Section 5). These advantages and disadvantages include those summarized in available reviews of the various alternative approaches. Sediment quality standards for chemicals are used in combination with a biological testing strategy that addresses ecological and human health concerns (see Sections 4 and 5) to recommend a sediment regulatory level for contamination (see Section 6).

2.1. REVIEW OF ALTERNATIVE APPROACHES

In the past two decades, several federal, regional, and state agencies have developed numerical criteria or assessment methods for evaluating contamination in sediments and dredged material. Most early efforts for developing criteria were based on comparing chemical concentrations in contaminated areas to those in reference areas, and did not consider biological effects. More recently, approaches for evaluating sediment quality have focused on determining relationships between sediment contaminant levels and adverse biological impacts. Much of the information and analysis presented in this section is contained in recent reviews of approaches to sediment quality value development (e.g., Beller et al. 1986; Lyman et al. 1987; Battelle 1988; and Chapman in press). Based on these documents, the following approaches are reviewed and summarized in this section:

- Field-based approaches
  - Reference area
  - Field-collected sediment bioassay
  - Screening level concentration (SLC)
  - Sediment quality triad (Triad)
  - Apparent effects threshold (AET)
- Laboratory/theoretically-based approaches
  - Water quality criteria (WQC)
  - Equilibrium partitioning (sediment-water)
  - Equilibrium partitioning (sediment-biota)
  - Spiked sediment bioassay.

Field-based approaches rely on empirical chemical and/or biological measurements of sediments to establish sediment quality values. Some of these approaches are purely chemical (e.g., reference area approach) or biological (e.g., field-collected sediment bioassay approach). Other approaches such as SLC, Triad, and AET correlate biological responses (e.g., bioassays on field-collected sediment, *in situ* biological effects observed in organisms associated with sediments) and chemical concentrations measured in sediments to develop sediment quality values. Laboratory/-theoretically-based approaches rely on extrapolation of water quality criteria to sediments, models of environmental fate of chemicals (e.g., equilibrium partitioning), or extrapolation of laboratory cause-effect studies to develop sediment quality values (e.g., spiked sediment bioassays).

None of the available approaches is fully capable of addressing all concerns over interactive effects among chemicals and the effects of multiple chemicals on organisms. Hence, field

verification using diverse environmental samples is important to the evaluation of each approach for current use in Puget Sound (see Section 2.2). A definition of each approach as well as major advantages and disadvantages are described in the following sections.

2.1.1. Field-Based Approaches

Reference area, field-collected sediment bioassay, SLC, Triad, and AET approaches are discussed below. These approaches focus on the development of sediment quality values based at least in part on the collection and evaluation of field data for sediments.

Reference Area Approach—Using the reference area approach, sediment quality values are derived based on comparisons of chemical concentrations at a site with concentrations in an appropriate reference area. Reference areas may be pristine, or considered to have acceptably low levels of contamination and adverse biological effects (Beller et al. 1986). The primary advantage of this approach is that it has minimal data requirements. In many cases, comparisons of sediment chemistry values can rely on historical data for the reference areas. In addition, the reference area approach is the only approach that does not require quantitative toxicological data for contaminants of concern in sediments.

A disadvantage of the reference area approach is the difficulty and subjectivity inherent in selecting an "appropriate" reference area. The selection of any given reference area for establishing sediment quality values implies a management decision to use the reference area chemical concentrations, whether these levels are below or above the threshold at which biological effects may occur. Because only chemical concentrations are considered in this approach, variations in the sensitivity among organisms to different contaminants is not taken into account. Also, sediment quality values developed using this approach are generally considered specific to the region where the reference samples were taken and would not necessarily be applicable to other areas (e.g., Lyman et al. 1987).

A modified reference area approach (modified by incorporation of bioassay testing) was developed by EPA Region 10 and Ecology to evaluate the suitability of dredged material for open water disposal in Puget Sound. In essence, dredged material is judged as unsuitable for open water disposal if chemical concentrations in dredged material exceed values reported for the disposal sites, and if mortality or abnormality in bioassay tests using dredged sediments are significantly higher than mortality or abnormality using sediment from the disposal sites. Biological testing greatly enhances the usefulness of the modified reference area approach. However in the specific case cited, the approach is limited by a scarcity of data from reference areas (i.e., disposal sites) which reduces the statistical confidence of the comparison. Even with an adequate database, the modified reference area approach is site-specific and does not establish relationships between sediment contamination and adverse biological impacts.

Field-Collected Sediment Bioassay Approach—The field-collected sediment bioassay approach consists of exposing test organisms to field-collected sediment, and comparing mortality or sublethal effects to effects observed in experiments using sediment from a reference area. The field-collected sediment bioassay approach has several advantages over other approaches. Bioassays are laboratory controlled experiments that can provide a relatively high degree of repeatability and precision (Battelle 1988). Bioassays can be made as sensitive as necessary by careful selection of test species and experimental endpoints. Bioassays can be used to develop dose-response relationships for assessing potential impacts from areas with varying degrees of contamination. Finally, bioassays are a well accepted scientific technique and have a strong regulatory basis (e.g., EPA WQC). A further advantage of this approach is that chemical data are not required to classify a sediment as adversely impacted; hence, the approach addresses concerns over unmeasured chemicals.

This latter advantage is also a limitation of the field bioassay approach because sediment is treated as a "black box" by measuring the combined effect of all contaminants present in a sample without benefit of any chemical characterization. One consequence of this limitation is the constraint placed on identification of appropriate management alternatives or source accountability. Another limitation is that field-collected sediment bioassays deal only with selected organisms (i.e., the test species) under controlled conditions. To address a wide range of biological responses, an expensive battery of tests may be required for each sample. Also, bioassessment tools may either be not available for or not capable of addressing all chemical effects of concern. The interpretation of bioassay responses requires the extrapolation of results obtained under controlled laboratory conditions to field conditions where environmental factors (e.g., temperature, salinity, and depth) vary and communities are composed of many species with different sensitivities to toxicity. By itself, this approach is useful for identifying problem sediments, but would require integration with another approach to determine chemical-specific sediment quality values.

Screening Level Concentration Approach—For any given chemical, the SLC is determined as the sediment concentration above which less than 95 percent of the total enumerated species of benthic infauna are present (Battelle 1986; Neff et al. 1988). This approach was initially developed as a technique for differentiating between concentrations of nonpolar organic chemicals that pose a threat to biota and those that do not.

Development of SLC requires arranging stations (at which a particular benthic invertebrate species is present) sequentially with respect to increasing concentration of the contaminant of interest. Several species-specific values (SSLC) are developed and arranged in order of increasing contamination. The SLC is the concentration of a contaminant above which 95 percent of the SSLC are found. SLC calculation has several minimum data requirements, including: 20 stations for each SSLC calculation, 10 taxa for each SLC calculation, arrangement of stations spanning a contaminant concentration gradient, and taxonomically homogeneous taxa (e.g., all identified to species level or genus level; Battelle 1986; Beller et al. 1986; Chapman in press). SLC have been developed for nine chemicals in both fresh and marine sediments (Neff et al. 1988).

One advantage of the SLC approach is that it uses site-specific field data and is based on a method designed to be consistent with the goals of EPA WQC (i.e., protection of 95 percent of aquatic species). Although the nonreliance on comparisons of contaminated areas to reference areas is a potential disadvantage of the SLC approach because natural and sampling variability are not directly considered, it has the advantage of not being affected by biases inherent in reference area selection (Battelle 1988). The approach has been applied mainly to nonpolar organic contaminants to enable use of the organic carbon normalization theory for sediment chemistry. However, the approach has been applied in preliminary tests to other selected contaminants using Puget Sound data normalized to dry weight (Beller et al. 1986, 1988). Development of SLC is data intensive, requiring a fair amount of field data spanning a range of concentrations, and infaunal taxonomic identification to the species level.

The SLC approach implicitly assumes that observed alterations to benthic communities are due to contaminants in the sediment, irrespective of other possible causes (e.g., natural variability in substrate depth and sediment texture; Lyman et al. 1987). The original approach uses only presence and absence of species to measure effects and therefore does not address the major reductions in numbers of individuals that would occur before a particular species becomes absent. Also, no criteria are applied for indicator species selection. SLC may not protect a substantial portion of the biota if much less than 5 percent of the species are sensitive to the contaminant of concern. Alternatively, SLC may be overly protective if much more than 5 percent of the species selected are sensitive to the contaminant of concern. Finally, SLC values are highly sensitive to the range and distribution of sampling locations (e.g., to accurately represent a gradient of chemical concentrations and biological impacts; Lyman et al. 1987 and Chapman in press).

Sediment Quality Triad—The Triad approach involves analyzing relationships among contaminant concentrations in sediment, sediment bioassay endpoints, and *in situ* studies (e.g.,

benthic infaunal community alterations). A wide variety of bioassay organisms and endpoints may be used in this approach, including lethality, alterations in respiration rate, developmental abnormalities, mutagenicity, and cytotoxicity. The most common *in situ* studies used by this approach are benthic community structure and fish histopathology. The Triad approach subjectively establishes sediment quality values in terms of chemical concentrations below which biological effects would be expected to be minimal, and above which biological effects would be expected to be severe (Chapman in press). The Triad approach assumes (Chapman in press) that 1) a variety of endpoints of various bioassays using different contaminants, and a variety of measurements of benthic community structure are appropriate indicators of biological impact, and 2) these indicators can be treated in an additive manner and each have equal weight.

The combination of *in situ* studies and laboratory bioassays in this approach enables biological effects associated with sediment contamination to be differentiated from effects caused by natural variability and/or laboratory artifact. Another advantage of the Triad approach is that it can be used to develop sediment quality values for any measured contaminant (Chapman in press and Battelle 1988).

Disadvantages of the sediment Triad approach are that it is data intensive (e.g., strongly favors identification of benthic invertebrates to species level), and currently, sediment quality values have been developed only for lead, polycyclic aromatic hydrocarbons (PAH), and polychlorinated biphenyls (PCBs). The Triad approach also incorporates subjective rather than objective (e.g., statistical) criteria for determining sediment quality values. Also, although comparative data for sediment chemistry, bioassays, and *in situ* effects are derived from the same general geographic area, these data may not always represent the same sample or station (i.e., the data are not necessarily synoptic). These two aspects are the major distinctions between the Triad and AET approaches (see discussion below).

Apparent Effects Threshold Approach—An AET for a chemical is defined as the concentration in sediments above which statistically significant biological effects (relative to reference sediments) would always be expected to occur (Barrick et al. 1985; Beller et al. 1986). The AET approach uses matched (i.e., synoptically collected) data on sediment chemistry, benthic infaunal effects, and sediment bioassays to determine concentrations above which all samples for a particular biological indicator exhibit adverse effects. Adverse effects are defined as a statistically significant difference ($P \leq 0.05$) between conditions in a study area relative to conditions in an appropriate reference area. AET can also be established for biological indicators that reflect area-wide conditions (i.e., over multiple sediment stations) such as bioaccumulation in fish, and fish histopathology. AET have been developed for 64 organic chemicals and metals in Puget Sound and for 4 independent biological indicators (amphipod, oyster larvae, and Microtox bioassays, and benthic infaunal abundance).

AET (and other approaches except the spiked bioassay approach) do not prove cause-effect relationships between contaminants and effects (Beller et al, 1986; Battelle 1988). Instead, AET identify concentrations of contaminants that are associated exclusively with sediments having statistically significant biological effects relative to reference sediments. There are several advantages to the AET approach. Unlike field-collected sediment bioassays, the AET approach does not treat sediments as a "black box" and takes a step toward differentiating between effects associated with different contaminants. The approach relies on objective statistical criteria for determining adverse effects for each biological indicator. In addition, there are no constraints on the types of chemicals for which AET can be developed, and various biological effects can be used to develop AET. Also, the AET approach is flexible because values developed for a diverse set of biological indicators can reflect different levels of environmental protectiveness. This last attribute is seen by some as a disadvantage because of the desire to specify only a single value for management or regulatory purposes (Lyman et al. 1987).

The major disadvantage of the AET approach is that it requires extensive data collection for chemical variables and at least one biological indicator. There is also a possibility that AET could be set at a level higher than required for complete environmental protection due to the fact that

biological effects can be observed at levels below AET. The approach assumes that other chemicals or environmental conditions contribute to effects below the AET for a particular chemical. As with any approach that associates chemical concentrations with biological effects, AET may also be set low relative to actual environmental concerns in cases where factors other than contamination contribute to adverse biological effects (Beller et al. 1986 and Chapman in press).

Benthic infauna AET currently developed for Puget Sound have been criticized for oversimplifying benthic community structure analysis by identifying organisms only to major taxonomic levels (Battelle 1988). AET can be developed based on species level identification of benthic infauna, but sufficient data for this application would be costly to acquire and analyze. In addition, there is greater variability in the abundance of individual species and all species of interest may not be found in suitable reference areas. Preliminary benthic AET have been developed for selected contaminants based on species-level identification in Commencement Bay. These AET are typically within a factor of 2 of AET based on identification to major taxa (Beller et al. 1988).

There are three major sources of uncertainty in determining AET (Beller et al. 1986):

1. Statistical classification error ($P<0.05$) associated with the significance of bioassay and benthic infauna results

2. The difference between the maximum concentration not associated with an effect and the next highest concentration that is associated with an effect (the "gray area")

3. Sampling intensity sufficient to ensure representation of a wide range of chemical concentrations and biological effects.

Beller et al. (1986) and Chapman (in press) estimate that uncertainty for individual AET ranges from less than a factor of 2 to less than 100 among chemicals. AET developed for non-site specific biological indicators (e.g., fish histopathology) may incorporate additional uncertainty by requiring averaging of chemical data over large geographic areas. Because AET are based on statistically significant biological effects relative to reference area conditions, the approach has the disadvantage of the reference area approach in which reference conditions are selected on a somewhat subjective basis. Based on this consideration, screening criteria (e.g., acceptable mortality in bioassays) have been developed to preclude the selection of reference areas in which biological conditions may indicate stress (Barrick et al. 1988).

As with all approaches except the spiked bioassay approach, interactive effects of chemicals cannot be distinguished using the AET method. The AET approach incorporates to some degree the influence of such effects in the setting of sediment quality values. The potential influence of these effects is discussed in this section. Additivity and synergism can result in the lowering of AET for a given chemical relative to when such effects do not occur. Analogously, antagonism can result in raising AET for a given chemical by masking impacts that would otherwise be more severe. Whether potential interactive effects result in AET that are not representative of general environmental conditions can only be speculated. Such AET would not be expected to be highly predictive of adverse impacts. The predictive success of AET has been tested in Puget Sound using independent data sets in which complex mixtures of chemicals are present. The generally high reliability with which adverse impacts are predicted using Puget Sound AET (Barrick et al. 1988) indicates that the potential existence of interactive effects does not hamper application of AET.

2.1.2. Laboratory/Theoretically-Based Approaches

The approaches described below generally use laboratory spiking studies, water quality bioassays, or theoretically-based relationships between sediment contaminant concentrations and adverse biological conditions as a premise for the development of sediment quality values. WQC used in several of the approaches discussed in this section were developed from laboratory toxicity bioassays on aquatic species generally inhabiting the water column rather than sediments. The most

rigorous of the approaches described in this section is the spiked sediment bioassay approach. Such bioassays are designed to establish direct cause-effect relationships between sediment contamination and biological effects under controlled laboratory conditions much the same as WQC are designed to establish cause-effect relationships between water-column concentrations of contaminants and biological effects.

Water Quality Criteria Approach—The WQC approach consists of measuring chemical concentrations in interstitial water and comparing them directly to EPA WQC. The primary advantage of this approach is that it draws upon the existing EPA toxicological database for WQC and directly applies this knowledge to measurement of chemical concentrations in interstitial water. Biological measurements of field samples are not required for the application of this approach.

The two most evident drawbacks of this approach are that 1) EPA WQC for salt water have only been developed for 10 metals and 9 organic chemicals (not including PAH), and 2) there are no standard procedures for obtaining and analyzing interstitial water samples (especially for organic compounds). In addition, important assumptions are implied in the application of this approach and cannot be validated with existing data. For example, dissolved organic matter was not a controlled variable in toxicity tests used to establish WQC, yet sediment bioassays indicate that organic carbon content of sediment may have a major effect on toxicity (e.g., Battelle 1988 and Chapman in press). In addition, there are questions about the applicability of these tests to organisms that live in close association with sediments and that may ingest contaminated organic material that is part of the sediments (in addition to exposure to contaminants from interstitial water; e.g., Battelle 1988). Use of WQC for evaluating contaminated sediment would not account for potential contaminant interactions that may result in higher (or lower) toxicity than indicated by the contaminant-specific criterion alone.

Equilibrium Partitioning (Sediment-Water) Approach—A sediment quality value based on sediment-water equilibrium partitioning is the sediment contaminant concentration (organic carbon normalized) that would be expected to result in an interstitial water concentration equivalent to the corresponding EPA WQC. This approach is based on a simple model that describes the equilibrium partitioning of a contaminant between sedimentary organic carbon and interstitial water, with little dependence on other physical or chemical factors. The approach is assumed applicable to all sediments, including those with very low (<0.1 percent) total organic carbon (TOC) content (Battelle 1986; Chapman in press).

The primary advantage of this approach is that it uses the existing EPA WQC toxicological database and does not require incurring the expense of collecting biological data. Also, equilibrium partitioning is readily applicable to various environmental settings. For nonionic compounds, the approach has a firm theoretical and empirical basis.

The sediment-water equilibrium partitioning approach has the same limitations as the WQC approach regarding uncertainties over application of such criteria to sediment-interstitial water systems (e.g., questions concerning the applicability of water column toxicity tests to organisms that may ingest sediments). In addition, the primary implicit assumption of the approach is that steady-state equilibrium exists in all aquatic environments. This assumption is susceptible to the following three major sources of uncertainty relative to estimated or calculated partition coefficients: 1) there is considerable variation of partition coefficient values reported in the literature, 2) suspended or dissolved organic matter in interstitial water may cause deviations from values predicted from experiments using a pure aqueous phase, and 3) laboratory-determined partition coefficient values are dependent on volume ratios of sediment-water that may not represent field conditions. Finally, the approach is currently of very limited use for ionic, polar organic contaminants and metals and metalloids, and does not address potential interactions (e.g., synergism and antagonism) among contaminants. Beller et al. (1986) and Chapman (in press) estimate that uncertainty factors for identifying problem sediments for individual contaminants using the sediment-water equilibrium partitioning approach, vary from less than 10 to more than 1,000,000.

Equilibrium Partitioning (Sediment-Biota) Approach—Using the sediment-biota equilibrium partitioning approach, sediment quality values are determined by estimating the sediment concentration of a contaminant that would be expected to result in a body burden of the contaminant (in benthic organisms) exceeding an existing regulatory limit [e.g., U.S. Food and Drug Administration (FDA) Action Limit]. The primary advantage of this approach is that sediment quality values can be established using only sediment chemistry data and appropriate tissue guidelines (or WQC and partition coefficient values if necessary, to estimate sediment contaminant concentrations that would result in tissue concentrations of concern).

This approach is limited to hydrophobic neutral organic compounds (Lyman et al. 1987). Other major disadvantages of this approach stem from its underlying assumptions, many of which are similar to those of the sediment-water partitioning approach. For example, the approach assumes that thermodynamic equilibrium exists among sediment, organisms, and interstitial water; however, poor correlations have been observed between partition coefficients and bioconcentration factors for compounds that are readily metabolized by fishes (e.g., PAH; Connor 1984; Oliver and Niimi 1985). In addition, the approach assumes that hydrophobic compounds associate predominantly with lipids in all aquatic organisms, with no variation among organisms. This assumption has been supported by some studies, but refuted by others (e.g., references cited in Beller et al. 1986). Also, a chemical-specific approach such as equilibrium partitioning does not account for effects of chemical mixtures.

A potential disadvantage is that the approach is based on the use of numerical guidelines that exist for few contaminants and that may be established for reasons other than environmental impact. For example, FDA limits have only been promulgated for 10 pesticides and PCBs, and are designated primarily to protect human health, taking into account socioeconomic factors (e.g., potential economic effects of a fishery closure). This disadvantage can be partially countered by use of quantitative risk assessment to establish tissue guidelines for multiple chemicals based on human health concerns related to consumption of contaminated seafood.

Spiked Sediment Bioassay Approach—The spiked sediment bioassay approach consists of developing dose-response relationships by exposing test organisms to sediments that have been inoculated with known concentrations or amounts of chemicals (or chemical mixtures). In contrast to the field-collected sediment bioassay approach, this approach can be used to develop cause-effect relationships between specific chemicals (or mixtures) and biological responses. The spiked sediment bioassay approach is the only reliable method for testing interactive effects because of the laboratory controls that can be imposed.

As with the field-collected sediment bioassay approach, this approach requires extrapolation of laboratory-derived results to field conditions; a process which implicitly assumes that spiked sediment under laboratory conditions is analogous to contaminated sediment in the environment (e.g., Chapman in press). Another major limitation of the spiked sediment bioassay approach is that it would require a large expenditure of resources to develop sediment quality values for all chemicals and chemical mixtures of concern in Puget Sound (or any other area; e.g., Lyman et al. 1987). The major technical source of uncertainty in using the spiked sediment bioassay approach is the degree to which chemical-sediment associations in the environment differ from those created under laboratory conditions.

2.2. SUMMARY COMPARISON OF AVAILABLE APPROACHES

In this section, the approaches discussed above are rated according to the degree to which they satisfy the requirements or conditions of 10 criteria (Table 2). The criteria were selected to assess the approaches with respect to two generally desirable characteristics relevant to the development of Puget Sound standards:

TABLE 2. SUMMARY EVALUATION OF AVAILABLE SEDIMENT QUALITY VALUE APPROACHES RELATIVE TO USE IN PUGET SOUND SEDIMENT MANAGEMENT PROGRAMS

Evaluation Criteria	Reference Area	Field Collected Sediment Bioassay	Screening Level Concentration	Sediment Quality Triad	Apparent Effects Threshold	Water Quality Criteria	Equilibrium Partitioning (Sediment-Water)	Equilibrium Partitioning (Sediment-Biota)	Spiked Sediment Bioassay
Generally low data requirements and low cost of sediment quality value development	0 (+)[a]	0 (+)	-	- (0)	- (+)	-	+	+	-
Routine application as regulatory tool is probably not costly[b]	0	+	0	0	0	-	0	0	0
Allows for the development of chemical-specific values	+	-	+	+[c]	+	+	+	+	+
Allows for the development of sediment quality values for a wide range of chemicals	+	-	+	+[c]	+	0	0	0	+
Values for a wide range of Puget Sound problem chemicals are currently available	+	NA	-	-	+	-	-	-	-
Incorporate the influence of chemical mixtures in sediments	-	+	+	+	+	-	-	-	+
Incorporates a range of biological indicator organisms	-	+	+	+	+	+	+	+	0[d]
Incorporates direct measurement of sediment biological effects	-	+	+	+	+	-	-	-	+
Predictions are generally applicable to historical sediment chemistry data	+	-	+	+	+	-	0	0	+
Has been field verified to some extent in Puget Sound	NA	+	-	0	+	-	0	-	-
Provides proof of cause-effect relationships between chemicals and biological effects	-	-	-	-	-	-	-	-	+[e]

[a] Parentheses indicate score based on the relative incremental cost of developing the approach for application in Puget Sound.

[b] Assumes equal level of development for all approaches (i.e., equal number of sediment quality values for each approach).

[c] The triad approach is primarily a method for intercomparing sediment chemistry and biological effects information, but can be used to subjectively estimate sediment quality values for specific chemicals.

[d] A range of organisms could be used in the spiked bioassay approach with a wide range of contaminants, but at present appears to be too costly and time-consuming to be practical.

[e] Proof under controlled laboratory conditions only.

- Applicability to existing and planned sediment management programs in Puget Sound [including the requirements of the Plan (PSWQA 1987)]
- Feasibility of implementation in the near term (i.e., before mid-1989).

The approaches are assigned a subjective scoring of "-", "0", or "+" to enable a relative comparison based on each criterion. A "-" is assigned in cases in which an approach does not meet the conditions of a criterion (e.g., relative to a cost criterion, a method is expensive to develop), a "0" is assigned in cases in which an approach somewhat meets the conditions of a criterion (e.g., an approach may be moderately expensive to develop), and a "+" is assigned in cases in which an approach substantially or fully meets the conditions of a criterion (e.g., an approach is not expensive to develop). A "NA" is assigned in cases in which a criterion is not applicable to an approach. The scoring rationale for each criterion is discussed below.

2.2.1. Data Requirements and Cost of Sediment Quality Value Development

This criterion is a relative measure of the data requirements and cost of initial development of sediment quality values for an approach. The SLC, Triad, AET, and spiked sediment bioassay approaches are assigned a score of "-" because they have relatively extensive data requirements (e.g., sediment chemistry, one or more *in situ* biological effects measurements, and one or more laboratory bioassays) for sediment quality values development (Table 2).

At present, the additional cost of AET development for Puget Sound is minimal because a large Puget Sound database has already been compiled and a wide range of AET values is available for use. Therefore, a "+" rating is shown in parentheses for the AET approach to reflect the current minimal cost of development. However, there would be additional costs in confirming that this database is or is not applicable statewide. Similarly, with moderate cost, existing Puget Sound data could be used to further expand the application of the Triad approach in Puget Sound. Therefore, a "0" rating is shown in parentheses for the Triad approach. The reference area approach receives a score of "0" because it initially requires only the collection of sediment chemistry data. Currently, the PSEP database for Puget Sound could also be used at negligible costs to provide reference area values. Therefore, a "+" rating is shown in parentheses for the reference area approach.

The WQC approach scores a "-" because of the need to develop values based on interstitial water measurements that are largely unavailable and cannot be routinely collected. The field-collected sediment bioassay approach scores a "0" because of relatively low costs associated with the performance of sediment bioassays (barring a requirement for conducting a large number of different bioassays for each sample). Because the PSEP database could also be used at negligible costs to provide bioassay data, a "+" rating is shown in parentheses. Both equilibrium partitioning approaches receive a score of "+" because they use the existing EPA toxicological database and would not require the collection of large amounts of additional data for their implementation (e.g., bulk sediment and water chemistry alone; a large sediment chemistry database is already available).

2.2.2. Cost of Routine Application as a Regulatory Tool

This criterion is a relative measure of costs (primarily associated with sample collection and laboratory testing) assuming that all approaches are equally implementable as regulatory tools. The WQC approach receives a score of "-" for this criterion because of higher costs and required expertise associated with the collection and analysis of interstitial water samples (a special field sample collection and laboratory analytical process). The reference area, SLC, Triad, AET, spiked sediment bioassay, and both equilibrium partitioning approaches receive a score of "0" because their implementation requires only the collection of sediment chemistry data for comparison to chemical-specific standards developed for each approach. The field-collected sediment bioassay approach is scored "+" because of the relatively low cost of sample collection and laboratory testing (barring a requirement for an extensive battery of bioassays for each sample or a particular bioassay that requires specialized expertise).

2.2.3. Ability to Develop Chemical-Specific Sediment Quality Values

Element P-2 of the Authority's Plan mandates Ecology to develop standards for identifying "sediments having adverse effects" (PSWQA 1987). These standards are to be used for a number of management objectives described in the introduction to this report. Some of these objectives require establishing relationships between contaminant concentrations in sediment and adverse biological effects (or increased human health risks), and linking sediment contamination to potential contaminant sources. These and other management objectives require that sediment standards be established on a chemical-specific basis. However, approaches that do not establish contamination-effects relationships may still be useful for identifying problem sediments.

The field-collected sediment bioassay approach is the only approach reviewed that can not be used to develop chemical-specific sediment quality values and it receives a score of "-" (all other approaches receive a "+"). In its present stage of development, the Triad approach is primarily a method for qualitative intercomparisons of sediment chemistry and biological effects information, but can be used to subjectively estimate sediment quality values for specific chemicals.

2.2.4. Ability to Develop Sediment Quality Values for a Wide Range of Chemicals

The field-collected sediment bioassay approach receives a score of "-" for this criterion because it does not incorporate sediment chemistry analyses and thus can not be used to develop chemical-specific values. The WQC approach and both equilibrium partitioning approaches are applicable only to a limited set of chemicals [i.e., only those for which WQC exist (WQC approach) and only nonionic, nonpolar compounds for which WQC exist (both equilibrium partitioning approaches)]. However, these approaches were assigned a "0" instead of a "-" because they may be applicable to a wider range of chemicals with additional development. The reference area, SLC, Triad, AET, and spiked sediment bioassay approaches are applicable to a wide variety of chemicals and receive a score of "+".

2.2.5. Current Availability of Values for a Wide Range of Puget Sound Problem Chemicals

The field-collected sediment bioassay approach is not amenable to a rating for this criterion "NA" because it can not be used to develop chemical-specific values. The SLC, Triad, WQC, spiked sediment bioassay, and both equilibrium partitioning approaches, receive a score of "-" for this criterion because either sediment quality values have not been developed for the approach or they have been used to develop only a few values specific for Puget Sound problem chemicals. The reference area and AET approaches receive a "+" score because they draw upon a large database (representing more than 300 samples for some biological indicators) already assembled during development of over 60 chemical-specific AET.

2.2.6. Incorporates Influence of Chemical Mixtures in Sediments

The reference area approach scores "-" for this criterion because it is applicable to comparisons on the basis of a single chemical alone. The WQC and both equilibrium partitioning approaches are based on tests or thermodynamic modeling that are applicable to single chemicals rather than chemical mixtures and therefore are also scored "-". The field-collected sediment bioassay, SLC, Triad, and AET approaches receive a "+" score because they all incorporate biological testing using field-collected sediments that invariably contain chemical mixtures. Although these approaches incorporate the influence of additivity, synergism, and antagonism, they do not provide a means for directly quantifying these interactive effects. Only laboratory-spiked sediment bioassays offer a systematic and reliable method for identifying and quantifying interactive effects. Hence, the spiked sediment bioassay approach also receives a "+" for this criterion; however, considerable research effort would be required to test the range of chemicals potentially occurring in the

environment (both individually and in combination), a sufficiently wide range of organisms, and a wide range of sediment matrices to establish criteria based on spiked bioassays.

2.2.7. Incorporates a Range of Biological Indicator Organisms

The reference area approach receives a score of "-" for this criterion because it relies on sediment chemistry data only. Despite its ability to incorporate a range of biological indicator organisms, the spiked sediment bioassay approach receives a "0" score because use of a range of organisms to develop values for a wide range of chemicals will likely be too costly and time consuming. All other approaches receive a "+" score because they incorporate by design, a range of biological indicator organisms.

2.2.8. Incorporates Direct Measurement of Sediment Biological Effects

The reference area approach receives a "-" score for this criterion because it does not incorporate any biological effects measurements. The WQC approach and both equilibrium partitioning approaches also receive a "-" because while they do incorporate bioassay information on aquatic organisms, they do not incorporate direct biological effects testing on sediment. The field-collected sediment bioassay, SLC, Triad, AET, and spiked sediment bioassay approaches incorporate direct measurement of biological effects of sediments and are scored "+".

2.2.9. Applicability of Predictions to Historical Sediment Chemistry Data

Historical sediment chemistry data will be one useful source of data for developing an inventory of contaminated sediments for Puget Sound after specification of sediment standards. The field-collected sediment bioassay approach does not generate chemical-specific criteria and therefore receives a score of "-" for this criterion. The WQC approach requires data on interstitial water chemistry for its application. This approach is scored "-" because there are few existing data on interstitial water chemistry in Puget Sound (especially for organic compounds). Application of both of the equilibrium partitioning approaches requires a sediment chemistry data set that includes values for TOC content. Historical data for Puget Sound sometimes lack measurement of TOC although many recent studies have routinely measured this variable. Therefore, these approaches receive a score of "0" for this criterion. Sediment quality values based on the reference area, SLC, Triad, AET, and spiked sediment bioassay approaches can be applied to virtually all historical sediment chemistry data (assuming data meet quality assurance requirements) and thus are scored "+".

2.2.10. Ease and Extent of Field Verification in Puget Sound

This criterion is a relative measure of the ease and degree to which each approach has been tested using data or environmental samples from Puget Sound. For example, field verification of the WQC approach would involve collection and analysis of interstitial water from Puget Sound locations and evaluation of chemical concentrations relative to observed and predicted (by results reported in the WQC toxicological database) biological effects. Similarly, field verification of the spiked sediment bioassay approach would consist of comparing bioassay results using Puget Sound sediments with bioassay results using sediments that have been inoculated with a suite of contaminants similar to those commonly present in the Puget Sound test sediments.

The SLC, WQC, sediment-biota equilibrium partitioning, and spiked sediment bioassay approaches have not been adequately field verified in Puget Sound and therefore receive a score of "-" for this criterion. The Triad and sediment-water equilibrium partitioning approaches have been verified to a limited extent in Puget Sound and thus receive a score of "0". The AET approach and the field-collected sediment bioassay approach (incorporated into AET) have been substantially verified in Puget Sound and receive a "+" score.

2.2.11. Proof of Cause-Effect Relationships

This criterion addresses the concern of the degree to which sediment quality values provide proof of cause/effect relationships between chemical contaminants and adverse biological effects. All of the approaches except the spiked sediment bioassay would receive a "-". The spiked sediment bioassay would receive a "+" because it is the only approach capable of directly determining cause/effect relationships. All other approaches contain confounding factors, although they may provide a preponderance of evidence of potential relationships. Even in the case of spiked sediment bioassays, proof is provided only for laboratory relationships which must then be extrapolated to potential cause-effect relationships in the field.

2.3. CONCLUSIONS AND RECOMMENDATIONS

The AET approach was selected by Ecology as the currently preferred method for developing sediment quality standards that address adverse biological effects in Puget Sound. Other approaches will still be considered as they are developed or tested, including approaches that directly address human health concerns. The primary reason for selecting AET was its relatively high reliability in classifying Puget Sound samples as "impacted" or "not impacted". The reliability of the AET has been assessed using a large database comprising samples from 13 Puget Sound embayments (all biological indicators were not available in all embayments). In at least 85 percent of the available samples for each biological indicator, the approach either correctly classifies as "impacted" samples that exhibit adverse biological effects or correctly classifies as "not impacted" samples that do not exhibit adverse biological effects. In addition to its reliability in classifying sediments, the AET approach can be used to provide sediment quality values for the greatest number and widest range of chemicals of concern in Puget Sound. The approach also incorporates the widest range of biological indicators that are directly applicable to sediment conditions.

3. IDENTIFICATION AND EVALUATION OF CRITERIA DEVELOPMENT ISSUES

In this section, three issues are evaluated with respect to the use of AET values as a basis for setting sediment quality criteria for Puget Sound. The first two issues relate to the suitability of existing AET to intertidal and low-salinity areas. Uncertainty exists because existing AET have been developed primarily using data from marine subtidal areas. The third issue is based on the implications of excluding Microtox data from compilation of the lowest AET (LAET) for a range of biological indicators, and the influence of selected chemicals of concern from PSDDA and PSEP on the reliability of LAET.

Reliability is defined in terms of the following measures, which are evaluated with actual field data (Barrick et al. 1988):

- **Sensitivity** in detecting environmental problems (i.e., are all biologically impacted sediments identified by the predictions of the chemical sediment criteria?)
- **Efficiency** in screening environmental problems (i.e., are only biologically impacted sediments identified by the predictions of the chemical sediment criteria?).

As a measure of reliability, sensitivity is defined as the proportion of all stations exhibiting a particular adverse biological effect that are correctly predicted using sediment quality values for that biological indicator. Efficiency is defined as the proportion of all stations predicted to have a particular adverse biological effect that actually are impacted. The overall reliability of any sediment criteria approach addresses both components of sensitivity and efficiency. This measure is defined as the proportion of all stations for which correct predictions were made for either the presence or absence of adverse biological effects. High reliability results from correct prediction of a large percentage of the impacted stations (i.e., high sensitivity, few false negatives) and correct prediction of a large percentage of the nonimpacted stations (i.e., high efficiency, few false positives). These measures of reliability are used as the primary means of evaluating AET in this section.

3.1. SUITABILITY OF AET VALUES IN INTERTIDAL AREAS

At present, little information exists with respect to the biological effects of sediment chemical contamination in intertidal areas of Puget Sound. In addition, there are no readily available data on tidal elevations at the stations sampled in the intertidal areas. Because intertidal areas have a number of physical/chemical and biological characteristics that differ from subtidal areas, the suitability of existing AET (developed primarily in subtidal areas of Puget Sound) was examined for application in the intertidal zone. The primary purpose of this section is to identify the major factors that could limit the application of Puget Sound AET to intertidal areas and to qualitatively evaluate the implications of this application.

3.1.1. Evaluation of Reliability Using Puget Sound Data

At present, 16 of the stations in the database used to generate Puget Sound AET are from intertidal areas in Elliott Bay and Everett Harbor (Figures 1 and 2). The amphipod mortality bioassay was conducted at all of these stations, and five were found to have significant responses. Application of AET to these stations in a predictive mode showed that AET were 80 percent sensitive (4/5 stations) in predicting known impacts and 100 percent efficient (5/5 stations) in predicting only impacted stations. Although these results are based on a small sample size, they suggest that existing AET may be useful for predicting effects in intertidal areas.

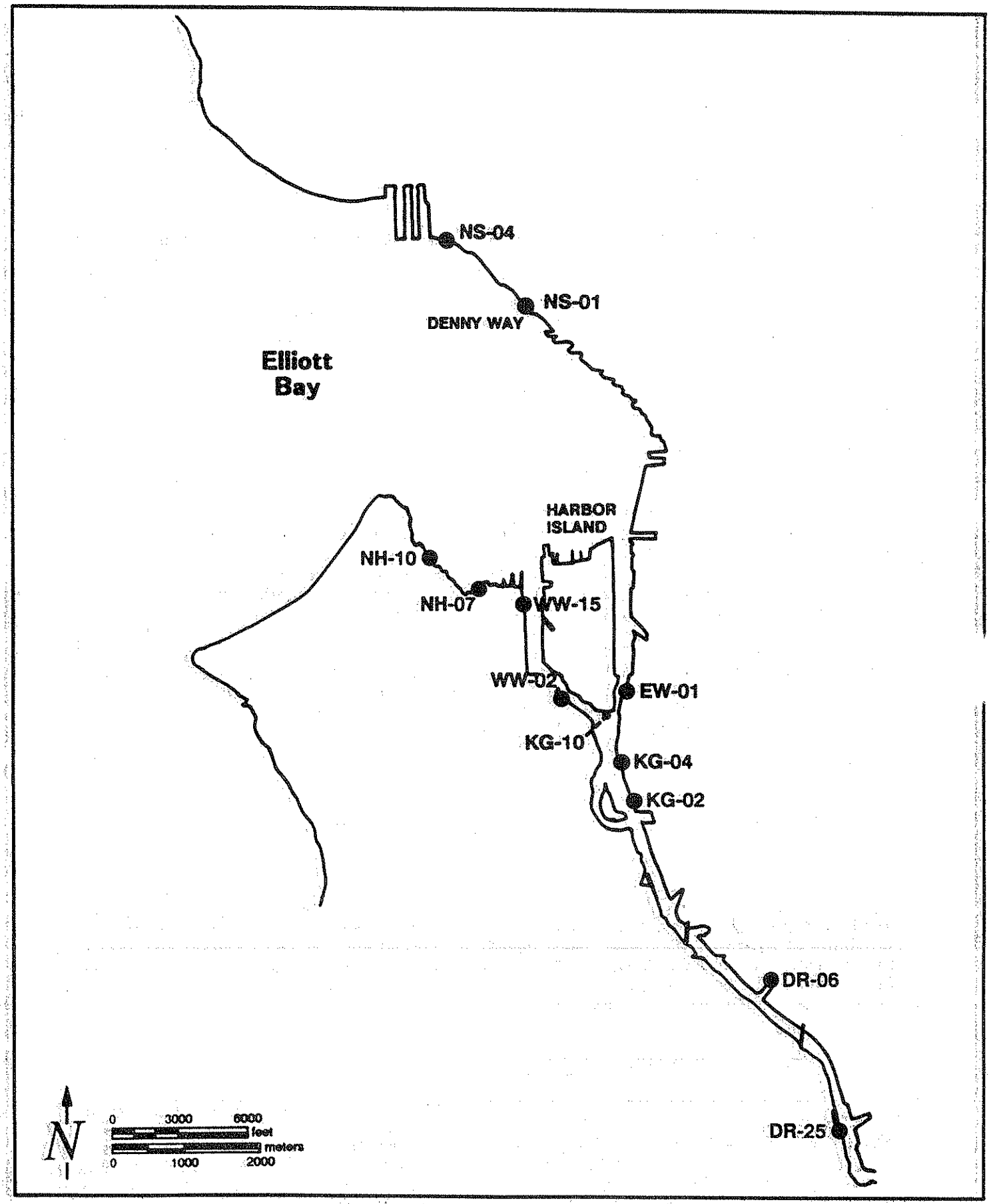


FIGURE 1. Intertidal stations in Elliott Bay (additional station location information is presented in Barrick et al. 1988).

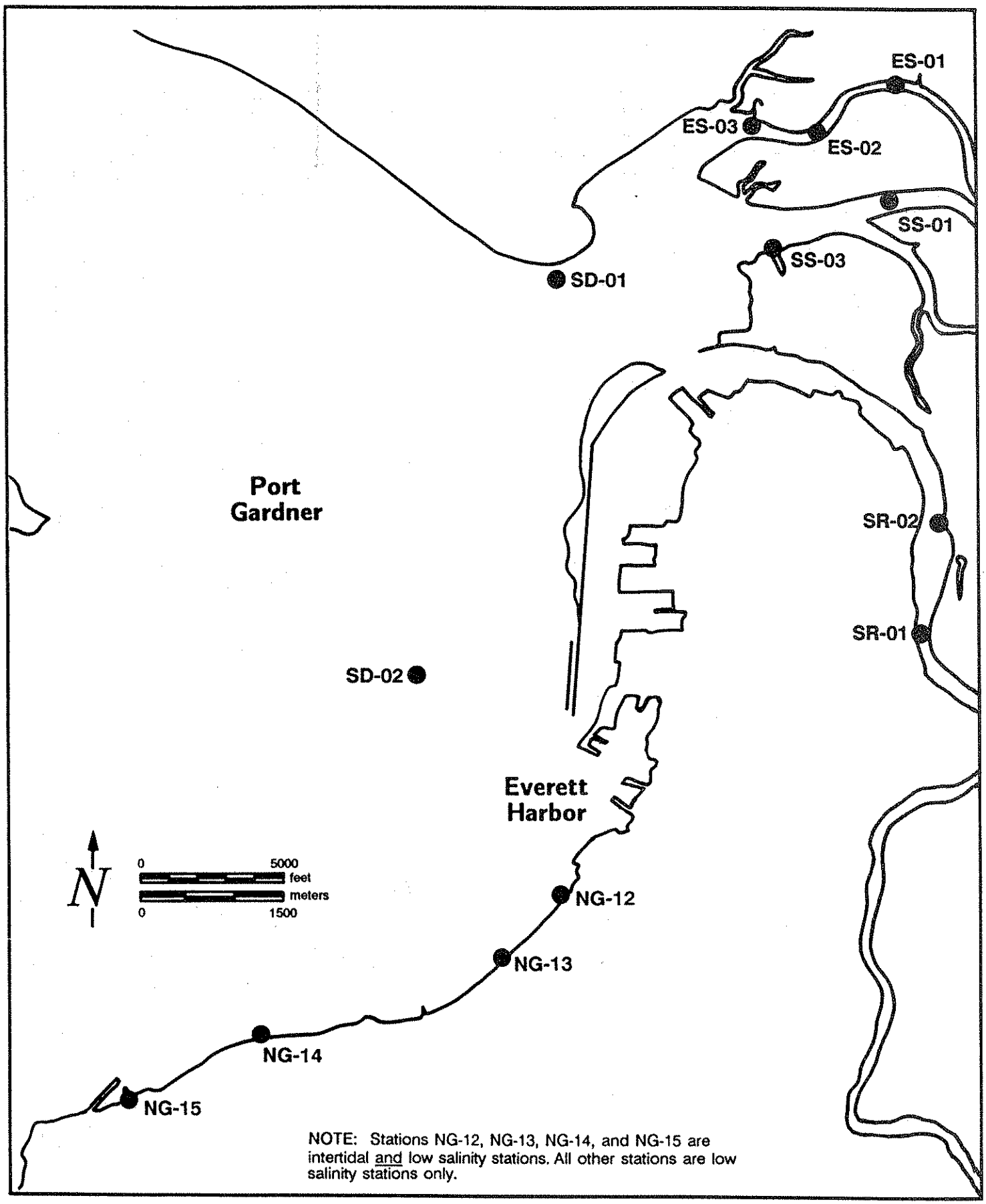


FIGURE 2. Intertidal and low salinity stations in Everett Harbor and Port Gardner (additional station location information is presented in Barrick et al. 1988).

3.1.2. Physical/Chemical Factors

Subtidal and intertidal environments display differing physical/chemical properties. These differences may alter the relationship between contaminant concentration and bioavailability, and consequently influence the applicability of AET in intertidal environments. Various organisms may be differentially sensitive to contaminants in dissolved (i.e., interstitial water) and particulate (i.e., sediments) form. Consequently, properties or processes that result in a major shift in contaminant phase association (i.e., dissolved or particulate) may affect the applicability of AET to intertidal areas.

The assumption that organisms may be differentially sensitive to dissolved- and particulate-associated contaminants is consistent with current thinking on factors controlling body burden-effects relationships, and the inferred relationship between bioavailability and the feeding habits of specific organisms (Adams 1984). A number of studies have identified a direct relationship between the concentration of a contaminant in interstitial water and the percent survival of certain organisms (summarized in DiToro 1988). There is a smaller body of evidence supporting the burden-effects relationship between the concentration of particle-associated sediment contamination and biological effects. This data gap is partly due to the complexity of matrix effects (i.e., how tightly is a chemical bound, and to what) and the numerous ways in which an organism can ingest sediment particles (e.g., bulk sediment ingestion, deposit feeding, filter feeding). It is further assumed that the labile (i.e., easily extracted) fraction of total particulate contaminants would have the greatest impact on organisms sensitive to particulate contamination.

The following properties distinguish the intertidal environment from the subtidal environment where AET have generally been applied:

- Potentially greater particle size (except in mud flats) and increased temporal and spatial variability in sediment chemistry due to increases in wave energies and tidal cycling
- Increased effects due to the microlayer and its potential interaction with intertidal organisms
- Greater potential for disequilibrium (i.e., instability) between pore water and solid phases due to tidal pumping and groundwater infiltration
- Enhanced photolytic reactions
- Enhanced gas exchange.

Intertidal environments are exposed to greater wave energy than adjacent subtidal environments, which acts to sort sediment particles by size. Exposed intertidal areas receiving high energy wave action typically contain larger particles. The labile fraction of contamination associated with large particles would be expected to be less than that of finer-grained sediments because the surface/volume ratio is much less (surfaces are where exchange takes place). The physical and chemical characteristics of the intertidal environment can be highly stratified due to the differential impacts of tidal cycling. For example, the more elevated portions of the intertidal environment would be exposed to solar radiation for a longer period of time than would less elevated sections. Similarly, the intertidal environment is more influenced by storm events. This dynamic environment should be expected to exhibit more spatial and temporal variability than subtidal environments.

Unlike subtidal sediments, the intertidal environment can be exposed to the surface microlayer, where contaminants may accumulate. The importance of the microlayer and its interaction with the intertidal environment is largely unknown. Contamination supplied to intertidal environments by shoaling of the microlayer would probably be relatively labile, and thus inherently more toxic to organisms sensitive to particulate-associated contamination.

Intertidal sediments have a greater potential for disequilibrium between interstitial water and solid phase due to tidal pumping and groundwater infiltration. Tidal pumping enhances the flow of interstitial water through sediments. This process enhances the infiltration of groundwater into the intertidal environment. The net effect of this process depends on the quality of the surface aquifer. Tidal pumping and groundwater infiltration in the vicinity of a contaminant-free upland aquifer could decrease contaminant concentrations in interstitial water, rendering them less toxic to organisms sensitive to dissolved contamination. This disequilibrium would drive contamination out of the particle phase and into the interstitial water, enhancing sediment recovery by flushing the sediments. Conversely, tidal pumping and groundwater infiltration in the vicinity of a contaminated aquifer could increase contaminant concentrations in interstitial water in intertidal areas, rendering them more toxic to organisms sensitive to dissolved contamination.

Photolytic reactions (reactions driven by solar radiation) are more prevalent in the intertidal zone. The effect of these types of reactions on intertidal sediment contamination is unknown. Photolytic reactions are known to enhance the degradation of some contaminants and thereby reduce their toxicity, but it is also possible that photolytic degradation products could be more toxic than the original compound. Gas exchange would also be enhanced. This loss would, however, be reflected in the chemical measurement of contamination in sediments. Photolytic reactions and gas exchange would probably not affect the applicability of AET.

Differences in the bioavailability of chemicals can influence the setting of AET values in different environments. However, the degree to which bioavailability differs in these environments is not directly quantified using the AET approach. The net effect on bioavailability of these processes characteristic of the intertidal zone may be minor or site-specific. As with low-salinity environments, it is likely that the different contaminated matrices to which AET have been applied in subtidal environments represent a much broader range of matrix types, and consequent variations in bioavailability than do differences between subtidal and intertidal environments. This suggests that existing AET may be directly applicable to the intertidal environment. The preliminary Puget Sound verification results described in the previous section support this contention. Ongoing review of any additional verification data is recommended.

3.1.3. Biological Factors

From a biological perspective, the intertidal zone represents a highly variable and complex environment (Smith and Carlton 1975). As a result, intertidal species assemblages differ markedly from the assemblages found in the more stable subtidal environments. In general, intertidal assemblages are comprised of relatively hardy species that can tolerate the large fluctuations in environmental conditions experienced in the intertidal zone. In addition, the hard substrates (e.g., rocks, cobble, gravel) found in some intertidal areas provides a substrate for "hard-bottom" species that generally is unavailable in most subtidal areas. Some of the major environmental factors that influence intertidal organisms include:

- Kind of substrate (e.g., rocks, cobble, gravel, sand, mud)
- Degree of wave shock and current action
- Variable temperature and salinity
- Exposure to air (i.e., threat of desiccation)
- Exposure to predators.

Given the differences in environmental conditions and biological assemblages between intertidal and subtidal environments, it is uncertain whether AET developed in subtidal areas are applicable to intertidal habitats.

At present, little information exists with respect to the effects of toxic chemicals on intertidal benthic macroinvertebrate assemblages or on sediment bioassays conducted using intertidal

sediments. Because intertidal organisms are adapted to the stresses of a more variable environment than subtidal organisms, they may be more resistant to additional stresses caused by chemical toxicity than the latter organisms. If this is the case, then the use of AET based on subtidal assemblages to predict effects on intertidal organisms should be a conservative approach. That is, the approach should be environmentally protective because it would use AET values that overestimate the toxicity of intertidal sediments to resident organisms. However, a potential limiting factor to applying subtidal AET to intertidal areas is the potential differences in bioavailability of contaminants between the two habitats (see earlier discussion of physical/chemical factors). Thus, subtidal AET may not be environmentally protective if contaminants are more bioavailable in the intertidal zone than in subtidal areas.

Two of the three sediment bioassays on which AET are based use test species that occur naturally in both intertidal and subtidal environments. *Rhepoxynius abronius*, the test species used for the 10-day amphipod mortality bioassay, occurs predominantly in subtidal areas along the North American Pacific coast and in both subtidal and intertidal areas in the estuaries of Oregon and Washington (Kemp et al. 1985). The Pacific oyster (*Crassostrea gigas*), the test species used for the 48-hour bivalve abnormality bioassay, is also found in both intertidal and shallow subtidal areas during post-larval stages (Smith and Carlton 1975), although the larval stage used for the bioassay generally is found only in the water column. Therefore, it appears appropriate to use these species as representatives of intertidal organisms in general.

3.1.4. Summary

The major conclusions of this section are:

- Different physical and chemical processes operating in the intertidal environment are predicted to both increase (e.g, potentially greater influence of microlayer contaminants) and decrease (e.g., generally increased particle size) the labile fraction of particle bound contamination relative to conditions in the subtidal.

- It is likely that the different contaminated matrices to which AET have been applied in the subtidal environment represent a broader range in matrix types, and associated variations in bioavailability, than do differences between subtidal and intertidal environments.

- Physical and chemical conditions in the intertidal environment suggest that natural recovery will be enhanced relative to the subtidal environment by a number of factors that are generally unique to the intertidal, including direct exposure to solar radiation (i.e., increased potential for photodegradation reactions), potentially enhanced groundwater flushing, and enhanced sediment reworking.

- Intertidal biological assemblages generally differ markedly from subtidal assemblages.

- In general, intertidal assemblages are comprised of relatively hardy species that can tolerate the large variations in environmental conditions often experienced in the intertidal zone.

- Although little is known of the effects of toxic chemicals on intertidal organisms, they may be more tolerant than subtidal organisms because they are adapted to a more variable (i.e., stressful) environment.

- If intertidal organisms are more tolerant of chemical toxicity than are subtidal organisms, use of AET developed using subtidal organisms may be a conservative approach to identifying potential problems.

- Because two of the test species used to develop existing AET occur in intertidal areas (i.e., *Rhepoxynius abronius, Crassostrea gigas*), it may be appropriate to use these species as representatives of intertidal organisms in general.

- Existing AET are recommended for use at intertidal stations in Puget Sound based on preliminary reliability studies. Ongoing review of any additional verification data is recommended. Based on the existing reliability results and the small area relative to subtidal sediments in Puget Sound, development of separate standards for intertidal sediments is not recommended as an efficient regulatory approach.

3.2. SUITABILITY OF AET VALUES TO LOW-SALINITY ENVIRONMENTS

Little information was found with respect to the biological effects of sediment chemical contamination in low-salinity areas of Puget Sound. Because these areas have a number of physical/chemical and biological characteristics that differ from marine areas, the suitability of existing AET (developed primarily in marine areas of Puget Sound) was examined for application to low-salinity environments. The primary purpose of this section is to identify the major factors that could limit the application of Puget Sound AET to low-salinity areas and to qualitatively evaluate the implications of this application.

In determining the area (or salinity range) in which AET are most applicable, it is important to recognize that surface water salinity measurements may not represent conditions in overlying bottom sediments. Differences in these two regimes arise from the presence of dense, high-salinity waters that can be overlain by low-salinity water masses or freshwater lenses. This condition would be encountered in the vicinity of rivers or other freshwater sources, and would be most pronounced during conditions of high flow (i.e., spring or summer runoff). The interstitial water salinity of the underlying sediments would most closely approximate that of the bottom high-salinity water, not the surface low-salinity water.

3.2.1. Evaluation of Reliability Using Puget Sound Data

At present, 13 of the stations available in the database used to generate Puget Sound AET were from areas with an interstitial salinity $\leq$25 ppt in Everett Harbor (Figure 2). Interstitial water salinity at these stations ranged from 8 to 25 ppt and averaged 18.8 ppt. None of these stations exhibited significant adverse biological effects, so sensitivity and efficiency could not be calculated. However, because none of these stations were predicted to exhibit adverse effects on the basis of Puget Sound AET, it can be concluded that AET were efficient in predicting the absence of adverse effects.

3.2.2. Physical/Chemical Factors

Low-salinity and marine waters differ markedly in certain physical/chemical properties. These differences may potentially alter the relationship between contaminant concentration and bioavailability, and consequently influence the applicability of AET (which were developed for marine environments) to low-salinity environments. Identification of those properties that are expected to influence bioavailability requires that the relationship between chemical concentration and biological effects be conceptualized. For the purpose of this discussion, it is assumed that various organisms may be differentially sensitive to contaminants in dissolved (i.e., interstitial water) and particulate (i.e., sediments) form. (The rationale behind this assumption is presented in Section 3.1.2.)

The following changes in physical/chemical properties occur with decreases in salinity:

- Decreased ionic strength

- Altered dissolved metal speciation

- Decreased particle affinity of neutral organic compounds
- Decreased buffering capacity (and thus greater sensitivity to pH shifts)
- Increased flocculation (at the freshwater/seawater interface).

Ionic strength is a measure of the concentration of charged ions in water. Decreases in ionic strength increases the activity (i.e., the effective concentration) of dissolved metals. The concentration of anions that complex with metals (e.g., sulfate, chloride) also decreases as ionic strength decreases, further increasing metal activity. Changes in dissolved metal speciation may be counterbalanced by the interaction of dissolved metals with particles, which is a function of metal activity. Increases in metal activity would tend to drive the metal into particulate phase, reducing total dissolved metal concentration. The net effect of decreasing ionic strength is difficult to assess and would probably depend on the relative sensitivity of an organism to dissolved and solid-phase contamination.

The interaction of organic chemicals with particles is also influenced by decreases in ionic strength. In general, the affinity of a neutral organic molecule for a particle decreases with decreasing ionic strength. This means that at a constant sediment concentration, a greater portion of the organic chemical would be present in interstitial water in an estuarine environment than in a marine environment. In general, this effect is relatively minor; the partition coefficient (a measure of particle affinity) of neutral organic compounds would typically change by only 20 percent (Karickhoff 1984).

Colloids are very small particles that are not influenced by gravity. As colloidal river-borne material is discharged to the marine environment, a process known as flocculation takes place. Flocculation is the aggregation of colloidal material as freshwater mixes with seawater. Differences in ionic strength (electrical charge) between fresh and salt water cause changes in the charges of the colloids and they attach to one another and settle out of the water column. While this process may remove contaminants from the water column, it also creates a relatively labile particulate form that may be more bioavailable to organisms sensitive to solid-phase contamination. The mass of these flocculated sediments is probably small relative to the total mass of deposited sediments.

The increased susceptibility to pH shifts is a potentially important property of low-salinity environments. Seawater is relatively resistant to changes in pH because it is well buffered by carbonate alkalinity. Depending on the geology of the drainage basin, freshwater can also be well buffered (e.g., if source waters are exposed to carbonate rocks); however, this is not the case for most rivers that drain into Puget Sound. The greatest sensitivity to pH changes would be observed at lower ionic strength. The particle affinity of contaminants would generally decrease in low-salinity environments exhibiting a pH less than the range typically observed in the marine environment [i.e., (<7.0 to >8.5)]. The pH actually observed in low-salinity water would depend on the type of waste material present (e.g., acids or bases discharged to the area) and the processes of photosynthesis, respiration, and degradation of organic matter. The importance of decreased buffer capacity could be determined by pH measurements.

It is likely that the different contaminated matrices to which AET have been applied in marine environments represent a much broader range of potential matrices (e.g., sewage, petroleum, creosote, ores, slag, sandblast grit, and chemical wastes) potentially affecting bioavailability than do differences between marine and low-salinity waters. Based on these considerations and the relatively few data available for low-salinity stations, existing AET may be applicable to low-salinity sediments, but the uncertainty is much greater than for intertidal sediments.

3.2.3. Biological Factors

Species distributions usually exhibit substantial alterations as salinity changes from marine to estuarine to freshwater conditions (Gross 1972). The effects of varying salinity on the physiology of organisms is a primary limiting factor, but variations in other physical/chemical conditions such

as temperature, pH, and dissolved oxygen can also be contributing limiting factors. In general, the number of species inhabiting aquatic habitats declines with decreasing salinity from a maximum in marine areas to a minimum in estuarine areas, and then increases in freshwater areas (Gross 1972). However, species abundances in estuarine areas can surpass those in marine areas in response to the higher levels of nutrients frequently found in estuarine environments. Given the differences in environmental conditions and biological assemblages between marine, estuarine, and freshwater environments, it is uncertain whether AET developed in marine areas are applicable to low-salinity habitats.

At present, little information exists in Puget Sound with respect to the effects of toxic chemicals on benthic invertebrate assemblages or sediment bioassays from low-salinity areas. Because organisms in these areas are adapted to the stresses of a more variable environment than marine organisms, they may be more resistant to additional stresses caused by chemical toxicity than the latter organisms. If this is the case, the use of AET based on marine organisms should be protective for organisms in low-salinity habitats, assuming that contaminant bioavailability is not enhanced in those areas relative to marine habitats.

Of the three sediment bioassays on which AET are based, only the oyster larvae abnormality and Microtox tests are based on organisms that normally occur in low-salinity waters. The bacterium *Photobacterium phosphoreum* can tolerate salinities as low as 5 ppt (Holt 1977). In addition, the Pacific oyster can tolerate salinities considerably lower than those in marine areas (Kozloff 1983). Because these organisms are found in low-salinity areas, they may serve as appropriate indicator species.

By contrast with the previous two species, *R. abronius*, the test species used for the 10-day amphipod mortality bioassay, usually is stressed at salinities less than 25 ppt (Swartz et al. 1985a). Therefore *R. abronius* is not expected to occur at salinities much less than this value. DeWitt et al. (in review) recently compared the toxicity of fluoranthene to *R. abronius* (a marine/estuarine species), the estuarine species *Eohaustorius* sp., and the freshwater species *Hyalella azteca*. *R. abronius* was the most sensitive species (LC_{50} = 6.6 mg/g), *H. azteca* was the least sensitive species (LC_{50} = 21.2 mg/g), and *E. estauarius* was intermediate in sensitivity (LC_{50} = 13.8-17.5 mg/g). These results suggest that AET based on *R. abronius* may be protective of biota from low-salinity areas.

3.2.4. Summary

The major conclusions of this section are:

- The physical/chemical changes that accompany decreases in salinity include the following:
  - Decreased ionic strength
  - Altered dissolved metal speciation
  - Decreased particle affinity of neutral organic compounds
  - Decreased buffering capacity (and thus greater susceptibility to pH shifts
  - Increased flocculation (at the freshwater/seawater contact zone).
- The net effect of decreases in salinity on contaminant bioavailability cannot be predicted because the physical/chemical processes that accompany these changes are complex and in some cases poorly understood.
- It is likely that the different contaminated matrices to which AET have been applied in the subtidal environment represent a broader range in matrix types, and associated variations in bioavailability, than do differences between subtidal and low-salinity environments.
- Biological assemblages in low-salinity habitats generally differ markedly from assemblages in marine habitats.

- In general, assemblages in low-salinity habitats are comprised of relatively hardy species that can tolerate the large variations in environmental conditions often experienced in those environments.

- Although little is known of the effects of toxic chemicals on organisms from low-salinity habitats, it might be surmised that they are more tolerant than marine organisms because they are adapted to a more variable (i.e., stressful) environment.

- If organisms from low-salinity habitats are more tolerant of chemical toxicity than are marine organisms, use of AET developed using marine organisms may be a conservative approach to identifying potential problems.

- Because two of the test species used to develop existing AET occur in low-salinity areas (i.e., *Photobacterium phosphoreum*, *Crassostrea gigas*), it may be appropriate to use these species as representatives of organisms from low-salinity habitats in general.

- Almost no information exists on the reliability of AET (which are derived in saline environments) when they are applied to low-salinity environments. Until verification studies have been conducted, application in low-salinity environments is not recommended.

The available field data in low-salinity environments consists exclusively of stations that were low in chemical contamination and did not exhibit biological effects. Before further application, it is recommended that the AET approach be field verified in low-salinity environments, especially in areas approaching freshwater conditions and in areas in which chemical contamination is of concern. A field verification program to characterize the applicability of AET in low-salinity environments is recommended (see Section 3.4).

3.3. RELIABILITY TESTS FOR ALTERNATIVE LAET

Using the LAET for available biological indicators on a chemical-by-chemical basis is a means for generating a set of AET that should be protective for that range of indicators. Reliability tests were performed to examine 1) the influence of Microtox AET values on LAET reliability, 2) the influence of PSDDA chemicals of concern that do not have established AET, and 3) the influence of chemicals observed in Puget Sound that are not included on the PSDDA list.

3.3.1. Microtox Bioassay Effect on LAET Reliability

The Microtox bioassay has been considered for exclusion from LAET generation. The effect of the exclusion of these data was evaluated to determine the contributions of the Microtox bioassay to LAET reliability for other indicators, and to determine if the resulting LAET would still be predictive of Microtox results. Initially, LAET were generated using the entire 334 sample Puget Sound database (287 amphipod bioassay stations, 56 oyster larvae bioassay stations, 50 Microtox bioassay stations, and 201 benthic infauna stations). The Microtox AET uniquely established the LAET for approximately 18 percent of the chemicals included in this evaluation. The results are presented in Table 3.

Excluding Microtox data from the LAET resulted in relatively small decreases in sensitivity and increases in efficiency for all but the Microtox bioassay stations, at which the opposite trend was observed. Sensitivity decreased from 99 to 88 percent for benthic infauna stations, from 94 to 86 percent for amphipod bioassay stations, and from 93 to 86 percent for Microtox bioassay stations. Sensitivity did not change for oyster larvae bioassay stations. Efficiency decreased from 73 to 68 percent for Microtox bioassay stations, and increased from 61 to 67 percent for benthic

TABLE 3. CONTRIBUTIONS OF MICROTOX TO RELIABILITY OF LAET

Test Type	LAET with Microtox		LAET without Microtox	
	Sensitivity	Efficiency	Sensitivity	Efficiency
Benthic Infaunal Abundance **(n=201)**	93% (99/107)	64% (99/154)	87% (93/107)	74% (93/126)
	Overall Reliability[a]: 69%		Overall Reliability: 77%	
Amphipod Mortality Bioassay **(n=287)**	86% (91/106)	44% (91/206)	81% (86/106)	50% (86/172)
	Overall Reliability: 55%		Overall Reliability: 63%	
Microtox Bioassay **(n=50)**	93% (27/29)	77% (27/35)	79% (23/29)	74% (23/31)
	Overall Reliability: 80%		Overall Reliability: 72%	
Oyster Larvae Abnormality Bioassay **(n=56)**	94% (16/17)	40% (16/40)	88% (15/17)	47% (15/32)
	Overall Reliability: 55%		Overall Reliability: 66%	

[a] Overall reliability was calculated as the sum of the number of correctly predicted impacted stations and the number of correctly predicted non-impacted stations divided by the total number of stations.

infauna stations, from 41 to 47 percent for amphipod bioassay stations, and from 40 to 48 percent for oyster larvae bioassay stations (Table 3).

When the Microtox bioassay was excluded from LAET generation, overall reliability increased by up to 11-15 percent for the amphipod bioassay, oyster larvae bioassay, and benthic infauna stations (Table 3). The overall reliability for the Microtox bioassay stations was relatively unchanged (3 percent decrease). Of these three measures of reliability, sensitivity focuses exclusively on the correct prediction of impacted stations, which is most relevant to sediment quality values such as LAET that are designed to be environmentally protective of impacts. Based on decreases in sensitivity, and small increases in efficiency and overall reliability, the exclusion of Microtox results from LAET generation is not strongly warranted.

3.3.2. Chemicals of Concern Lacking AET Values

This section evaluates the influence of PSDDA chemicals of concern on the reliability of sediment quality values. Chemicals may not have established AET if they are infrequently detected in the existing AET database or if the highest concentration of a chemical occurs at a nonimpacted station (resulting in a ">", or preliminary AET value) (Barrick et al. 1988). In this evaluation, LAET were used whenever available, and PSDDA screening levels were used for chemicals without established AET. The contribution of non-AET PSDDA chemicals of concern to reliability was evaluated by comparing the sensitivity and efficiency of two sets of sediment quality values:

- LAET for the 51 PSDDA chemicals of concern that have established AET
- LAET and screening level values for all 58 PSDDA chemicals of concern, using PSDDA screening level values for those chemicals without established AET.

Of the 58 PSDDA chemicals of concern, aldrin, chlordane, dieldrin, heptachlor, lindane (gamma-HCH), trichloroethene and hexachloroethane are all without defined AET. Screening level values for trichloroethene and hexachloroethane were calculated as 10 percent of the maximum level value set by the sediment-water equilibrium partitioning approach (Phillips et al. 1988), and the pesticide screening levels were set to 5 times an assumed analytical detection limit of 2 ug/kg dry weight sediment. The stations with concentrations exceeding these chemical screening levels also exceeded at least one LAET value. Therefore, there was no difference between the reliabilities of the two chemical sets identified above (Table 4).

3.3.3. Other Potential Chemicals of Concern

An additional test was conducted to evaluate the effects on reliability of augmenting the PSDDA chemicals with PSEP chemicals that were detected in greater than 5 percent of the Puget Sound sediment samples in the existing AET database, and for which AET have been established. A combined list of the PSDDA chemicals of concern (using established AET plus screening levels for those non-AET chemicals listed in Section 3.3.2) and the PSEP chemicals of concern yielded the same results as the PSDDA chemicals alone (Table 4). The only chemical added from the PSEP list, according to the considerations listed above, was 2-methoxyphenol (guaiacol). Guaiacol made no unique contribution to the reliability of the LAET data set because the three stations that had concentrations above the LAET for guaiacol were accounted for by LAET of other chemicals. Chemicals used in the tests described above are listed in Table 5.

An assessment was conducted of the effects on reliability of adding chemicals that are not on the PSDDA or PSEP lists but for which 1988 AET values exist (see Appendix B). Non-PSDDA and non-PSEP chemicals (used in calculation of sensitivity and efficiency shown in Table 3) result in an increase of 0 to 7 percent in sensitivity and decrease of 0 to 6 percent in efficiency relative to the sensitivity and efficiency shown in Table 4 (in which these chemicals were excluded from reliability tests).

TABLE 4. LAET[a] RELIABILITY FOR COMBINATIONS OF PSDDA AND PSEP CHEMICALS OF CONCERN

Test Type	Sensitivity		Efficiency	
51 PSDDA Chemicals of Concern That Have 1988 AET				
Benthic Infaunal Abundance	92%	(98/107)	61%	(98/161)
Amphipod Mortality Bioassay	91%	(96/106)	42%	(96/227)
Microtox Bioassay	90%	(26/29)	79%	(26/33)
Oyster Larvae Abnormality Bioassay	94%	(16/17)	43%	(16/37)
58 PSDDA Chemicals of Concern (using SL values for those not included above)[b]				
Benthic Infaunal Abundance	92%	(98/107)	61%	(98/161)
Amphipod Mortality Bioassay	91%	(96/106)	42%	(96/227)
Microtox Bioassay	90%	(26/29)	79%	(26/33)
Oyster Larvae Abnormality Bioassay	94%	(16/17)	43%	(16/37)
Combined List of the PSDDA and PSEP Chemicals of Concern[c]				
Benthic Infaunal Abundance	92%	(98/107)	61%	(98/161)
Amphipod Mortality Bioassay	91%	(96/106)	42%	(96/227)
Microtox Bioassay	90%	(26/29)	79%	(26/33)
Oyster Larvae Abnormality Bioassay	94%	(16/17)	43%	(16/37)

[a] Lowest AET including Microtox bioassay data.

[b] Adds aldrin, chlordane, dieldrin, heptachlor, gamma-hexachlorocyclohexane (lindane), hexachloroethane, and trichloroethene.

[c] Adds 2-methoxyphenol.

TABLE 5. (Continued)

Chemical	PSDDA	PSEP	AET Data Set
Miscellaneous Nonpolar Compounds (continued)			
Hexachloroethane	X		X[b]
N-Nitrosodiphenylamine	X	X	X
2,3,7,8-TCDD (Dioxin)		X	
Cymene isomer[a]			X
Retene[a]			X
Kaur-16-ene (or related diterpenoid hydrocarbon)[a]			X
Isopimaradiene[a]			X
Diterpenoid hydrocarbon (dehydroabietane?)[a]			X
1,2,4-Trithiolane[a]			X
Volatile Organic Compounds			
Chloroform		X	X[b]
Trichloroethene	X	X	X
Tetrachloroethene	X	X	X
Ethylbenzene	X	X	X
Total xylenes	X		X
Pesticides			
p,p'- DDT[c]	X	X	X
p,p'- DDD[c]	X	X	X
p,p'- DDE[c]	X	X	X
Aldrin	X	X	X[b]
Chlordane	X		X[b]
Dieldrin	X	X	X[b]
Heptachlor	X		X[b]
Lindane	X	X	
Total PCBs	X	X	X
SEDIMENT CONVENTIONALS			
Sulfides			X
Total Organic Carbon			X
Total Volatile Solids			X
Nitrogen			X
Cyanide		X	
ORGANOMETALLIC COMPOUNDS			
Organotin Complexes		X	

[a] Tentatively identified organic compound.

[b] Detected fewer than 6 times in AET database.

[c] Total DDT, a parameter under the PSDDA program, is a sum of p,p'-DDT, p,p'-DDE, and p,p'-DDD.

A study of AET (Barrick et al. 1988) indicates that certain tentatively identified organic (TIO) compounds and other chemicals not included on the PSDDA or PSEP lists may contribute to sensitivity of AET. Although additional chemicals did not greatly improve the performance of LAET, it is recommended that chemicals that may be important near certain kinds of sources (e.g., guaiacols and dehydroabietic acid near pulp mill discharges) be considered for sediment quality management in localized areas. Because concentrations of TIO compounds are estimated rather than rigorously quantified, their use in developing sediment quality standards is not recommended until data with more rigorous quantification are available.

3.4. CONCLUSIONS AND RECOMMENDATIONS

The major conclusions of this section are:

- Based on available data, existing AET are recommended for use in both subtidal and intertidal environments of Puget Sound

- Based on a smaller database and greater uncertainty in their applicability, existing AET are not recommended for use in low-salinity environments (e.g., <25 ppt salinity) without further field verification

- A field verification program to characterize the applicability of AET in low-salinity environments should incorporate the following considerations:

  - Biological and chemical samples should be collected synoptically over a wide range of chemical concentrations and should include determination of interstitial water salinity and bottom-water salinity (i.e., salinity within the 1 meter of water overlying the sediments)

  - Sediment samples should be collected in relatively stable low-salinity environments for comparison with those collected in areas with considerable temporal variability in salinity (e.g., samples at the entrance of river deltas where bottom salinities may be depressed but are relatively constant, compared with samples near the upper limits of the tidal excursion where salinities regularly change

  - Environments with potentially confounding factors should be avoided (e.g., areas with unrepresentative grain size distributions and erosional environments)

  - The number of samples collected specifically for low-salinity environments should correspond to recommendations for AET development in general (e.g., preferably greater than 50 samples; Barrick et al. 1988)

  - Interstitial water pH should also be monitored to determine if pH extremes (<7 or >8.5) are present.

4. ROLE OF BIOLOGICAL TESTING IN DEVELOPING SEDIMENT QUALITY STANDARDS

4.1. INTRODUCTION

As mentioned in Section 1, Element P-2 of the 1987 Plan specifies that sediment quality standards be developed for identifying and designating sediments having observable acute or chronic adverse effects on biological resources. The Plan also specifies that these standards may use physical, chemical, and biological tests with clearly identified pass/fail criteria. As mentioned in Section 2, the AET approach has been proposed for use in developing sediment quality standards for Puget Sound. AET values can be modified to identify chemical concentrations below which adverse biological effects would rarely be expected. Biological testing can be useful for accurately classifying sediments having chemical concentrations above the screening levels based on these modified sediment quality values.

In this section, the potential role of biological testing in meeting the specifications of the Plan is presented. Discussions are included on 1) the general characteristics of the two major kinds of biological tests commonly used to assess adverse effects in sediments, 2) the characteristics of the specific tests considered available for immediate use in Puget Sound, and 3) a scheme for classifying sediments in Puget Sound with respect to the presence or absence of adverse biological effects.

Biological testing of field sediments has a number of advantages over strictly chemical measurements for assessing the environmental effects of sediment chemical contamination. First, it allows an evaluation of the potential effects of chemicals for which standards are not available and chemicals that may not be measured during typical assessments. Second, it allows an assessment of the effects of complex chemical mixtures and thereby accounts for interactions among chemicals (e.g., synergistic, antagonistic). Finally, biological testing provides an empirical assessment based on the actual bioavailability of chemicals and the manner in which they actually behave in the environment (e.g., dilution, complexing with organic matter), which cannot always be predicted on a theoretical basis.

In most cases, it is recommended that biological testing of field sediments be supplemented with measurements of individual chemicals (U.S. EPA 1985b). Chemical measurements are particularly useful for evaluating specific kinds of problem chemicals (e.g., carcinogens, bioaccumulative substances), for comparing observed sediment concentrations with existing criteria or standards, and for identifying the potential sources of contaminating chemicals.

Two major kinds of biological tests are commonly used for environmental assessment of chemical contamination: sediment bioassays and evaluations of indigenous biota. Sediment bioassays involve the controlled exposure of test organisms (usually a single sensitive species) to test sediment for a fixed period of time. Although bioassays can be conducted *in situ*, most are conducted in the laboratory. Bioassays have at least two major advantages over evaluations of indigenous biota. First, because most experimental conditions can be controlled during bioassays (e.g., temperature, dissolved oxygen, lighting, sediment grain size, predation), measured effects can be attributed to chemical toxicity (i.e., the uncontrolled variable of interest) with reasonable confidence. Second, bioassays generally are considerably less expensive than evaluations of indigenous biota. A major disadvantage of most sediment bioassays is the lack of knowledge as to how the results apply to assemblages of diverse species under variable field conditions (Long and Chapman 1985; Swartz et al. 1985b; Chapman et al. 1987). Part of this uncertainty can be evaluated by comparing bioassay responses with effects on indigenous biota.

Evaluations of sediment toxicity to indigenous biota can involve any kind of organism, but usually are focused on benthic macroinvertebrates. These organisms are preferred because they

live in close contact with bottom sediments, are relatively stationary, can be sampled quantitatively, and have been found to exhibit predictable patterns in response to environmental stress. Unlike bioassays, many environmental variables cannot be controlled during evaluations of indigenous biota. The relationship between measured effects on indigenous biota is therefore less certain than for bioassays. However, because effects on indigenous biota are measured in the field, there are no limitations encountered with extrapolating laboratory results to field situations, as is encountered with laboratory bioassays.

4.2. GENERAL CHARACTERISTICS OF BIOLOGICAL TESTS

In this section, the general characteristics of sediment bioassays and evaluations of indigenous organisms are discussed. Emphasis is placed on presenting the advantages and disadvantages of the major aspects of each kind of test with respect to use for regulatory purposes. The specific aspects of candidate tests available for use in Puget Sound are discussed in the following section (see Section 4.3).

4.2.1. Sediment Bioassays

In general, sediment bioassays can be divided into two categories with respect to length of the exposure period to contaminated sediment: acute (or short-term) and chronic (or long-term). Although the critical exposure period that distinguishes between the two categories varies throughout the United States, there is general agreement in Puget Sound that acute tests are ≤ 10 days in length and chronic tests are >10 days in length (Tetra Tech and E.V.S. 1986). Both acute and chronic bioassays can be based on endpoints (i.e., measured effects) that are lethal (i.e., death) or sublethal (e.g., reduced growth, impaired reproduction). This section describes the desirable characteristics of bioassay species, the major kinds of available acute and chronic bioassays, and a method of relating the results of the two kinds of bioassays based on an acute-to-chronic ratio (ACR).

Species Selection—Bioassays can be conducted on almost any species. However, to provide meaningful results at a reasonable cost it is desirable that the test species be readily available, capable of being held in captivity, responsive to chemical toxicity, and representative of some component of the ecosystem being protected. Two additional considerations that help ensure protective, interpretable results are that the species be particularly sensitive to chemical toxicity and that standardized testing protocols be available.

Different species often exhibit sensitivities to toxic chemicals that vary by as much as several orders of magnitude (U.S. EPA 1985b). In addition, different life stages (e.g., embryo, larvae, juvenile, adult) and different bioassay endpoints (e.g., mortality, growth, reproduction) may also exhibit considerable differences in sensitivity within the same species. To be environmentally protective, it is desirable that bioassays be conducted using the more sensitive species, life stages, and endpoints. It is assumed that standards based on the more sensitive criteria will be protective of less sensitive species, life stages, and endpoints.

Considerable effort is generally required to develop standard bioassay protocols (i.e., methods that have been well-defined, tested, and peer-reviewed). It is also desirable that both intra-laboratory and interlaboratory comparisons be conducted to ensure that each test can be routinely conducted with adequate precision. Standardized tests are therefore available for only a limited number of species. Ideally, the test species should be a resident of the area of interest. However, bioassays using nonresident species may be appropriate if they are adequately standardized and sensitive and no comparable test is available using a resident species (U.S. EPA 1985b). The use of an unstandardized test with a resident species generally is not recommended because such tests can be difficult to conduct, highly variable, and difficult to interpret. For example, if protocols differ among studies, observed differences in bioassay responses may be largely due to methodological variations rather than variations in chemical toxicity.

Acute Sediment Bioassays—Acute bioassays are conducted during relatively short exposure periods and measure the kinds of biological effects one might expect to find following a brief exposure to toxic chemicals. Because the exposure period is short, it often is thought that chemical toxicity must be high to elicit bioassay responses. Although this is frequently true, relatively low levels of chemical toxicity can elicit an acute response if the bioassay test organism and endpoint are exceptionally sensitive to chemical contamination.

Because of their relatively short duration, acute tests can usually be conducted under static conditions. That is, the seawater in the test chambers is not replaced during the experiment. In some cases, acute tests are conducted under static-renewal conditions, in which a portion of the seawater in each test chamber is replaced manually at one or more fixed time intervals. The use of either static or static-renewal systems allows testing to be conducted with a minimum of equipment and maintenance and thereby keeps costs relatively low. In addition, use of these systems does not require that laboratories have direct access to seawater and thereby allows a relatively large number of laboratories to conduct the tests.

Although a wide variety of biological endpoints can be used for acute bioassays, a limited number has been commonly used throughout the United States. Results from many of these tests have been compared with effects on indigenous biota and are ready as technical tools for routine regulatory applications. The most common acute endpoints include:

- Adult mortality
- Larval abnormality
- Genotoxicity (i.e., damage to genetic material)
- Reductions of bacterial bioluminescence.

Adult mortality tests have been applied to various benthic invertebrates, but particularly to amphipods, a group of pollution-sensitive benthic invertebrates (Bellan-Santini 1980). These tests have an unambiguous endpoint (i.e., death). In addition, their ecological significance is relatively certain. If adult organisms cannot survive in an environment, then it is likely that severe alterations of benthic assemblages will be found. Although adult mortality may not be considered an environmentally protective endpoint, the use of a sensitive species can partially compensate for this potential limitation.

Larval abnormality tests have generally been applied to the early life stages of bivalve molluscs and echinoderms. Because the determination of abnormal development can be somewhat subjective, the abnormality endpoint is more ambiguous than mortality. However, by standardizing and clearly defining abnormalities and by using experienced personnel, much of the potential subjectivity of the abnormality endpoint can be avoided. The ecological relevance of larval abnormality is less certain than for adult mortality. Although the presence of larval abnormality suggests that recruitment to benthic assemblages may be impeded, the assemblages could be sustained by immigration of adult organisms. Because larvae represent a sensitive life stage of benthic invertebrates and because abnormalities may be expected to occur prior to the onset of mortality, the larval abnormality tests can generally be considered environmentally protective.

Genotoxicity tests have generally been applied to bacteria (e.g., Ames test) or single cells of higher organisms (e.g., rainbow trout). The various genotoxic endpoints include genetic mutations and chromosomal abnormalities. These tests are among the only ones that specifically evaluate the presence of mutagens and carcinogens in the environment. Genotoxicity tests require highly trained personnel to ensure that the endpoints are evaluated quantitatively and objectively. The ecological relevance of genotoxic effects are uncertain. They may or may not result in the death of affected organisms. In addition, because many chemical contaminants do not readily induce genotoxic effects, only a subset of chemicals are evaluated by these tests. Because few of these tests have been field validated, their sensitivity relative to identifying effects on higher organisms is not clear.

A reduction in bacterial bioluminescence is the primary endpoint of the Microtox test. This endpoint is the result of chemical toxicity influencing the electron transport systems of affected bacteria. It generally is unknown whether affected cells are permanently damaged or killed. The Microtox test can be readily conducted in a quantitative manner after a limited amount of training. Effects on bacteria are important in marine and estuarine ecosystems because bacteria play a major role in recycling detrital material at the base of the food chain (Steele 1974). However, because bacteria reproduce rapidly and thereby quickly repopulate disturbed areas, it is uncertain whether short-term alterations can substantially influence their functions in the ecosystem.

Chronic Sediment Bioassays—Chronic bioassays are conducted during relatively long exposure periods and measure the kinds of biological effects one might expect to find following prolonged exposure to toxic chemicals. Chronic tests generally are conducted over a full or partial life cycle of the test species. Because the exposure period is relatively long, it generally is expected that chronic responses will occur at lower levels of chemical toxicity than acute responses. Although this is frequently true, the use of a test organism for the acute tests that is more pollution-sensitive than the organism used for the chronic tests could result in acute responses being found at lower levels of toxicity than chronic responses.

By contrast with acute bioassays, most chronic bioassays require that seawater be continuously renewed in the test chambers (i.e., by a flow-through system). This requirement arises largely from the prolonged duration of chronic tests and the resulting potential for substantial changes in natural water quality variables (e.g., dissolved oxygen, pH) that could confound the experimental results. Because flow-through tests are more resource intensive and require more complex equipment than static or static-renewal tests, the cost of chronic testing generally is considerably higher than that of acute testing. In addition, flow-through testing usually requires direct access to seawater and thereby limits the number of laboratories that can conduct such tests.

Although a wide variety of biological endpoints can be used for chronic bioassays, a limited number has been commonly used throughout the United States. By contrast with the common acute bioassays, results from only a few of the chronic bioassays have compared effects on indigenous biota, and few are considered technically ready for routine regulatory application. However, as these tests are developed more fully and applied more frequently in support of regulatory activities, these technical shortcomings may be resolved. The most common chronic endpoints include:

- Adult mortality
- Reduced growth
- Reproductive effects
- Recruitment
- Histopathological abnormalities
- Bioaccumulation
- Intrinsic rate of population growth (IRPG).

Chronic mortality has most of the advantages and disadvantages discussed previously for acute mortality. The major advantage of this endpoint over acute mortality is that it addresses this relatively severe effect over a longer time scale and therefore may be more representative of many environmental exposure regimes. Because of this difference in length of exposure, it generally is assumed that chronic mortality will occur at lower chemical concentrations than acute mortality for a particular test species.

Reduced growth is generally measured in juvenile organisms (i.e., when growth is generally expected to be rapid) and can be estimated directly by measuring organisms prior to and following testing. Growth can also be represented as an instantaneous measurement called "scope for growth," which is based on physiological variables such as feeding rate, absorption efficiency, respiration

rate, and excretion. Growth itself does not require extensive expertise to measure, but scope for growth does. Both measures of growth can be determined quantitatively and objectively. However, the ecological relevance of reductions in organism growth is uncertain. If organisms can maintain their normal level of fecundity, then no reductions in population levels may be experienced. A reduction in size of adult organisms may cause organisms to be less desirable to predators and thereby influence trophic relationships. Alternatively, growth reductions may enhance the risk of being preyed upon because organisms cannot reach a size large enough to escape predation.

Reproductive effects are frequently measured in adult female organisms as number of eggs per individual, percentage of ovigerous individuals, and time to sexual maturity. These measurements can be made in a relatively quantitative and objective manner with a reasonable amount of training. The ecological relevance of reproductive effects may be more significant than growth because they imply that the local supply of recruits to adult populations may be reduced. However, adult populations could be sustained by recruitment of pelagic larvae from surrounding areas or immigration by adult organisms.

Recruitment is a measure of the introduction of new organisms into a benthic macroinvertebrate assemblage. Tagatz (1986) has developed a bioassay for evaluating the effects on recruitment of clean sediments spiked with test chemicals. Trays of sediment can be held in the laboratory (with larvae introduced via unfiltered seawater) or placed in the field. At the end of the exposure period, sediments are sieved and the retained organisms are identified and counted. This technique can be conducted in a relatively quantitative manner, and the endpoint (i.e., numbers of organisms) can be measured objectively. The ecological relevance of recruitment is similar to that described earlier for reproductive effects. Although the recruitment bioassay has been applied successfully to spiked clean sediment, its use for field-collected test sediment requires further development.

Histopathological abnormalities include measures of degeneration, necrosis, and other abnormalities in cells and tissues. The determination of these disorders requires a highly trained pathologist and frequently is relatively subjective. The ecological relevance of histopathological disorders are uncertain as they may or may not influence an organism's life functions. In the case of a malignant tumor it is relatively certain that the affected individual will experience negative consequences. With most other kinds of abnormalities, the affected individual may experience no negative consequences and may eventually be relieved from the disorder.

Bioaccumulation is the accumulation of chemical contaminants in animal tissue to concentrations greater than the ambient concentration (U.S. EPA 1985b). It can be measured quantitatively by a chemical laboratory and provides an estimate of the chemicals being consumed and retained by the test organisms. Bioaccumulation results can also be used to evaluate the trophic transfer of contaminants as a result of predation on contaminated organisms and to estimate risk to public health from consumption of contaminated organisms by humans (see Section 5). The ecological relevance of bioaccumulation is uncertain, because the presence of chemical contaminants in tissue does not necessarily imply that the affected organism is negatively influenced. However, U.S. EPA (1985b) recommends that any chemical having a high potential for bioaccumulation should be of concern until it can be demonstrated that it does not result in adverse effects. Because only a limited number of chemicals tend to bioaccumulate to measurable levels, this endpoint does not provide a complete assessment of the effects of all contaminants to which the test organisms are exposed.

IRPG is a method of predicting the consequences to a population of a species from contaminant-related stresses on individual organisms (Gentile et al. 1982). It mathematically integrates measures of mortality and fecundity to estimate the potential a particular population has for increasing in size. That is, the removal of organisms through mortality is balanced against the introduction of new organisms (estimated from fecundity) to determine the net effect on population growth. The estimation of IRPG is data intensive, as it generally requires developing information on age-specific survival and fecundity of the test species. In addition, it must be assumed that the mathematical equation used to estimate IRPG for some species accurately integrates the

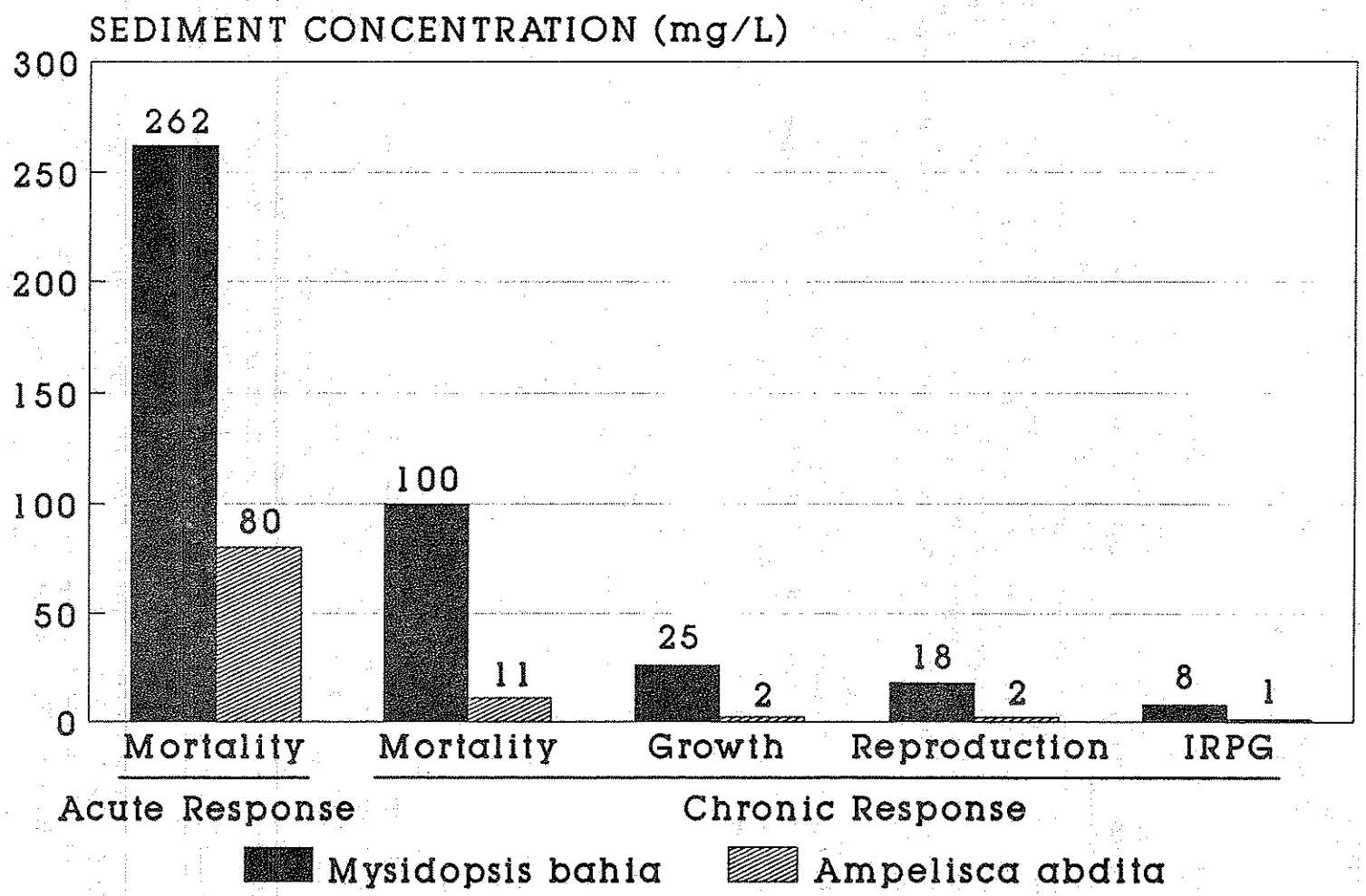
population effects of the mortality and fecundity of individual organisms. Nevertheless, this method is one of the only ones that attempts to determine directly the relationship between the responses of individuals under laboratory conditions with effects on populations in the field. The ecological relevance of IRPG is relatively high because it can indicate expected population trends in the field. However, a potential confounding factor to these expectations is recruitment or immigration of individuals from surrounding areas. IRPG can be a very sensitive measure of chronic toxicity (Gentile et al. 1982).

Acute-Chronic Comparisons—Very little information exists with respect to the relationship between species-specific acute and chronic toxicity in sediments. As mentioned previously, it usually is assumed that chronic effects occur at lower chemical concentrations than acute effects. The most detailed evaluations to date suggest that this assumption is valid. Gentile et al. (1988) subjected the epibenthic mysid *Mysidopsis bahia* and benthic amphipod *Ampelisca abdita* to various concentrations of contaminated sediment from Black Rock Harbor, Connecticut. The contaminated sediment was delivered in a suspended phase as the organisms were residing in or on bedded reference sediment. Acute effects were represented by mortality after an exposure period of 96 hours. Chronic effects were represented by mortality, growth reductions, reproductive impairment, and reductions in IRPG after exposure periods of 28 to 56 days. The authors found that for both species, all chronic effects occurred at sediment concentrations lower than those inducing acute effects (Figure 3). In general, the sensitivity of the various endpoints were acute mortality < chronic mortality < growth ≤ reproduction < IRPG. The authors also found that *A. abdita* was considerably more sensitive than *M. bahia*, such that acute mortality in the former species occurred at lower sediment concentrations (80 mg/L) than those inducing chronic mortality in the latter species (100 mg/L). These results show that although chronic effects generally occur at lower chemical concentrations than acute effects within a species, interspecific differences in sensitivity can result in acute effects of one species being a more sensitive indicator of chemical toxicity than chronic effects in a second species.

In addition to the scarcity of information comparing acute and chronic toxicity in sediments, there is a relative scarcity of information on chronic effects alone in natural sediments. This latter information gap is largely a result of the historical emphasis on acute tests, the relatively high cost of many chronic tests, and the relatively recent development of chronic tests that can be conducted in a consistent and reliable manner. The relative scarcity of information on chronic effects in natural sediments is a major impediment to determining whether sediments evaluated in the past have chronic effects (e.g., for inventory purposes).

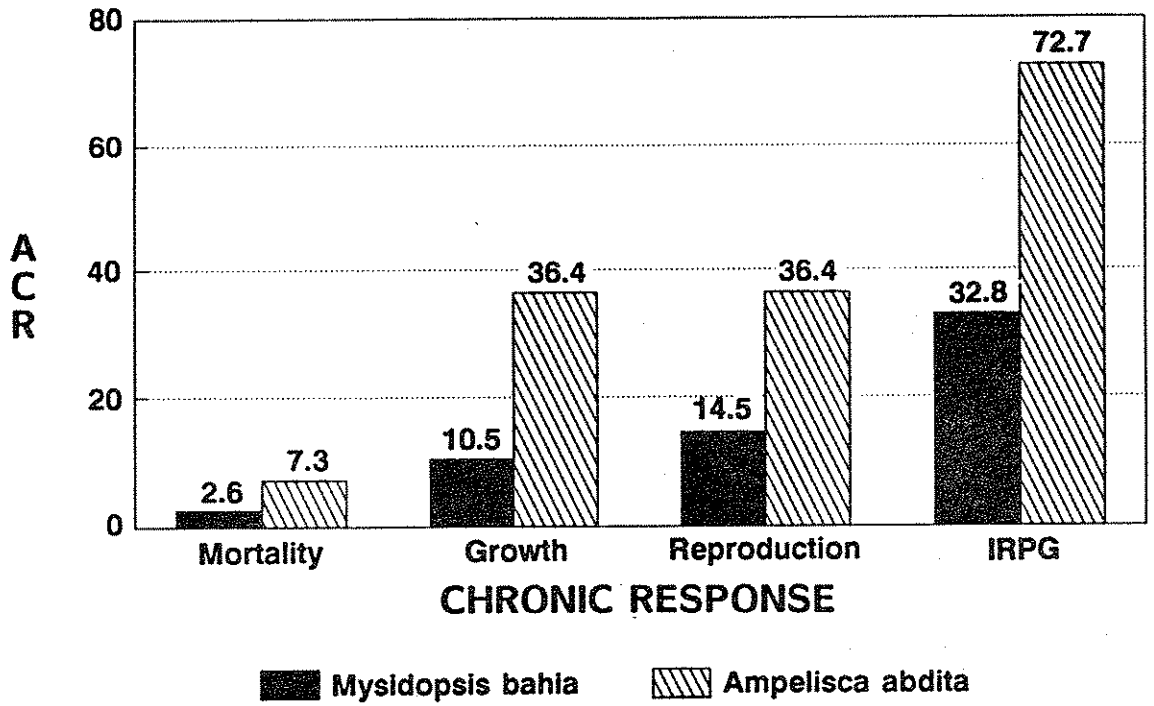
One method of estimating the relationship between acute and chronic effects with limited information is to use an ACR (U.S. EPA 1985b). The ACR expresses this relationship as a ratio of the chemical concentration causing acute effects in a species to the concentration causing chronic effects in that same species. Because ACRs usually vary among chemicals and species, there is some degree of uncertainty in applying ACRs to situations other than the ones used to derive them. For water, U.S. EPA (1986g) has developed marine chronic criteria for approximately 25 percent of chemicals having acute criteria and freshwater chronic criteria for approximately 40 percent of the chemicals having acute criteria. Saltwater ACRs range from 1 to 866 with a median value of 5. Freshwater ACRs range from 1 to 1,316 with a median value of 6. With respect to complex chemical mixtures in effluents, EPA recommends that an ACR of 10 be used in the absence of specific information (U.S. EPA 1985b).

By contrast with water and effluents, ACRs have rarely been applied to sediments. Calculation of ACRs for the data on *M. bahia* and *A. abdita* discussed earlier (Figure 3), shows that they range from 2.6 to 32.8 for *M. bahia* and from 7.3 to 72.7 for *A. abdita*, depending on the kind of chronic effect (Figure 4). For both species, regulations based on an ACR of 10 (i.e., that recommended for effluents) would account for chronic mortality but not account for the more sensitive kinds of chronic effects (i.e., growth reductions, reproductive impairment, reduced IRPG).



Reference: Gentile et al. (1988)

FIGURE 3. Concentrations of sediment from Black Rock Harbor found to be toxic to *Mysidopsis bahia* and *Ampelisca abdita*.



Reference: Gentile et al. (1988)

FIGURE 4. Acute-chronic ratios (ACRs) for two species exposed to sediment from Black Rock Harbor.

3. Ease of use—the level of technical skill and resources required to conduct a bioassay.

4. Availability of test organisms—the ease with which test organisms can be acquired and prepared for testing.

5. Endpoint reliability—the level of confidence in the accuracy of a bioassay endpoint.

6. Relationship to indigenous biota—the correspondence between the results of a bioassay and field evaluations of indigenous biota.

7. Holding constraints—any limitations in holding test organisms or sediments that could influence routine technical application of a bioassay.

8. Stage of protocol development—the degree to which a bioassay protocol has been documented and standardized.

Amphipod (*R. abronius*) Test—This bioassay involves a 10-day exposure of adult organisms to a 2-cm layer of bedded (i.e., settled) test sediment (Swartz et al. 1985a; ASTM in review) For each field sample, 20 organisms are tested in each of 5 test chambers. The primary endpoint is mortality. A secondary endpoint is nonreburial by the surviving amphipods. Bioassay responses at test sites are compared with responses at appropriate reference areas using statistical techniques.

Cost-Effectiveness: Both the mortality and nonreburial endpoints were found to exhibit a moderate level of cost-effectiveness by Pastorok and Becker (1989). The cost of this bioassay is moderate (i.e., average = $550 per field sample). Sensitivity and statistical power of the mortality endpoint are both considered to be moderate. By contrast, sensitivity and statistical power of the nonreburial endpoint are considered to be low.

Ecological Relevance: *R. abronius* is a resident of Puget Sound. As an amphipod, it is expected to be pollution-sensitive (Bellan-Santini 1980) and an important component of the diet of numerous juvenile and adult fishes (Simenstad et al. 1979; Wingert et al. 1979). The adult life stage evaluated probably is not the most sensitive stage in the organism's life cycle. In nature, the potential for exposure of this species to sediment contaminants is high because the organisms burrow into the sediment and feed upon material found naturally in the sediment. The primary endpoint (i.e., mortality) has relatively clear ecological meaning. That is, if adult organisms cannot survive in an environment, it is likely that severe alterations of benthic assemblages will be found. The secondary endpoint (i.e., nonreburial) has a less certain ecological relevance than mortality. However, it generally is assumed that if the amphipods do not burrow into the sediment they will experience increased risk of predation. Nonreburial may therefore indicate eventual mortality.

Ease of Use: The amphipod mortality test is relatively easy to conduct. A fixed number of organisms is introduced to each test chamber at the start of the experiment, recovered by sieving the test sediment after 10 days, and counted. Maintenance of test chambers during the 10-day exposure period is minimal, although observations are made daily and organisms sometimes found trapped by surface tension at the water surface are gently freed. Although the conduct of the amphipod mortality test is relatively straightforward, the design of the sampling scheme is complicated by the sensitivity of the test species to sediment grain-size properties. Although the preferred habitat of *R. abronius* is well-sorted, fine sand (Oakden 1984), many contaminated sites are characterized by fine-grained (i.e., muddy) sediment. DeWitt et al. (1988) found that sediments having a high percentage of fine-grained material could increase the mortality rate of these organisms in the absence of chemical contaminants. Thus, it is necessary to account for the influence of this natural variable when conducting bioassays. To avoid the potential confounding effects of sediment grain size on the measurement of chemical toxicity when making comparisons with reference samples, sediments from the reference area should have the same grain-size properties as sediments from each test site being compared. This approach was followed in the present study. An alternative approach is to use a regression equation developed by Dewitt et al.

(1988) using data from various Puget Sound reference areas. The equation predicts the relationship between sediment grain size and amphipod mortality. Test results that lie outside the 95 percent prediction limits are considered indicative of chemical toxicity.

Availability of Test Organisms: The test organisms must be collected from the field, as rearing of *R. abronius* in the laboratory has yet to be accomplished with adequate results. Organisms are collected subtidally using a dredge operated from a small boat. Organisms can be collected at a number of sites throughout Puget Sound, but they are usually collected from West Beach on Whidbey Island. Because organisms are collected from the field, where environmental conditions vary, they must be acclimated to the experimental conditions for 4-10 days.

Endpoint Reliability: The primary endpoint (i.e., mortality) is unambiguous and relatively easy to determine. Animals are considered dead if they show no signs of movement after gentle prodding with a probe. Animals not recovered at the end of the exposure period are presumed to have died and decomposed. The secondary endpoint (i.e., nonreburial is also unambiguous and relatively easy to determine. An organism either burrows into the sediment or it remains on the sediment surface. The reliability of both endpoints is enhanced by using positive controls (i.e., clean sediment spiked with a reference toxicant such as cadmium chloride or sodium pentachlorophenate) to demonstrate that the test organisms were adequately responsive to toxic effects, and negative controls (i.e., clean native sediment) to demonstrate that organisms were sufficiently healthy (i.e., control mortality <10 percent). The interpretability of the endpoint with respect to toxic effects is enhanced if the potential confounding influence of sediment grain size is accounted for (see Ease of Use section above).

Relationship to Indigenous Biota: Results of the amphipod mortality test agreed with the effect/no effect designation of benthic assemblages in Commencement Bay and Carr Inlet at 60 percent (28/47) of the stations sampled in 1984 during the Commencement Bay remedial investigation (Becker et al. 1988). Of the 19 stations with altered benthic assemblages, only 42 percent (8/19) were identified by the bioassay as impacted, indicating that this test is not a particularly sensitive indicator of benthic effects. Of the 16 stations identified by the bioassay as impacted, only 50 percent (8/16) were characterized by altered benthic assemblages, indicating that this test did not identify altered assemblages with a high degree of efficiency. However, 7 of the 8 stations with significant bioassay responses, but apparently normal benthic assemblages were characterized by sediments having a high percentage of fine-grained material (i.e., ≥75 percent). If these seven stations are disregarded, then 8 of the 9 stations (89 percent) identified by the bioassay as impacted also had altered benthic assemblages.

Swartz et al. (1985b) compared the results of the amphipod mortality bioassay with impacts on benthic macroinvertebrate assemblages near a major municipal sewage outfall off Los Angeles. They concluded that the bioassay was not particularly sensitive to environmental degradation but was usually associated with major perturbations of benthic assemblages. Chapman et al. (1987) found similar results in a study conducted in three areas of San Francisco Bay. In that study, significant bioassay responses were found only in the most pollution-degraded areas. Severity of pollution was judged in part by the degree of alteration of benthic macroinvertebrate assemblages.

In summary, most studies agreed that the amphipod mortality bioassay was effective in identifying substantially altered benthic macroinvertebrate assemblages. However, the test was not particularly effective in identifying moderately altered assemblages.

Holding Constraints: The test organisms can be held (under controlled conditions) for a maximum of 10 days prior to testing. Current PSEP protocols specify that the test sediment can be held (at 4° C in the dark) for a maximum of 14 days after collection. Sediments must not be frozen prior to testing.

Stage of Protocol Development: Protocols for the amphipod mortality test using *R. abronius* are well standardized. They were first developed for ASTM by Swartz et al. (1985a) and were later modified for use in Puget Sound (Tetra Tech and E.V.S. 1986). More recently, the methods were included as part of a generic amphipod mortality protocol developed by ASTM (in review). The protocols for this test have also been applied in numerous field studies, including several in which comparisons were made with responses of other bioassays or effects on indigenous organisms (e.g., Swartz et al. 1982, 1985b, 1986; Barrick et al. 1985, 1986; Chapman et al. 1987; Becker et al. 1988; Beller et al. 1988; Pastorok et al. 1988).

Amphipod (*E. estuarius*) Test—This bioassay involves a 10-day exposure of adult organisms to a 2-cm layer of bedded (i.e., settled) test sediment (ASTM in review) For each field sample, 20 organisms are tested in each of 5 test chambers. The primary endpoint is mortality. A secondary endpoint is nonreburial by the surviving amphipods. Bioassay responses at test sites are compared with responses at appropriate reference areas using statistical techniques.

Cost-Effectiveness: Both the mortality and nonreburial endpoints were found to exhibit a moderate level of cost-effectiveness by Pastorok and Becker (1989). The cost of this bioassay is moderate (i.e., average = $550 per field sample). Sensitivity and statistical power of the mortality endpoint are considered to be moderate. By contrast, sensitivity and statistical power of the nonreburial endpoint are considered to be low.

Ecological Relevance: *E. estuarius* is a resident of Puget Sound. It is a free-burrowing amphipod that generally is found in fine-grained intertidal sands (DeWitt et al. in review). However, subtidal populations also exist to some extent. This species is found in greatest numbers in estuarine areas where salinity ranges from 15 to 25 ppt. The ecological relevance of this test is similar to that discussed for *R. abronius* [see Amphipod (*R. abronius*) Test section above].

Ease of use: Ease of use of the amphipod test is evaluated above [see Amphipod (*R. abronius*) Test section above]. As with *R. abronius*, *E. estuarius* is somewhat sensitive to sediment grain size properties. DeWitt et al. (in review) found that sediments having a high percentage of fine-grained material could increase the mortality rate of *E. estuarius* in the absence of chemical contamination. Approaches to account for the influence of this natural variable when conducting bioassays are discussed above.

Availability of Test Organisms: The test organisms must be collected from the field, since rearing of *E. estuarius* in the laboratory has yet to be accomplished with adequate results. The abundance and distribution of this species throughout Puget Sound is presently unknown. Because organisms are collected from the field, where environmental conditions vary, they must be acclimated to the experimental conditions for 4-10 days.

Endpoint Reliability: The endpoints (i.e., mortality and nonreburial) are unambiguous and relatively easy to determine [see Amphipod (*R. abronius*) Test section above].

Relationship to Indigenous Biota: No direct comparisons have been made between the results of this bioassay and effects on indigenous biota in Puget Sound.

Holding Constraints: The test organisms can be held (under controlled conditions) for a maximum of 10 days prior to testing. To be consistent with current PSEP protocols for other bioassays, the test developer agreed that the test sediment can be held (at 4° C in the dark) for a maximum of 14 days after collection. Sediments must not be frozen prior to testing.

Stage of Protocol Development: Protocols for the amphipod mortality test using *E. estuarius* are relatively well standardized. Based largely on the well standardized protocols developed previously for *R. abronius*, they are included as part of a generic amphipod mortality protocol

developed by ASTM (in review). The protocols for this test are currently undergoing peer review, and have been applied only in a single field study (DeWitt et al. in review), excluding the present study.

Juvenile Bivalve (*P. generosa*) Test—This bioassay involves a 10-day exposure of juvenile geoducks to a 2-cm layer of bedded (i.e., settled) test sediment (Johns 1988). For each field sample, 10 organisms are tested in each of 5 test chambers. The primary endpoint is mortality. Bioassay responses at test sites are compared with responses at appropriate reference areas using statistical techniques.

Cost-Effectiveness: The mortality endpoint was found to exhibit a low level of cost-effectiveness by Pastorok and Becker (1989). The cost of this bioassay is relatively high (i.e., average = $840 per field sample). Sensitivity of the endpoint is considered to be low, but statistical power of the endpoint is considered to be moderate.

Ecological Relevance: The test species is a resident of Puget Sound. The juvenile life stage evaluated is probably intermediate in sensitivity to the earlier embryonic and larval stages and the later adult stage. The potential for exposure of the test organisms to sediment contaminants is relatively high because geoducks reside in unlined burrows in the sediment, and they are relatively stationary once the burrows are occupied. However, because geoducks are suspension feeders, the potential for exposure to sediment-bound contaminants is probably lower than it would be for a deposit-feeding organism. In addition, exposure to contaminants may be reduced during testing because organisms are provided with uncontaminated diatom paste as a food source. The primary endpoint (i.e. mortality) has relatively clear ecological meaning. That is, if juvenile organisms cannot survive in an environment, it is likely that severe alterations of benthic assemblages will be found. However, the inability to colonize an environment in the juvenile stage could be compensated for to some degree through adult immigration by motile species.

Ease of Use: The juvenile bivalve test is relatively easy to conduct. A fixed number of organisms is introduced to each test chamber at the start of the experiment, recovered by sieving the test sediment after 10 days, and counted. Maintenance of the test chambers is minimal, although feeding is conducted every other day.

Availability of Test Organisms: At present, the test organisms can be obtained only from a single state-operated shellfish hatchery. Organisms are available throughout the year. Although field collection of test organisms may be possible, it is considered impractical for bioassay purposes. Organisms must be acclimated to test conditions and fed prior to testing.

Endpoint Reliability: The endpoint (i.e., mortality) is unambiguous and relatively easy to determine. Animals are considered dead if they show no signs of movement after gentle prodding of the siphon, foot, or mantle tissue with a probe. The reliability of the endpoint is enhanced by using negative controls (i.e., clean sediment) to demonstrate that organisms were sufficiently healthy (i.e., control mortality <10 percent).

Relationship to Indigenous Biota: No comparisons have been made between the results of this test and effects on indigenous biota in Puget Sound.

Holding Constraints: Geoducks can be held for prolonged periods prior to testing, as long as water temperatures are relatively low. To be consistent with current PSEP protocols for other bioassays, the test developer agreed that the test sediment can be held (at 4° C in the dark) for 14 days after collection. Sediments must not be frozen prior to testing.

Stage of Protocol Development: Preliminary protocols for the juvenile bivalve mortality test were developed as the test results reported herein were being generated (Johns 1988). These methods were based partly on those of the well-standardized protocols used for the amphipod mortality test (ASTM in review), but have yet to be peer-reviewed and tested in other field studies.

Bivalve (*C. gigas*) Larvae Test—This bioassay involves a 48-hour exposure of embryos (2 hours after fertilization) to 15 grams of bedded test sediment (Chapman and Morgan 1983; ASTM 1985). For each field sample, 20,000-40,000 developing embryos are tested in each of 5 test chambers. The primary endpoint is larval abnormality or failure to develop to the fully shelled, hinged "D-shaped" prodissoconch I stage. Bioassay responses at test sites are compared with responses at appropriate reference areas using statistical techniques.

Cost-Effectiveness: The cost-effectiveness of this bioassay could not be determined by Pastorok and Becker (1989), because of unacceptably high larval mortality in the negative controls (i.e. clean seawater). Repeated analyses of this test using an alternate species (the edible mussel, *Mytilus edulis*) also failed to meet the performance criteria for the negative controls (i.e., mortality <30%). The cost of this bioassay is moderate (i.e., average = $630 per field sample). Sensitivity and statistical power of the abnormality endpoint have not been evaluated because of the problem mentioned above.

Ecological Relevance: The test species is a resident of Puget Sound, although it was originally introduced from Japan (Kozloff 1983). The life stages evaluated (embryo and larva) represent two of the most sensitive stages in the life cycle of the organism. The potential for exposure of the test organisms to sediment contaminants is moderate because although bedded sediments are present in each test chamber, bivalve embryos and larvae reside primarily in the water column and therefore rarely are in direct contact with bedded sediments. The primary endpoint (i.e., abnormality) has a relatively clear ecological meaning for the test species and other species that rely primarily on larval recruitment to colonize areas (i.e., species with relatively sedentary juvenile and adult stages). That is, abnormal larvae are unlikely to survive and the establishment of adult assemblages would thereby be prevented. The ecological relevance of the test for motile organisms that can colonize a contaminated area in the juvenile and adult stages is less certain, because successful embryonic and larval development could occur in areas removed from contamination.

Ease of Use: The bivalve larvae abnormality test requires a fair amount of skill and experience to conduct. Embryos for testing are obtained by inducing field-collected organisms to spawn in the laboratory. Improper handling of the adult organisms can result in premature spawning or failure to spawn adequately. Microscopic evaluations are required to estimate densities of embryos and larvae at the beginning and end of each test and to evaluate larval abnormalities at the end of testing. Maintenance of test chambers during the 48-hour exposure period is minimal. At present, it is unknown whether natural factors such as sediment grain size can influence results of the test and thereby complicate the design of studies using this bioassay. However, it is known that larvae of *C. gigas* normally are not associated with the kinds of fine-grained sediment commonly found at contaminated sites.

Availability of Test Organisms: Because the test species is a resident of Puget Sound, adults can be collected from numerous locations, including commercial rearing facilities. A major limitation with the test species is that spawning occurs naturally in the Sound only during the summer. If testing is scheduled for other times of the year, then adults must be conditioned to spawn. That is, they are gradually exposed to increasing water temperatures up to 20° C to stimulate maturation of gametes. Depending on the physiological and gametogenic status of the organisms at the time of collection, conditioning can extend from several days to several weeks. Induction of spawning sometimes results in a decreased viability of gametes. An alternative for conducting this test when *Crassostrea gigas* is not spawning naturally is to use *Mytilus edulis* as an alternate test species.

Endpoint Reliability: The endpoint (i.e., abnormality) is more ambiguous than mortality. However, by standardizing and clearly defining abnormalities and by using experienced personnel, much of the potential subjectivity of the endpoint can be avoided. The reliability of the endpoint is enhanced by using positive controls (i.e., clean, filtered, UV-treated seawater spiked with a reference toxicant such as cadmium chloride or sodium pentachlorophenate) to demonstrate that the test organisms were adequately responsive to toxic effects, and negative controls (i.e., clean seawater and clean sediment) to demonstrate that organisms were sufficiently healthy (i.e., control mortality <30 percent).

Relationship to Indigenous Biota: Results of the bivalve larvae abnormality test agreed with the effect/no effect designation of benthic assemblages in Commencement Bay and Carr Inlet at 81 percent (38/47) of the stations sampled. Of the 19 stations with altered benthic assemblages, 68 percent (13) were identified as impacted by the bioassay, indicating that this test is a moderately sensitive indicator of benthic effects. Of the 16 stations identified as impacted by the bioassay, 81 percent (13/16) were actually characterized by altered benthic assemblages, indicating that this test identified altered benthic assemblages with a relatively high degree of efficiency. In summary, the bivalve larvae test was relatively effective in identifying altered benthic assemblages.

Holding Constraints: Adult oysters can be held for a maximum of 2-3 weeks after attaining acceptable maturity. Gamete quality tends to deteriorate rapidly during longer holding periods. Current PSEP protocols specify that the test sediment can be held (at 4° C in the dark) for a maximum of 14 days after collection.

Echinoderm (*D. excentricus*) Embryo Test—This bioassay involves a 48-hour exposure of embryos (2 hours after fertilization) to 15 grams of bedded test sediment (Dinnel and Stober 1985). For each field sample, approximately 12,000 developing embryos are tested in each of 5 test chambers. The primary endpoint is larval developmental abnormality or failure to develop into a normal pluteus larvae. A secondary endpoint is larval chromosomal abnormality (i.e., a measure of genotoxicity). Bioassay responses at test sites are compared with responses at appropriate reference areas using statistical techniques.

Cost-Effectiveness: The developmental abnormality endpoint was found to exhibit a moderate level of cost-effectiveness by Pastorok and Becker (1989). By contrast, the chromosomal abnormality endpoint showed a low level of cost-effectiveness. The cost of this bioassay is relatively high (i.e., average = $750 per field sample for the developmental endpoint; an additional $270 per field sample for the chromosomal endpoint). Sensitivity and statistical power of the developmental endpoint are both considered to be high. By contrast, sensitivity and statistical power of the chromosomal endpoint are considered to be low and moderate, respectively.

Ecological Relevance: *D. excentricus* is a resident of Puget Sound. The life stages evaluated (embryo and larva) represent two of the most sensitive stages in the life cycle of the organism. The potential for exposure of the test organisms to sediment contaminants in nature is moderate, because echinoderm embryos and larvae reside primarily in the water column and therefore rarely are in direct contact with bedded sediments. The primary endpoint (i.e., developmental abnormality) has a relatively clear ecological meaning for the test species and other species that rely primarily on larval recruitment to colonize areas (i.e., species with relatively sedentary juvenile and adult stages). That is, abnormal embryos and larvae are unlikely to survive, and the establishment of adult assemblages would thereby be precluded. The ecological relevance of the test for motile organisms that can colonize a contaminated area in the juvenile and adult stages is less certain, because successful embryonic and larval development could occur in areas removed from contamination. The secondary endpoint (i.e., chromosomal abnormality) has a less certain ecological relevance than developmental abnormality. It is unknown whether any of these abnormalities will substantially influence the survival of affected organisms.

Ease of Use: The echinoderm embryo abnormality test requires a fair amount of skill and experience to conduct. Embryos for testing are obtained by inducing field-collected organisms to spawn in the laboratory. Improper handling of the adult organisms can result in premature spawning or failure to spawn adequately. Microscopic evaluations are required to estimate densities of embryos and larvae at the beginning and end of each test and to evaluate larval abnormalities at the end of testing. Maintenance of test chambers during the 48-hour exposure period is minimal.

Availability of Test Organisms: Because the test species is a resident of Puget Sound, adults can be collected from numerous locations. A major limitation with the test species is that spawning occurs naturally in the sound only during the summer, and ripe organisms normally are available only from May to October. If testing is scheduled for other times of the year, the use of an alternate echinoderm species (e.g. the green sea urchin, *Strongylocentrotus droebachiensis*, or the purple sea urchin, *S. purpuratus*) is generally recommended (Dinnel and Stober 1985). The green sea urchin normally is ripe from January to April, and the purple sea urchin normally is ripe from December to March.

Endpoint Reliability: The abnormality endpoints (i.e. developmental and chromosomal) are more ambiguous than mortality. However, by standardizing and clearly defining abnormalities and by using experienced personnel, much of the potential subjectivity inherent in use of these endpoints can be avoided. The reliability of either endpoint is enhanced by using positive controls (i.e., clean, filtered, UV-treated seawater spiked with a reference toxicant such as cadmium chloride or sodium pentachlorophenate) to demonstrate that the test organisms were adequately responsive to toxic effects and using negative controls (i.e., clean seawater and clean sediment) to demonstrate that organisms were sufficiently healthy (i.e., control mortality <30 percent and control developmental abnormality <10 percent).

Relationship to Indigenous Biota: No direct comparisons have been made between the results of this test and effects on indigenous biota in Puget Sound.

Holding Constraints: Adult sand dollars can be held for up to 30 days prior to testing. To be consistent with current PSEP protocols for other bioassays, the test developer agreed that the test sediment can be held (at 4° C in the dark) for a maximum of 14 days after collection.

Stage of Protocol Development: Protocols for the echinoderm embryo abnormality test are moderately well standardized for both the developmental (Dinnel and Stober 1985) and chromosomal (Hose and Puffer 1983; Hose 1985) endpoints. However, methods for both endpoints for *Dendraster excentricus* have yet to be peer-reviewed or tested in other field studies.

Microtox (*P. phosphoreum*) Saline Extract Test—This bioassay involves a 15-minute exposure of bacteria to aliquots of saline extract from 13-26 grams of test sediment (Bulich et al. 1981; Beckman Instruments 1982; Williams et al. 1986). For each field sample, a series of four dilutions and a diluent blank are evaluated. Two replicate measurements are made for each dilution and blank. Bioluminescence is measured using an automated toxicity analyzer system with a temperature-regulated photometer equipped with a photomultiplier. Bioassay responses at test sites are compared with responses at appropriate reference areas using statistical techniques.

Cost-Effectiveness: The luminescence endpoint was found to exhibit a high level of cost-effectiveness by Pastorok and Becker (1989). The cost of this bioassay is low (i.e., average = $190 per field sample, exclusive of extraction costs). Sensitivity and statistical power of the developmental endpoint are both considered to be high.

Ecological Relevance: The test species is a member of the estuarine and marine pelagic communities (Holt 1977). As a bacterium, it is representative of the group of organisms that forms the base of detrital-based food webs (Steele 1974). Bacteria play a major role in decomposing organic matter (i.e., detritus) and making it available to higher organisms (e.g., benthic macroinvertebrates). The potential for exposure of the test organisms to sediment contaminants is limited in the field because the organisms are pelagic. Exposure during laboratory testing is further limited by the fact that the bioassay is conducted on a saline extract of the test sediment (i.e., sediment is not present in the test chamber). The saline extraction tends to remove only water-soluble contaminants from the test sediment, and therefore it may not be representative of the full range of contaminants that would affect the organisms if they were exposed directly to test sediment. The primary endpoint (i.e., change in bioluminescence) is an indicator of changes in cellular metabolic state (Hastings and Nealson 1977). Although this endpoint is probably very sensitive, it is unknown whether it has serious consequences for the organisms. It is also uncertain whether the changes in metabolic state will have a substantial influence on the ecological role of the bacteria. If this ecological role is impeded, then it could deprive certain higher organisms of their primary food source and thereby alter the ability of these higher organisms to survive.

Ease of Use: The Microtox test is relatively easy to conduct after a minimal amount of training and practice. However, the conduct of the test requires access to the automated analyzing equipment. As the determination of luminescence is automated, the primary skill required is the ability to make dilutions quantitatively and to accurately transfer the required small amounts of test solution to the test cuvettes with precise timing. No maintenance is required during the 15-minute exposure period, and the light emissions are read directly off the analytical equipment. Because temperature can influence luminescence, all measurements must be made using a constant temperature (i.e., 15° C). One uncertainty that may influence the design of studies using this test is the effect on luminescence of different pore water salinities among sediment samples. If this influence is substantial, then a salinity correction factor may need to be developed when a study includes samples with widely varying pore water salinities.

Availability of Test Organisms: The test organisms are available commercially in freeze-dried form, so field collection and culturing are unnecessary. Samples are rehydrated in the laboratory prior to testing.

Endpoint Reliability: The endpoint (i.e., luminescence) can be measured objectively and relatively accurately using automated equipment. The reliability of the endpoint is enhanced by using positive controls (i.e., clean 2 percent saline diluent spiked with a reference toxicant such as sodium arsenate) to demonstrate that the test organisms were adequately responsive to toxic effects and using negative controls (i.e., clean 2 percent saline diluent and clean sediment extract) to demonstrate that organisms were sufficiently healthy.

Relationship to Indigenous Biota: Results of the Microtox test agreed with the effect/no effect designation of benthic assemblages in Commencement Bay and Carr Inlet at 68 percent (32/47) of the stations sampled in 1984 during the Commencement Bay Remedial Investigation (Becker et al. 1988). Of the 19 stations with altered benthic assemblages, 84 percent (16/19) were identified by the bioassay as impacted, indicating that this test is a very sensitive indicator of benthic effects. Of the 28 stations identified by the bioassay as impacted, 57 percent (16/28) were characterized by altered benthic assemblages, indicating that this test identified altered benthic assemblages with only a moderate degree of efficiency. In summary, the Microtox test is very effective at identifying altered benthic assemblages. However, the test also tends to identify a relatively large number of stations as impacted where benthic assemblages are not substantially altered.

Holding Constraints: The test organisms can be held indefinitely in freeze-dried form, but only for a maximum of 5 hours following rehydration. Current PSEP protocols specify that the test sediment can be held (at 4° C in the dark) for a maximum of 14 days after collection.

Stage of Protocol Development: Protocols for the general Microtox test are well standardized. They were developed originally for testing water samples (Bulich et al. 1981; Beckman Instruments 1982), but were subsequently modified for use with saline extracts of marine sediments (Williams et al. 1986; Tetra Tech and E.V.S. 1986). The marine protocols have been applied only in a single field study (Williams et al. 1986), excluding the present study.

Microtox (*P. phosphoreum*) Organic Extract Test—This bioassay involves a 15-minute exposure of bacteria to aliquots of organic extract from 10 grams of test sediment (Bulich et al. 1981; Beckman Instruments 1982; Williams et al. 1986). For each field sample, a series of four dilutions, a diluent blank, and an ethanol carrier blank are evaluated. Two replicate measurements are made for each dilution and blank. Bioluminescence is measured using an automated toxicity analyzer system with a temperature-regulated photometer equipped with a photomultiplier. Bioassay responses at test sites are compared with responses at appropriate reference areas using statistical techniques.

Cost-Effectiveness: The luminescence endpoint was found to exhibit a high level of cost-effectiveness by Pastorok and Becker (1989). The cost of this bioassay is low (i.e., average = $190 per field sample, exclusive of extraction costs). Sensitivity and statistical power of the developmental endpoint are both considered to be high.

Ecological Relevance: The ecological relevance of the Microtox test is discussed in general above [see Microtox (*P. phosphoreum*) Saline Extract Test section above]. The organic extraction tends to remove primarily neutral, non-ionic organic contaminants from the test sediment. Because the organism is not directly exposed to sediment in nature and because the organic extraction process is likely to remove contaminants that are not bioavailable, the Microtox organic test may overestimate potential biological effects.

Ease of Use: The ease of use of the Microtox test is discussed above [see Microtox (*P. phosphoreum*) Saline Extract Test section above]. Because the organic vehicle (i.e., ethanol) may influence bacterial luminescence, it is essential that its contribution to the observed responses be removed by blank correction.

Availability of Test Organisms: The test organisms are available commercially in freeze-dried form, so field collection and culturing are unnecessary. Samples are rehydrated in the laboratory prior to testing.

Endpoint Reliability: The endpoint (i.e., luminescence) can be measured objectively and relatively accurately [see Microtox (*P. phosphoreum*) Saline Extract Test section above].

Relationship to Indigenous Biota: No comparisons have been made between the results of this test and effects on indigenous biota in Puget Sound.

Holding Constraints: The test organisms can be held indefinitely in freeze-dried form, but only for a maximum of 5 hours following rehydration. Current PSEP protocols specify that the test sediment can be held (frozen at -20° C within 8 hours of collection) for a maximum of 6 months after collection.

Stage of Protocol Development: Protocols for the general Microtox test are well standardized. They were developed originally for testing water samples but were modified for use with organic extracts of marine sediment (Schiewe et al. 1985; Tetra Tech and E.V.S. 1986). The marine protocols have been applied in several field studies in Puget Sound.

Juvenile Polychaete (*N. arenaceodentata*) Test—This chronic bioassay involves a 20-day exposure of juvenile organisms to 150 grams of bedded test sediment (Johns 1988). For each field sample, five organisms are tested in each of five test chambers. The primary endpoints are mortality and reduction in biomass. Bioassay responses at test sites are compared with responses at appropriate reference areas using statistical techniques.

Cost-effectiveness: Both the mortality and biomass endpoints were found to exhibit a low level of cost-effectiveness by Pastorok and Becker (1989). The cost of this bioassay is high (i.e., average = $900 per field sample). Sensitivity and statistical power are both considered to be low for the mortality endpoint and moderate for the biomass endpoint.

Ecological Relevance: The life stage evaluated (i.e., juvenile) is probably intermediate in sensitivity compared to the earlier embryonic and larval stages and the later adult stage. The potential for exposure of the test organisms to sediment contaminants is relatively high because they nestle in surface sediments and feed upon deposited organic matter. However, exposure to contaminants during testing may be reduced because organisms are provided with uncontaminated prawn flakes as a food source.

The mortality endpoint has relatively clear ecological meaning. That is, if juvenile organisms cannot survive in an environment, it is likely that severe alterations of benthic assemblages will be found. However, the inability to colonize an environment in the juvenile stage could be compensated for to some degree through adult immigration by motile species. The biomass endpoint has a somewhat uncertain ecological meaning. Reductions in the biomass (or growth) of individuals can have substantial effects at the population level if spawning activity and fecundity are negatively affected. However, if organisms can maintain their normal spawning activity and level of fecundity, no reductions in population densities may be experienced. A reduction in the biomass of adult organisms may cause those individuals to be less desirable as prey and thereby enhance the survival potential. Alternatively, biomass reductions may enhance the risk of being preyed upon because organisms cannot reach a size large enough to escape predators. Although the ecological relevance of growth reduction is uncertain, this endpoint is probably the most meaningful sublethal response available.

Ease of Use: The juvenile polychaete test is relatively easy to conduct. A fixed number of organisms are introduced to each test chamber at the start of the experiment, recovered after 20 days, counted, and weighed. Although this test is relatively easy to conduct, a moderate amount of maintenance is required during testing. Maintenance includes renewing 33 percent of the seawater in each chamber every 3 days and feeding the test organisms every 2 days.

Availability of Test Organisms: The test organisms are obtained primarily from a single set of laboratory cultures and are available throughout the year. These organisms are relatively easy to culture. Although field collection of test organisms may be possible, it is considered impractical for bioassay purposes. Organisms must be acclimated to the test conditions and fed prior to testing.

Endpoint Reliability: The mortality endpoint is unambiguous and relatively easy to determine. Animals are considered dead if they show no signs of movement after gentle prodding with a probe. Animals not recovered at the end of the exposure period are presumed to have died and decomposed. The biomass endpoint can be determined objectively and with high precision by weighing individuals to the nearest 0.01 mg. Precision is particularly improved by using dry weight rather than wet weight measures of biomass. At present, there are no quality assurance/quality control (QA/QC) performance criteria for the negative controls.

Relationship to Indigenous Biota: No comparisons have been made between results of this test and effects on indigenous biota in Puget Sound.

Holding Constraints: Depending on the size of the organisms, holding times should not exceed 10-14 days prior to testing, or the organisms will outgrow the juvenile stage. To be consistent with current PSEP protocols for other bioassays, the test developer agreed that the test sediment can be held (at 4° C in the dark) for a maximum of 14 days following collection. Sediments must not be frozen prior to testing.

Stage of Protocol Development: Preliminary protocols for the juvenile polychaete test were developed as the test results reported herein were being generated (Johns 1988). These methods have yet to be tested in other field studies. An interim protocol based on the initial work by Johns (1988), has been developed through an experts workshop (in April 1989) sponsored by Ecology and EPA.

4.3.2. Indigenous Biota

Available Tests—Two kinds of evaluations of the effects of toxic chemicals on indigenous biota have been commonly used in Puget Sound 1) evaluations of benthic macroinvertebrate assemblages, and 2) bioaccumulation in benthic organisms. For example, both kinds of evaluations have been used in Superfund investigations of Commencement Bay and Eagle Harbor (Barrick et al. 1985; Beller et al. 1986; Becker et al. 1988) and in assessments of Elliott Bay and Everett Harbor (Beller et al. 1988; Pastorok et al. 1988). Evaluations of benthic assemblages have included classification analyses of species abundances and pairwise statistical comparisons of the abundances of major taxa (i.e., total taxa, Polychaeta, Mollusca, Crustacea) between potentially impacted and reference stations. In addition, the major taxa evaluations have been used to develop Puget Sound AET for benthic assemblages (Beller et al. 1986). Evaluations of bioaccumulation have primarily addressed potential bioaccumulative contaminants (e.g., PCBs, mercury) in edible muscle tissue of English sole (a bottom-dwelling species) and Dungeness crab. The bioaccumulation information has been used to make statistical comparisons between potentially impacted and reference areas and to evaluate potential human health risks from consuming contaminated seafood.

Evaluation of Available Tests—In this section, the advantages and disadvantages of the two available tests on indigenous biota in Puget Sound are presented. Evaluations are made with respect to the use of each test for regulatory purposes. Criteria include cost, ecological relevance, ease of use, endpoint reliability, and holding constraints.

Benthic Macroinvertebrate Assemblages—These tests generally involve the collection of sediment samples using a bottom grab or box corer and the sieving of the samples through a screen having a mesh size of 1.0 mm. The organisms retained on the screen are collected, preserved using formalin, and later identified and counted in the laboratory. The kinds of species and numbers of individuals present at each station are then evaluated to determine whether the overall benthic assemblage appears to be altered relative to reference conditions. At each station, four to five replicate field samples are generally collected and analyzed.

Cost: Relatively high (i.e., approximately $1,500 (major taxa level identifications) to $3,000 (species-level identifications) per station, assuming five replicate samples per station.

Ecological Relevance: The ecological relevance of alterations of benthic macroinvertebrate assemblages generally is high. Because these organisms live in close contact with bottom sediments and are relatively stationary, they have one of the highest potentials for exposure to sediment contaminants in marine and estuarine ecosystems. In addition, benthic assemblages typically include organisms that are very sensitive to chemical toxicity (e.g., amphipods). The high exposure potential and inclusion of sensitive species makes benthic organisms an excellent indicator group. That is, if adverse effects are not detected in these organisms, it is unlikely that they are present in most other components of the ecosystem.

In addition to being indicators for the larger ecosystem, alterations of benthic macroinvertebrate assemblages can directly influence marine and estuarine ecosystems. Many benthic organisms feed upon the organic matter in sediments and, in turn, are preyed upon by larger invertebrates (e.g., crabs) and bottom-dwelling fishes. In this manner, benthic macroinvertebrates form a primary pathway through which the energy in sediment organic matter is transferred to higher organisms (Steele 1974). Alterations of benthic assemblages could therefore substantially influence this important pathway.

Ease of Use: Benthic macroinvertebrate assemblages are relatively easy to sample and sieve with a minimal amount of training. However, the design of benthic studies and the identification of organisms generally requires extensive expertise. Because alterations of benthic assemblages can occur from such natural factors as season, depth, sediment character (e.g., grain size, organic content), salinity, and physical disturbance (e.g., wave action, current scour), it is essential that studies be designed to avoid the potential confounding influence of these natural variables on the measurement of interest (i.e., effects due to chemical toxicity). In addition, the identification of most benthic species requires detailed microscopic evaluation. Because inaccurate identifications of organisms can lead to erroneous conclusions about the presence or absence of benthic alterations, it is essential that all species-level identifications be made by a trained taxonomist, and that all identifications of higher taxa be made under the supervision of a taxonomist.

Endpoint Reliability: The endpoint (i.e., present in the sample) is unambiguous and objective, providing that sampling techniques and taxonomic identifications were conducted appropriately. Endpoint reliability is enhanced by re-sorting 20 percent of each sample (ensure that $\geq$95 percent of the organisms were removed), verifying all taxonomic identifications through comparisons with a reference collection or examination by experts, and having 5 percent of all species re-identified by different taxonomists (to ensure that $\geq$95 percent of the total number of species were identified correctly).

Holding Constraints: Immediately after collection, benthic organisms are fixed in formalin. Organisms are later transferred to alcohol for long-term storage. If fixation and preservation are conducted properly and samples are maintained during archival, benthic samples can be held for months to years before being analyzed.

Bioaccumulation—This evaluation involves the measurement of chemical contaminants in the tissue of organisms. For fishes and crabs, contaminants generally are measured in edible muscle tissue to evaluate contaminant transfer to humans and liver or hepatopancreas tissue (respectively) to evaluate the chemicals entering the organisms. For bivalves, contaminants generally are measured in all soft tissue to evaluate both of the above concerns. Although possible, measurement of bioaccumulation in small benthic macroinvertebrates (e.g., polychaetes, amphipods) has rarely been conducted in Puget Sound.

Cost: Moderate to high, depending upon the kinds and numbers of chemicals analyzed for (e.g., approximately $300-$400 for PCBs and mercury, and $1,500-$2,000 for all EPA priority pollutants).

Ecological Relevance: The ecological relevance of bioaccumulation is important largely for evaluating the transfer of contaminants through food webs. One aspect of this transfer involves the evaluation of risks to humans from consuming contaminated seafood. Because it often is not known how the presence of contaminants in tissue influences the affected organisms, the relevance of this process with respect to causing adverse ecological effects is uncertain. However, U.S. EPA (1985b) recommends that any chemical having a high potential for bioaccumulation should be of concern until it can be demonstrated that it does not cause adverse effects.

Ease of Use: Bioaccumulation can be measured objectively and relatively accurately using standardized chemical protocols. However, measuring organic chemicals in tissue is sometimes more difficult than in sediments, because lipid material in tissue can interfere with chemical extractions. In addition, a number of organism-specific factors can influence the levels of observed bioaccumulation, including age, sex, reproductive state, physiological state, and lipid content (Phillips 1980). To ensure that meaningful results are obtained during bioaccumulation surveys, it is essential that all of the organism-specific factors be accounted for when designing each survey. Because numerous contaminants are lipophilic (i.e., attracted to lipid material), it is necessary to measure the lipid content in all studies of bioaccumulation. Results can thereby be corrected for differences in lipid material to remove any potential confounding influence of this variable on the chemical measurements used for bioaccumulation evaluations.

Endpoint Reliability: The endpoint (i.e., chemical concentration) is measured objectively and with reasonable accuracy, assuming appropriate chemical protocols are followed. QA/QC measures such as analyses of replicates, blanks, and standard reference materials can help ensure that chemical measurements are conducted properly.

Holding Constraints: Immediately after collection, tissue for bioaccumulation analysis is resected (if necessary) and frozen. Once frozen, tissues can be held for at least 6 months prior to analysis.

4.4. RECOMMENDED BIOLOGICAL TESTS

In this section, recommendations are made as to which kinds of biological tests are considered appropriate for evaluating sediment quality in Puget Sound. Recommendations are based largely on the evaluations described in Section 4.3. For each test, recommendations are also made as to how the test should be used in assessing sediment quality and, if appropriate, how the performance of each test can be improved.

4.4.1. Sediment Bioassays

A summary of the rankings of the eight candidate sediment bioassays is presented in Table 6 with respect to each of the eight evaluation criteria described in Section 4.3.1. Of these tests, the amphipod reburial test (for both test species), the juvenile bivalve test, and the echinoderm embryo chromosomal abnormality test are not recommended for future use. All of these tests were found to have only low to moderate levels of cost-effectiveness. In addition, the ecological relevance of all but the juvenile bivalve test was also considered to be only low to moderate. Although the ecological relevance of the juvenile bivalve test was considered to be high, the test endpoint (i.e., mortality) is redundant with that of the more cost-effective amphipod tests.

Both amphipod mortality tests are recommended for immediate use. Although these tests exhibited only moderate levels of cost-effectiveness, both ranked high with respect to ecological relevance, ease of use, and endpoint reliability. The *R. abronius* test also ranked high with respect to organism availability and stage of protocol development. The major uncertainty with respect to routine technical application of the *E. estuarius* test is organism availability in the Puget Sound region. If it is determined that local collection of this species is impractical, then the use of an alternate estuarine amphipod (e.g., *E. washingtonianis*) should be evaluated for testing sediments with interstitial salinities lower than 25 ppt.

The juvenile polychaete test is recommended for use, pending future test development and standardization of the test protocols. Although cost-effectiveness of both the mortality and biomass endpoints was found to be relatively low, further development of the test protocols may lead to improvements in sensitivity and power and reductions in cost. The primary reasons to further develop this bioassay are that it ranked moderate to high with respect to all evaluation criteria except cost-effectiveness and stage of protocol development, and that it is the only candidate

TABLE 6. SUMMARY EVALUATION OF BIOASSAY PERFORMANCE[a,b]

Bioassay/Endpoint	Cost-Effectiveness	Ecological Relevance	Ease of Use	Organism Availability	Endpoint Reliability	Field Validation	Holding Constraints	Protocol Development
Acute Tests								
Rhepoxynius mortality	M	H	H	H	H	M	M	H
Rhepoxynius nonreburial	M	M	H	H	H	nd[c]	M	H
Eohaustorius mortality	M	H	H	?[d]	H	nd	M	M
Eohaustorius nonreburial	M	M	H	?[d]	H	nd	M	M
Panope mortality	L	H	M	M	H	nd	H	L
Crassostrea abnormality	?[e]	M	M	M	M	H	H	H
Dendraster developmental abnormality	M	M	M	M	M	nd	H	M
Dendraster chromosomal abnormality	L	L	M	M	M	nd	H	M
Photobacterium (Microtox) - saline	H	L	M	H	H	M	H	H
Photobacterium (Microtox) - organic	H	L	M	H	H	nd	H	H
Chronic Test								
Neanthes mortality	L	H	M	M	H	nd	M	L
Neanthes biomass	L	M	M	M	H	nd	M	L

[a] Table modified from Pastorok and Becker (1988).

[b] H = High; M = Moderate; L = Low. High scores indicate that a test has desirable qualities. Evaluation criteria are explained in text.

[c] nd = No Data.

[d] The abundance and distribution of this species in Puget Sound is uncertain.

[e] Not determined by Pastorok and Becker (1988).

bioassay that addresses chronic effects. The latter characteristic is particularly important, given the Authority's goal of protecting against both acute and chronic adverse biological effects.

The bivalve larvae and echinoderm embryo developmental abnormality tests are recommended for use with minor revision to the test protocols. To enhance the precision of the abnormality endpoint, a minimum number of larvae (e.g., 40-100) should be evaluated for each replicate analysis. Use of a consistent sample size among replicates within a study should also be evaluated. Cost-effectiveness was found to be moderate for the echinoderm embryo test, but could not be evaluated for the bivalve larvae test. Two major limitations are evident for these tests. First, unacceptably high mortality in the negative controls can sometimes limit routine technical application of the bivalve larvae test. Second, because spawning adults of both test species are not available throughout the year, alternate species (e.g., *M. edulis*, *S. purpuratus*, or *S. droebachiensis*) are recommended for use during particular time periods so the tests can be applied during a greater portion of the year. Because the responses of the primary and alternate test species have not been adequately intercalibrated, it is unknown whether they can be used interchangeably. It therefore is recommended that intercalibration studies be conducted to resolve these uncertainties.

Both Microtox tests are recommended for use because they ranked highly with respect to cost-effectiveness and several other evaluation criteria (i.e., ease of use, organism availability, endpoint reliability, holding, and stage of protocol development). However, because these tests are considered to have a low ecological relevance, it is recommended that they be used only as screening tools. The Microtox tests are considered useful as screening tools because they were found to have high values of sensitivity by Pastorok and Becker (1989).

4.4.2. Indigenous Biota

Of the two tests on indigenous biota evaluated in Section 4.3, only the test of effects on benthic macroinvertebrate assemblages is recommended for use. Bioaccumulation in not recommended because of its moderate to high cost and uncertain ecological relevance. However, use of bioaccumulation to assess human health risks may be appropriate (see Section 5.2.4).

Despite its high cost, the test of effects on benthic macroinvertebrate assemblages is recommended for use because of its high ecological relevance and incorporation of chronic effects. The latter characteristic is particularly important because of the Authority's goal of protecting against both acute and chronic adverse biological effects. Before this test is used as a routine technical tool, it is recommended that two factors be evaluated further. First, a list should be developed of the kinds of habitats that are appropriate for evaluation using this test. Second, minimal requirements for reference conditions and acceptable reference areas in Puget Sound should be specified.

4.5. CLASSIFICATION OF SEDIMENT QUALITY BASED ON BIOLOGICAL EFFECTS

4.5.1. Introduction

A scheme to classify sediments with respect to the presence or absence of adverse biological effects is presented in this section. The key elements of this classification scheme involve the integration of chemical criteria and biological testing. This scheme will allow sediments throughout Puget Sound to be inventoried with respect to sediment quality and thereby assist in sound-wide sediment management activities.

As much as possible, the classification scheme is consistent with similar schemes developed already by regional agencies. In this manner, decisions made using all schemes will be relatively consistent. In addition, the scheme recommended here will indirectly benefit from the considerable amount of technical and managerial expertise already used to develop the existing schemes.

One major enhancement to existing classification schemes is the inclusion of chronic effects as an integral component. The importance of chronic effects in the scheme considered here derives from the Authority's goal of "no adverse effects" in Puget Sound. Because chronic effects generally occur at lower levels of chemical toxicity than do acute tests, it is anticipated that the classification scheme recommended here may identify more stations as exceeding criteria than would other schemes based largely on acute effects. However, the recommended scheme may be similar to approaches that incorporate a "safety factor" to account for potential chronic effects.

4.5.2. Recommended Classification Scheme

The general elements of the recommended classification scheme are presented in Figure 5. At all stages, various options exist to modify or refine the scheme. Many of these options are discussed in Section 4.5.3.

The basic goal of the recommended scheme is to classify sediments as having or not having adverse effects, and to identify those sediments having adverse effects with respect to whether the effects are chronic or acute. It is expected that sediments having no adverse effects would pose little or no risk to aquatic biota, whereas sediments having acute effects would pose a substantial risk and possibly warrant some kind of management action. The environmental risks associated with sediments having only chronic effects are relatively less certain, but warrant consideration to attain the Authority's goal of protecting against adverse acute and chronic effects.

Determination of No Effects Concentrations—The first step in the recommended classification scheme involves the determination of whether any chemical in a particular sediment sample exceeds its NEC. The NEC is the concentration below which adverse effects are not expected to occur. If it is assumed that chronic effects generally occur at lower chemical concentrations than acute effects, NEC based on chronic effects should be environmentally protective and account for both chronic and acute effects. Because there is little information regarding chronic effects in Puget Sound, it is recommended that NEC be developed initially using ACRs (applied to chemical AET based on existing acute bioassays) and chemical AET based on benthic macroinvertebrates (which incorporate chronic effects). It is recommended that the ACR used to develop NEC equal 10, which is consistent with EPA recommendations for complex effluents (U.S. EPA 1985b) and, according to information collected for *Mysidopsis bahia* and *Ampelisca abdita* (Gentile et al. 1988), should generally account for chronic mortality.

At least three options are available for using an ACR approach to develop NEC. These options include the following:

1. Application of the ACR to the LAET for the acute bioassays and use of that value or the benthic AET, whichever is lower, as the NEC. Although this approach is very protective, it would probably generate many NEC with values comparable to or below those commonly found in reference areas.

2. Application of the ACR to the highest AET (HAET) for the acute bioassays and use of that value or the benthic AET, whichever is lower, as the NEC. Although this approach is less protective than Option 1, it would more frequently result in NEC that exceed reference area concentrations than would Option 1. In addition, this approach is consistent with the methods used by PSDDA when evaluating options for dredged material disposal in Puget Sound (Phillips et al. 1988). Using the PSDDA approach, a sediment sample is analyzed chemically, and each contaminant concentration is compared with its respective screening level (SL) concentration. If no SL are exceeded, the sample is considered suitable for unconfined open water disposal without biological testing. However, if one or more SL are exceeded, biological testing must be conducted and all biological criteria must be passed before the sediment sample can be considered suitable for unconfined open water disposal. The SL for each chemical is set at 10 percent of the HAET,

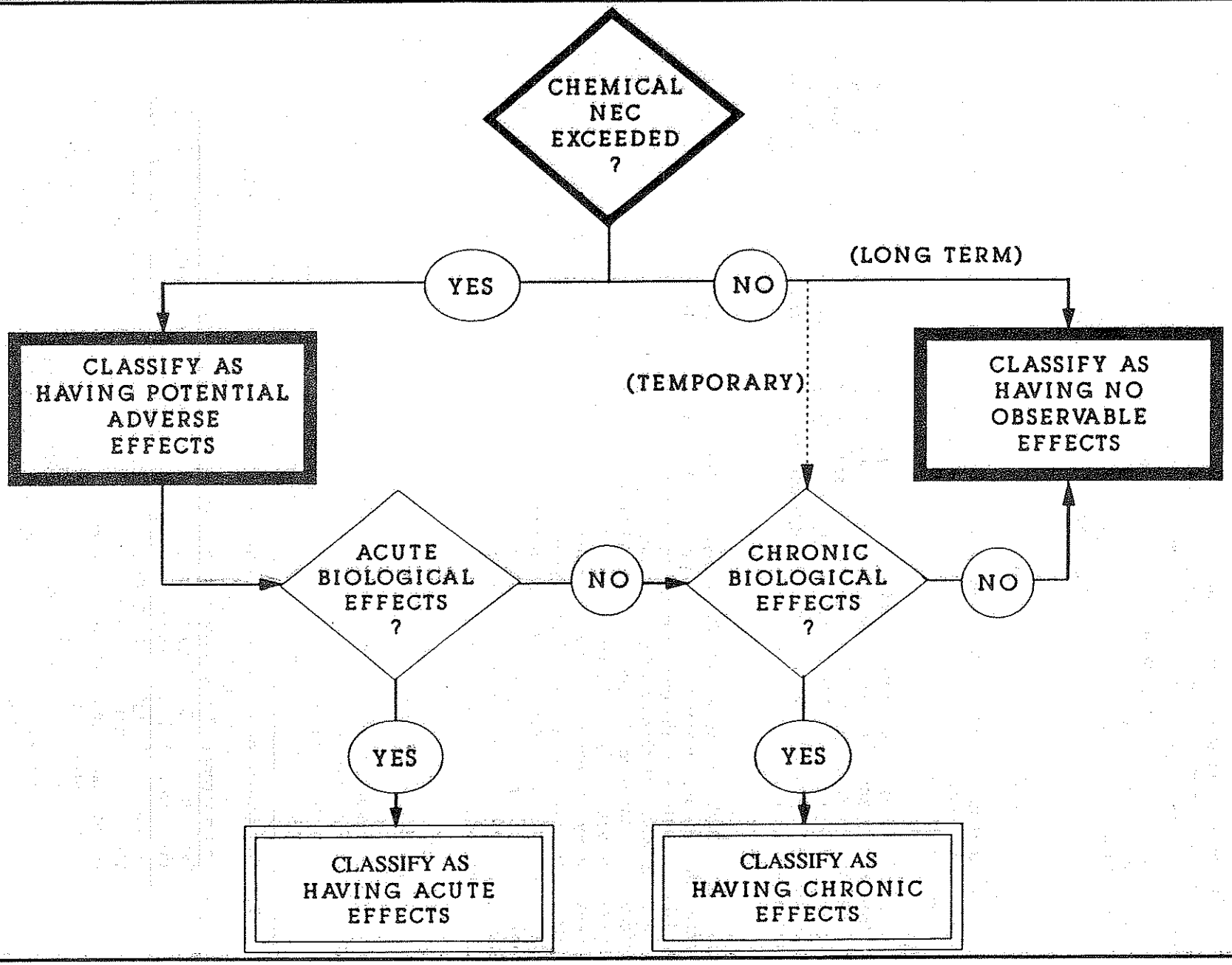


Figure 5. Summary of the detailed recommended scheme for sediment quality classification using No Effect Concentrations (NEC) criteria

provided that 1) the value is greater than or equal to the average concentration for the chemical in Puget Sound reference areas, and 2) the value is less than the LAET.

3. Application of the ACR to the amphipod mortality AET and use of that value or the benthic AET, whichever is lower, as the NEC. This approach would account primarily for chronic mortality, as estimated from the ACR applied to the acute mortality evaluated in the amphipod bioassay.

To help ensure that NEC are not unreasonably sensitive or inefficient, it is recommended that they be no lower than the 90th percentile for chemical concentrations measured in all Puget Sound reference areas. Reference areas are defined as areas removed from major contaminant sources that have relatively low observed levels of both sediment contamination and adverse biological effects. The use of reference conditions to set lower limits for NEC will help prevent test sediments similar to Puget Sound reference sediments from being classified as having adverse effects, and thereby ensure that management activities are focused on sediments having the highest priority (i.e., sediments having chemical concentrations that exceed those in most Puget Sound reference areas).

Because the NEC are relatively protective of the environment, it is unlikely that samples passing this screening step would pose a substantial environmental risk. However, NEC derived primarily on the basis of acute bioassays and effects on benthic macroinvertebrate assemblages (i.e., primarily acute and chronic mortality), may not completely account for the most sensitive kinds of chronic effects (e.g., growth, reproduction). It therefore is recommended that samples passing the NEC screening step initially be confirmed as not having adverse effects by being subjected to a chronic bioassay. This confirmation step should be considered temporary until a sufficient database exists to validate the use of existing NEC to account for most kinds of chronic effects. Once this validation has occurred, the confirmation step can be eliminated and samples passing the NEC screening step can be classified as not having adverse effects without subsequent biological confirmation. Samples that pass this confirmation step are classified as not having adverse effects, whereas samples that fail are classified as potentially having adverse effects.

Because NEC may not be available for all chemicals that sometimes may be present in samples, chemical screening alone may sometimes not be sufficiently protective. Therefore, if there is reason to suspect that a sample may contain potential problem chemicals that are not accounted for using NEC, biological testing could be required on a case-by-case basis despite the fact that no chemicals in the sample exceed existing NEC.

If a sample is classified as potentially having adverse effects based on NEC comparisons, and there is reason to believe the sample may not have acute or chronic effects, then it could be subjected to biological testing to confirm its designation based on NEC. The test series could consist of sequential analysis using acute and chronic biological tests. If the sample fails the acute test, it would be identified as having acute effects. If it passes the acute test, it would be subjected to a chronic bioassay and identified as having chronic effects or classified as not having adverse effects based on the results of the latter test.

Reliability Tests for NEC Options—Reliability tests were performed to compare the reliability of the three options for using an ACR approach to develop NEC (Table 7). The reliability of existing PSDDA SL values was also compared to that of the alternative NEC values. All of these approaches result in values that are highly sensitive (i.e., >90 percent sensitivity in identifying impacted stations). Their efficiency in only identifying impacted stations (according to available acute bioassays and analyses of benthic macroinvertebrates) was relatively low (i.e., typically 30-60 percent). Of the various options, including PSDDA SL values, Option 2 (application of the ACR to the HAET for the acute bioassays and use of that value or the benthic AET, whichever is lower, as the NEC) was the most efficient.

TABLE 7. COMPARISON OF RELIABILITY FOR PSDDA SCREENING LEVEL AND HAET MODIFIED BY ACUTE-TO-CHRONIC RATIO OF 10[a]

	PSDDA Screening Level		HAET Modified by ACR		Amphipod AET Modified by ACR		LAET Modified by ACR	
Test Type	Sensitivity	Efficiency	Sensitivity	Efficiency	Sensitivity	Efficiency	Sensitivity	Efficiency
Benthic Infaunal Abundance (n=201)	97% (104/107)	56% (104/186)	95% (102/107)	56% (102/181)	95% (102/107)	56% (102/183)	99% (106/107)	56% (106/190)
	Overall Reliability[b]: 58%		Overall Reliability: 59%		Overall Reliability: 57%		Overall Reliability: 58%	
Amphipod Mortality Bioassay (n=287)	93% (99/106)	38% (99/258)	96% (102/106)	41% (102/251)	95% (101/106)	40% (106/253)	98% (104/106)	38% (104/269)
	Overall Reliability: 42%		Overall Reliability: 47%		Overall Reliability: 45%		Overall Reliability: 42%	
Microtox Bioassay (n=50)	100% (29/29)	58% (29/50)	100% (29/29)	58% (29/50)	100% (29/29)	58% (29/50)	100% (29/29)	58% (29/50)
	Overall Reliability: 58%		Overall Reliability: 58%		Overall Reliability: 58%		Overall Reliability: 58%	
Oyster Larvae Abnormality Bioassay (n=56)	100% (17/17)	31% (17/55)	100% (17/17)	30% (17/56)	100% (17/17)	30% (17/56)	100% (17/17)	30% (17/56)
	Overall Reliability: 32%		Overall Reliability: 30%		Overall Reliability: 30%		Overall Reliability: 30%	

[a] The resulting ACR-modified values were further adjusted, if necessary, to ensure that they were not greater than the corresponding benthic AET value and no lower than the 90th percentile reference concentration for the chemical.

[b] Overall reliability was calculated as the sum of the number of correctly predicted impacted stations and the number of correctly predicted nonimpacted stations divided by the total number of stations.

4.5.3. Options for the Recommended Classification Scheme

Various options are possible for modifying or refining the recommended classification scheme. Some of the major options include the following:

- Because the initial chemical screening criteria do not use an NEC that accounts specifically for chronic effects, use a lowest effects concentration based on the LAET of all available biological tests. This approach would avoid making assumptions about chronic effects (e.g., use of an ACR of 10) that could result in test sediments similar to Puget Sound reference sediments being classified as having adverse biological effects. Because the benthic AET would be considered, chronic effects would be addressed to some extent. The LAET could also be set by a very sensitive acute sublethal test (e.g., Microtox bioassay) that has an AET lower than the benthic AET for many chemicals.

- Do not use chronic testing at all, but use proposed NEC without confirmation by chronic bioassays. This would require making the untested assumption that an ACR of 10 and effects on benthic macroinvertebrates account for most kinds of chronic effects.

- Use Puget Sound reference areas to set a lower limit for NEC, or use a limit that is less restrictive than the 90th-percentile limit (e.g., mean reference values as used by PSDDA). This could result in classifying many reference sediments and sediments having characteristics similar to reference sediments as potentially having adverse effects and thereby substantially increase the number of relatively low priority areas requiring sediment management.

- Because chronic tests can be relatively expensive, use a very sensitive acute test (e.g., Microtox) as a surrogate for chronic testing. This could require initial tandem testing of the acute and chronic tests to ensure that the acute test is an adequate surrogate.

- Because chronic endpoints can be very sensitive and thereby potentially be induced by factors other than chemical toxicity, confirm any effect discriminated by a chronic test by replicating the test. The lack of a chronic effect would not require replication. Although replication would increase confidence in the test results, it would be expensive.

- Because no acute or chronic test can account for all possible environmental effects, use two or more of each kind of test for each confirmation step. This could require an intercomparison among the different tests and comparisons with effects on indigenous biota to determine how the various tests relate to each other and which combinations of tests provides the broadest range of information (i.e., minimum overlap among tests).

- Instead of simply making effect/no effect determinations using acute and chronic tests, use the magnitude of each effect to further subdivide the sediment rankings with respect to severity of effects (e.g., severe, moderate, and low chronic effects). This option would allow sediments to be ranked on a finer scale of severity.

Although various combinations of the above options could produce a wide variety of classification schemes, the purpose of all of them is the same: to classify sediments as having or not having adverse effects, and to identify those sediments with adverse effects according to a graduated scale.

4.6. CONCLUSIONS AND RECOMMENDATIONS

In this section, a summary is presented of the major conclusions and recommendations derived from the preceding evaluations of the role of biological testing in developing sediment quality standards. Recommended options for some of the information presented below in Section 4.6.2 have been described earlier in Section 4.5.3.

4.6.1. Selection of Biological Tests

The major conclusions regarding selection of biological tests are:

- Of the candidate bioassays evaluated in this report, the amphipod nonreburial test (both test species), the juvenile bivalve test, and the echinoderm embryo chromosomal abnormality tests are not presently recommended for use in confirming the toxicity of Puget Sound sediments with chemical concentrations exceeding AET.

- Both amphipod mortality tests are recommended for immediate use. However, the availability of *E. estuarius* in the Puget Sound region should be determined. If it is found that local collection of this species is impractical, then the use of an alternate estuarine amphipod (e.g., *E. washingtonianis*) should be evaluated for testing sediments with interstitial salinities lower than 25 ppt.

- The juvenile polychaete test is recommended for use, pending future test development and standardization of the test protocols. This bioassay should be developed further primarily because it is the only candidate bioassay that directly addresses chronic effects.

- The bivalve larvae and echinoderm embryo developmental abnormality tests are recommended for use with minor revisions to the test protocols. To enhance the precision of the abnormality endpoint, a minimum number of larvae (e.g., 40-100) should be evaluated for each replicate analysis. In addition, the validity of using alternate test species should be determined.

- Both Microtox tests are recommended for use only as screening tools to determine which samples require further analysis using the other bioassays.

- The usefulness of any of the bioassays selected for regulatory use would be strengthened by evaluating intralaboratory and interlaboratory variability of bioassay responses.

- Bioaccumulation is not presently recommended for evaluation of effects on indigenous organisms, but may be appropriate for human health risk assessment.

- Evaluation of effects on benthic macroinvertebrate assemblages is recommended as a measure of effects on indigenous organisms. However, a list should be developed of the kinds of habitats that are appropriate for evaluation using this test. In addition, minimal requirements for reference conditions and acceptable reference areas in Puget Sound should be specified.

4.6.2. Classification Scheme of Sediment Quality Based on Biological Effects

The major conclusions regarding the recommended classification scheme are:

- To develop a classification scheme for identifying and designating sediments having observable acute or chronic adverse biological effects, it is recommended that biological testing be integrated with chemical screening criteria.

- Chemical screening criteria should identify sediments having a very low potential for causing adverse effects (i.e., NEC)

- NEC should be based on environmental effects-based sediment quality values that have been tested in Puget Sound (for example, Puget Sound AET).

- Because there is little information regarding chronic effects in Puget Sound, NEC should initially be developed using the Puget Sound AET based on benthic macroinvertebrates and an ACR of 10 relative to the acute bioassays used to develop Puget Sound AET.

- The 90th percentile of chemical concentrations from Puget Sound reference areas should set the lower limit for each NEC.

- Biological testing using acute and chronic sediment bioassays and effects on indigenous organisms should be used to confirm the designation of sediments with chemical concentrations above NEC, and to classify the sediments in accordance with the observed biological results.

- The proposed classification scheme will classify sediments in Puget Sound as having or not having adverse effects and identify those sediments with adverse effects according to whether the effects are chronic or acute. In doing so, the classification scheme will facilitate sediment management activities in the sound.

- The use of chemical screening criteria as an integral part of the proposed classification scheme will focus biological testing on the sediments having the greatest potential for causing adverse effects.

5. DEVELOPMENT OF HUMAN HEALTH GUIDELINES

Chemically contaminated sediments in Puget Sound may pose a potential health risk to humans who use the sound for recreation and seafood resources. Element P-2 of the 1987 Plan specifies that criteria for maximum allowable contaminant concentrations in sediments be developed to protect public health. These criteria should be based on consideration of environmental pathways leading to exposure of humans to contaminants and their potential risks. Sediments that fail the criteria would be classified as possibly posing a significant health risk. Determination of a significant human health risk is difficult because of the inability to measure directly the human health effects potentially resulting from contaminated sediments and to account for the many confounding factors (e.g., exposure to chemicals in air and drinking water, dietary factors, and variation in sensitivity to chemical effects among individuals). Although many assumptions and uncertainties are involved in predicting potential health effects (e.g., extrapolation of the results of animal cancer bioassays to humans), human health specialists are able to gain perspective on the implications of environmental contamination through a modeling procedure called risk assessment. Despite its limitations, risk assessment is a state-of-the-art tool for management of environmental contamination.

Options for development of sediment contamination guidelines based on consideration of potential human health risks are described in this section. Because of the importance of potential health effects of contaminated sediments, it is recommended that the sediment quality standards eventually incorporate guidelines derived from available health risk assessment techniques. The alternatives for use of human health guidelines in conjunction with proposed sediment quality standards for biological resources are also presented. However, critical uncertainties in modeling potential human health risks associated with sediments preclude adoption of guidelines for allowable sediment concentrations of contaminants at present. Therefore, developmental needs for application of human health guidelines are addressed.

5.1. DEFINITIONS AND OVERVIEW OF RISK ANALYSIS

An introduction to risk assessment concepts is provided below. Additional details on risk assessment approaches and issues in development of sediment standards can be found in Appendix A.

5.1.1. What is Risk?

Risk is essentially the probability of harm. In the present context, risk is the chance (probability) that potential exposure of humans to toxic chemicals in contaminated sediments of Puget Sound will result in adverse health effects. Potential health effects of concern include, for example, cancer and birth defects from chronic exposure to chemicals such as PCBs and PAH.

5.1.2. How is Risk Measured?

Actual health effects result from human exposure to a toxic chemical or mixture of chemicals. Epidemiological studies directly measure the occurrence of health effects in a human population. Epidemiological studies that attempt to identify the environmental causes of disease are most successful when there is a severe outbreak of a disease shortly after exposure in a well-defined subpopulation. In such cases, a clear link between chemical exposure and an observed health effect may be established. In contrast, appearance of possible effects of relatively low levels of toxic chemicals in Puget Sound sediments could require long-term (e.g., decades) exposure. Also, environmental health scientists and the public may be concerned about chemical hazards that

potentially affect only a small percentage (e.g., less than 1 percent) of a human population. Such low-level risks may be impossible to measure directly.

Because relatively low risks of concern cannot be measured directly and reliably in human studies, health scientists often rely on prediction of potential health risks by using mathematical models to extrapolate the results of laboratory studies. However, predicting potential health effects on humans involves many assumptions, leading to uncertainty in the conclusions of any evaluation of health risks. Consequently, health scientists usually calculate plausible upper-limit estimates of risk, so that any possible errors resulting from assumptions will be on the side of protecting human health. Such predictions form the basis for the development of human health guidelines for sediments recommended in later sections.

5.1.3. How Can Risk be Estimated?

Risk assessment is a scientific procedure used to estimate the probability of adverse health effects that may result from exposure to toxic chemicals. In risk management, the results of a risk assessment are interpreted and translated into public policy or actions to control risks (e.g., development of sediment standards). Many factors are considered during risk management, including the degree of confidence in a risk estimate, the size of the population at risk, regulatory requirements, and in some cases, public perceptions of risk.

As noted earlier, many assumptions and uncertainties may enter into an evaluation of human health risk. Scientific knowledge of the effects of toxic chemicals on humans is still rudimentary. Much of the existing information is extrapolated from results of laboratory tests performed on animals such as rats and mice. Although toxicologists are faced with many uncertainties when estimating the potential human health risks based on such data, the risk assessment approach provides a framework for consistent and systematic evaluation of health risks, with clear statements about assumptions and uncertainties.

Assessing risks of human exposure to toxic chemicals consists of the following four major steps:

- **Hazard Identification**—Qualitative evaluation of the potential for a substance to cause adverse health effects (e.g., birth defects, cancer) in animals or in humans
- **Dose-response Assessment**—Quantitative estimation of the relationship between the dose of a substance and the probability of an adverse health effect
- **Exposure Assessment**—Characterization of the populations exposed to the toxic chemicals of concern; the environmental transport and fate pathways; and the magnitude, frequency, and duration of exposure
- **Risk Characterization**—Estimation of risk for the health effect of concern based on information from the first three steps.

The concepts of exposure and dose, as defined below, are central to risk assessment:

- **Exposure**—Contact by an organism with a chemical or physical agent
- **Dose**—The amount of chemical uptake by an organism over a specified time as a consequence of exposure.

Estimates of human cancer risk from chemical exposure are typically expressed as the probability that each exposed individual will experience cancer within his or her lifetime (usually assumed to be 70 years). Probabilistic risk assessment methods for carcinogens are better developed than those for noncarcinogens (i.e., systemic toxicants). For noncarcinogenic effects, there is usually a threshold dose (i.e., a dose below which no adverse biological effects are observed in a human

population or an animal bioassay). The threshold dose is termed the No-Observed-Adverse-Effect-Level (NOAEL). The NOAEL is used to derive a Reference Dose (RfD), an estimate of the daily exposure to the human population (including sensitive subpopulations) that is unlikely to produce an appreciable risk of adverse health effects during a lifetime. The lack of a demonstrated threshold in dose-response relationships for carcinogens implies a finite risk of cancer even at very low doses of the carcinogen (U.S. EPA 1980; U.S. OSTP 1985). Thus, a model of the dose-response relationship is applied to predict carcinogenic risk. The slope of a linear dose-response relationship is used to derive an estimate of the carcinogenic potency of a substance.

Risk estimates are associated with specific exposure pathways and routes. For example, when evaluating health risks from chemically contaminated fish and shellfish in Puget Sound, the individuals considered to be at risk are those who consume seafood harvested from contaminated areas. Also, risk estimates account for excess lifetime risks due only to the specific exposures of interest (e.g., consumption of contaminated seafood), not the total risk from all exposures to toxic chemicals (e.g., through drinking water, diet other than seafood, or smoking). Nevertheless, it is important to compare the estimated risks from a specific activity with those associated with other exposure pathways (e.g., see Appendix A). Comparison with other risks provides perspective in interpreting the relative importance of the specific risk of concern.

5.1.4. What are the Exposure Pathways of Concern?

In the Puget Sound region, the most important way in which humans are exposed to toxic chemicals associated with sediments is through consumption of seafood organisms living in areas with contaminated sediments (e.g., bottomfish, crabs, clams). For most chemicals, direct contact with sediments or absorption of contaminants from water overlying contaminated sediments are probably relatively unimportant as human exposure pathways because of infrequent visits to beach areas, the short duration of exposure, and the small area of skin surface exposed. Direct exposure to sediments containing high concentrations of very potent carcinogens such as 2,3,7,8-tetrachlorodibenzodioxin is of potential concern. Although high concentrations of dioxins have not been found in intertidal and shallow subtidal sediments of Puget Sound, limited measurements of dioxin concentrations in Puget Sound sediments have been made. Even if other exposure pathways are important in some cases, consumption of contaminated fish and shellfish is likely to be the primary exposure pathway for toxic organic chemicals (including dioxins) with a high potential for accumulating in biological tissues (Burmaster et al. 1987; Whitmyre et al. 1987).

Some possible approaches to development of sediment standards based on human health risk assessment are presented in the next section.

5.2. OPTIONS FOR DEVELOPING HUMAN HEALTH GUIDELINES

The options outlined below for developing human health guidelines are general approaches that incorporate some or all of the steps of risk assessment defined earlier. Because the options are not mutually exclusive, more than one option may be integrated into the proposed sediment quality standards. Some options (i.e., Options 1, 4, and 5) address both carcinogenic and noncarcinogenic effects, whereas others (i.e., Options 2 and 3) address only carcinogenic effects.

5.2.1. Option 1: Hazard Assessment Literature Review

Option 1 involves using available information on the toxicity of chemicals detected in sediments to qualitatively evaluate the potential hazards to human health. Toxicity information for chemicals is available from many sources, including the EPA Integrated Risk Information System (IRIS). Under Option 1, available summaries of toxicity would be used to assess hazards. (Selected toxicity data are also compiled in Appendix A.) A concise format for hazard assessment

is recommended based primarily on information in EPA's IRIS and supportive health risk assessment documents.

Methods used to rank chemicals by relative toxic potency (e.g., carcinogenic potency or RfD) could be used to determine the relative importance of chemicals as potential human health hazards. For example, the relative hazard of several chemicals detected in a sediment could be ranked using this approach. Squire (1981), Theiss (1983), U.S. EPA (1985a), and Ames et al. (1987) describe various approaches for ranking carcinogens. The various ranking methods would need to be evaluated further to select an appropriate approach. However, because this option only provides information on relative hazards of chemicals, use of one of the other options explained below would be necessary to establish an absolute level of concern for each chemical.

Option 1 has the following advantages, disadvantages, and developmental needs:

- Advantages
  - Based on available literature
  - Information available on various potential health effects
  - Inexpensive.
- Disadvantages
  - Assesses relative hazard without providing specific guidelines for contaminant concentrations in sediments
  - Usually addresses single-chemical effects and ignores interactions among chemicals in complex mixtures
  - Possibly limited information for health effects of some chemicals in sediments
  - Ignores site-specific bioavailability of chemicals.
- Development Needs
  - Development of a state registry for toxicity profiles for contaminants of concern, possibly based on incorporation of key toxicological data from EPA IRIS into the PSEP pollutant of concern matrix
  - Evaluation of available systems for ranking chemical hazards and possibly development of a new system.

5.2.2. Option 2: Sediment-Specific Hazard Assessment

Option 2 involves directly measuring genotoxic effects of contaminated sediments in short-term bioassays, and using the results as a relative indicator of human cancer hazard associated with exposure to contaminated sediments. The development of guidelines for sediment concentrations of chemicals using this option as a sole test would require development of relationships between sediment chemistry and endpoints of the tests (e.g., genotoxicity) and a determination that direct sediment contact is an important exposure route for human health effects. At present, these requirements are not satisfied. Nevertheless, Option 2 may presently provide a screening tool for use in conjunction with other tests that relate to the primary human exposure pathway of food-chain bioaccumulation (e.g., see Option 4). Short-term bioassays could also be performed directly on tissue extracts from marine organisms (e.g., Sparks et al. 1981; Pittinger et al. 1987).

Potential short-term bioassay tests that show promise for testing of sediment or tissue extracts include:

- Ames test for mutagenicity (Ames et al. 1973)
- Sister chromatid exchange and chromosomal aberrations in hamster ovary cells (Galloway et al. 1985)
- Mouse lymphoma cell mutagenesis assay (Myhr et al. 1985)
- Cellular transformation in cultured mammal cells (Heidelberger et al. 1983).

These tests are appropriate for providing evidence of mutagenicity and potential carcinogenicity (Appendix A). However, short-term genotoxicity tests do not provide direct evidence of carcinogenicity or mutagenicity of chemicals in humans. Past studies have demonstrated that despite several limitations the Ames test in particular is a good predictor of carcinogenicity of a chemical in mammals (Tennant et al. 1987). Short-term bioassays can be an important supplement to risk assessment models in that they account for effects of multiple chemicals and chemicals that are not analyzed for. The short-term tests would be appropriate for sediments from single sampling sites or for composite samples from a larger sampling area.

Methods are available for analysis of tissue extracts using the Ames test (e.g., Sparks et al. 1981; Pittinger et al. 1987), but the analysis is complicated by the presence of histidine in tissues. Other short-term bioassays have not been applied widely to tissue or sediment extracts. Problems in measuring the mutagenic potential of complex mixtures of contaminants still need to be solved. For example, direct toxicity of contaminants may interfere with detection of mutagenicity. In the Ames test, antagonistic interactions among PAH compounds may mask the mutagenicity of individual PAH compounds that require enzyme-mediated activation [e.g., benzo(a)pyrene] (Fabacher et al. 1988). Calibration of short-term bioassays to various concentrations of known mutagens in tissue or sediment samples would be needed. Consequently, the Ames test or another short-term bioassay would likely require substantial development before a suitable test could be applied as part of the sediment quality standards.

The available short-term tests for mutagenicity are also relatively expensive (>$1000). The Microtox mutagenicity bioassay, which is being developed by Micronics, Inc. (Razza, J., 15 November 1988, personal communication) may be a relatively inexpensive alternative to the available short-term tests. A comparison of the Microtox mutagenicity bioassay and the Ames test should be available in approximately 6 months.

Option 2 has the following advantages, disadvantages, and developmental needs:

- Advantages
  - Measure sensitive endpoints of concern (mutagenicity/carcinogenicity)
  - Assess complex mixtures of chemicals directly
  - Assess effects of chemicals lacking toxicity data.
- Disadvantages
  - No inexpensive tests are available for chronic systemic toxicity (i.e., organ/tissue level effects)
  - Limited information is available on responses of tests to chemical mixtures
  - Inability to extrapolate dose-response data from these tests to humans. Despite this limitation, general classifications of mutagenic potency can be developed from genotoxicity data by calibrating the short-term tests against series of concentrations of known carcinogens.

- Developmental Needs
  - Further laboratory evaluation of the Ames test and possibly other short-term assays on sediments and tissues, especially for assessment of complex mixtures of chemicals
  - Comparison of Ames test and Microtox mutagenicity test (ongoing by Micronics) and development of Microtox mutagenicity bioassay for tissue extracts
  - Long-term calibration of mutagenic potential as measured by the preferred test with carcinogenic potential in rodents or other human models
  - Ongoing research by EPA and the University of Washington may be valuable in demonstrating the utility of other short-term genotoxicity tests (e.g., fish anaphase aberration) for assessing the potential human health hazards of sediments.

5.2.3. Option 3: Site-Specific Hazard Assessment

Additional indirect evidence of potential human health hazards related to sediments can be obtained by measurement of liver disease in English sole associated with contaminated sediments. English sole is the best indicator species for assessing contaminated sediments because of its apparent sensitivity to contaminants, its limited movements, its broad geographic distribution, and its association with depositional habitats that have a high potential for contamination (Appendix A). Although definitive studies of the relationship between contamination of English sole and other species have not been performed, available data suggest that the average concentration of PCBs in English sole muscle is within an order of magnitude of that found in other sportfish. However, there is little information on the relationship between the occurrence of liver cancer in English sole and contaminant concentrations in other species.

The relationship between fish liver cancer and human cancer is discussed in Appendix A. Although the prevalence of fish cancers can not be used directly to estimate human cancer risks (see Appendix A), fish disease data could provide a relative indication of the carcinogenic potency of complex mixtures of contaminants in sediments. Guidelines for concentrations of chemicals in sediments could be developed based on chemical relationships to liver disease in bottomfish (e.g., AET based on matched chemical and pathological data or similar guidelines). Interim AET for fish disease could be developed from available Puget Sound data.

Option 3 has the following advantages, disadvantages, and developmental needs:

- Advantages
  - Direct field assessment of effects
  - Uses vertebrate species
  - Measures sensitive effect of concern (carcinogenicity)
  - Accounts for multiple chemical effects.
- Disadvantages
  - Limited information on relationships of neoplasms to specific chemicals
  - Uncertain relationship of fish pathology indicator to primary human exposure pathways
  - Chemicals causing fish disease may not be accumulated in tissues eaten by humans
  - Mobility of fish.

- Developmental Needs
  - Development of AET or similar guidelines to relate significant levels of fish neoplasms to chemical concentrations in sediments
  - Development of relationships between fish disease and sediment chemistry by evaluating available data and by collecting additional data from a various areas in Puget Sound
  - Development of relationships between liver neoplasms in English sole and chemical concentrations in edible tissue of English sole and other commercially and recreationally harvested fish.

5.2.4. Option 4: Quantitative Risk Assessment to Develop Sediment Quality Standards with Confirming Bioaccumulation Test

In Option 4, the EPA risk assessment approach would be applied to develop guidelines for chemical concentrations in edible tissues of fish or shellfish. For example, a tolerable risk level would be selected and the concentration of a carcinogen (cancer-causing chemical) corresponding to the tolerable risk would be calculated based on an assumed exposure scenario (i.e., rate of seafood consumption by a hypothetical human population that harvests fish or shellfish from Puget Sound) (see Appendix A for interim guidelines). These tissue quality guidelines would be extrapolated to establish sediment quality values for concentrations of chemicals that would correspond to selected tolerable risk levels. Theoretical and mathematical models that could be used to relate concentrations of contaminants in fish to concentrations in sediments include equilibrium partitioning approaches (Lyman et al. 1987; Battelle 1988) and regression relationships (Connor 1984) (see Appendix A).

The modeling approach described above could be used initially as a screening tool in conjunction with laboratory testing of sediments. For example, the sediment quality values would serve as minimum values (i.e., protective trigger values) for determining when a direct bioaccumulation test on the sediment would be required. When the values are exceeded, clams (*Macoma* spp.) would be exposed to sediments in the laboratory following the protocols similar to those used for evaluation of dredged sediments in PSDDA. A written protocol for bioaccumulation tests of sediments is being developed by Dr. H. Lee of the U.S. EPA Office of Research and Development Laboratory at Newport, Oregon. After 30 days of exposure, contaminant concentrations would be measured in the soft tissue of the clams (including the gut). Chemical data would be compared with tissue quality guidelines derived from a risk assessment based on a hypothetical exposure scenario (e.g., Appendix A). The approach could be applied to consumption of either fish or shellfish by humans. If only shellfish (e.g., *Macoma* spp.) were used in the laboratory bioaccumulation test, then the test would serve as a worst-case assessment for some contaminants (e.g., PAH) that don't accumulate in the edible muscle of fish. If the bioaccumulation test provided confirming evidence that a potential risk to humans exceeded a chosen value, the sediment would fail the sediment quality standards. Present information would allow the full application of Option 4, including use of an equilibrium partitioning approach to derive screening-levels for chemical concentrations in sediments, to a limited number of contaminants of concern (e.g., PCBs, PAH, and DDT compounds). For other contaminants for which preliminary tissue quality guidelines are available (Appendix A), available data might not be sufficient to allow calculation of a risk-based screening level for sediments.

Option 4 assumes that consumption of fish/shellfish is the only significant pathway for potential exposure of humans to contaminants that may have been originally associated with sediments. As discussed in Appendix A, seafood consumption is the primary human exposure pathway for contaminants in sediments. Note that Option 4 is essentially similar to sediment evaluation procedures followed by PSDDA (Phillips et al. 1988). However, PSDDA evaluation procedures also allow for comparison of contaminant concentrations in clams exposed in the laboratory with tissue contamination guidelines derived from risk assessment. PSDDA's approach does not require extrapolation of tissue quality guidelines to derive sediment quality guidelines. The trigger values used for the initial evaluation of the need for bioaccumulation testing in PSDDA

were arbitrarily based on ecological effects guidelines related to AET. Under Option 4, sediment chemistry screening values would be related to human health risk estimates for selected chemicals.

Option 4 has the following advantages, disadvantages, and developmental needs:

- Advantages
  - Uses chemical data from sediment measurements
  - Addresses carcinogenic and noncarcinogenic risk
  - Addresses exposure pathway of primary concern (food-chain bioaccumulation of contaminants)
  - Provides quantitative criteria for concentrations of chemicals in sediments based on potential human health risks.
- Disadvantages
  - Uncertainty in tissue quality guidelines and sediment screening values due to assumptions of risk assessment models and extrapolation from sediments to tissue
  - May overestimate risk because of protective assumptions of risk assessment models
  - Assessment of multiple chemical effects via additive model only (i.e., by addition of individual risk values associated with single chemicals)
  - Lack of information for chemicals that are not analyzed for in sediments or are not quantified
  - Lack of dose-response data for some chemicals of potential concern.
- Developmental Needs
  - Development of relationships [e.g., bioconcentration factor (BCF)] between concentrations of chemicals in edible tissues of selected aquatic species (e.g., English sole, butter clams) and concentrations in sediments based on further sampling and analysis. Data to support Option 4 may be collected as part of the Puget Sound Ambient Monitoring Program (PSWQA 1988b)
  - Field validation of equilibrium partitioning models to predict concentrations of chemicals in tissues of aquatic organisms from concentrations in sediments
  - Uncertainty analysis of equilibrium partitioning models to predict tissue concentrations of contaminants and derivation of screening levels for chemical concentrations in sediments based on available data (mainly data for PAH and PCBs)
  - Evaluation of potential approaches to equilibrium partitioning models for metals as well as organic forms of tin and mercury
  - Evaluation of bioaccumulation potential of at least the primary contaminants (PAH, PCBs, DDT and related metabolites, *alpha*-HCH, arsenic and mercury) in *Macoma* spp. relative to harvested fish and shellfish.

Note that only the latter three developmental tasks would need to be conducted to establish interim guidelines based on protective screening values.

5.2.5. Option 5: Quantitative Risk Assessment to Develop Sediment Quality Standards

Option 5 is similar to Option 4 except that the confirmatory laboratory test for bioaccumulation of contaminants is eliminated.

Option 5 has the following advantages, disadvantages, and developmental needs:

- Advantages
  - Uses chemical data without added biological tests
  - Addresses primary exposure pathway (food-chain bioaccumulation of contaminants).
- Disadvantages
  - Similar to Option 4, but with additional uncertainty due to lack of confirmatory test.
- Developmental Needs
  - Similar to the first three items listed for Option 4.

5.2.6. Option 6: No Action

Sediment standards could be developed at present without guidelines derived from human health hazard or risk assessment.

Option 6 has the following advantages, disadvantages, and developmental needs:

- Advantages
  - Status quo maintained
  - Simplifies sediment quality standards to just biological resources criteria.
- Disadvantages
  - Possibility of significant human health risks
  - Public perception of agency inaction.
- Developmental Needs
  - None.

5.3. RECOMMENDED APPROACH TO HUMAN HEALTH GUIDELINES

The recommended approach to human health guidelines in the proposed sediment quality standards is described in the next section. In a following section, options for integrating human health guidelines with other sediment criteria are evaluated, and a preferred option is selected.

5.3.1. Development of Human Health Guidelines

The proposed approach to develop sediment standards relevant to potential human health effects consists of a combination of qualitative hazard assessment (Option 1) and quantitative risk assessment (Option 4). Options 1 and 4 are complementary. Option 1 addresses a wide variety of chemicals, but does not provide quantitative sediment standards. Under Option 4, risk assessment models based on bioaccumulation of contaminants in food chains leading to humans would be used

to derive quantitative sediment guidelines for use as screening levels in conjunction with confirmatory bioaccumulation tests. The methods for bivalve bioaccumulation tests should follow the protocol currently being developed by Dr. H. Lee of U.S. EPA Office of Research and Development Laboratory at Newport, Oregon. Recommended exposure scenarios and preliminary guidelines for tissue concentrations of contaminants are presented in Appendix A. The recommended approach is summarized in Figure 6. In the short term, available information may permit the implementation of Option 4 for a few chemicals (e.g., PAH, PCBs, and DDT compounds). These are some of the most widespread contaminants of concern in Puget Sound relative to human health issues. Further development of risk assessment models is needed especially those relating concentrations of contaminants in tissues of aquatic organisms to sediment contamination. Therefore, implementation of human health guidelines is not recommended at present.

Because Option 4 does not address concerns about chemical mixtures and unanalyzed chemicals, it is recommended that short-term bioassays (Option 2) be developed further for application to sediment testing. Studies of the Ames test and other bioassays for mutagenesis and carcinogenesis of sediments are presently being conducted by the University of Washington and the Waterways Experiment Station of the Corps. When the results of these studies are available, the use of short-term bioassays for sediment quality testing should be evaluated further. A test for mutagenicity is presently being developed based on *P. phosphoreum*, the bacterium used in the Microtox bioassay. Although this test is being developed for water and effluent samples, it could potentially be extended to sediment or tissue extracts. The results of developmental studies on the Microtox mutagenicity bioassay, including a detailed intercomparison with the Ames test should be available in approximately six months (Razza, J., 15 November 1988, personal communication). At that time, the Microtox mutagenicity bioassay should be evaluated for application to sediment quality standards.

5.3.2. Relationship of Human Health Guidelines to Other Sediment Criteria

Options for incorporation of human health guidelines into the proposed sediment quality standards are outlined below:

- **Option 1: Human health guidelines as primary criteria**—Under Option 1, the human health risk assessment would be performed before the ecological hazard evaluation. The ecological guidelines would be applied only to sediments that fail the human health risk evaluation (i.e., sediments with potentially adverse effects on human health). This approach assumes that unacceptable ecological effects would not occur unless human health guidelines are triggered. The results of the ecological evaluation would be of secondary importance and would be used mainly as additional information to support prioritization of sites.

- **Option 2: Human health guidelines as secondary criteria when other criteria are "flagged"**—Under Option 2, the human health guidelines would be evaluated only when a sediment failed the ecological guidelines. Thus, this option places human health risk assessment in a secondary role and assumes that unacceptable health risks would occur only if ecological guidelines were triggered. The human health risk information would play a supportive role for site ranking similar to that of the ecological guidelines under Option 1.

- **Option 3: Human health guidelines as secondary criteria when other criteria are not "flagged"**—Under Option 3, a sediment would be evaluated relative to the human health guidelines only when it passes the ecological evaluation. Unacceptable ecological hazard would be sufficient to classify sediments of concern on the inventory. The human health risk assessment would serve as a "double-check" to ensure that sediments without significant ecological effects, but with potentially adverse effects on humans, are considered in the inventory evaluation process.

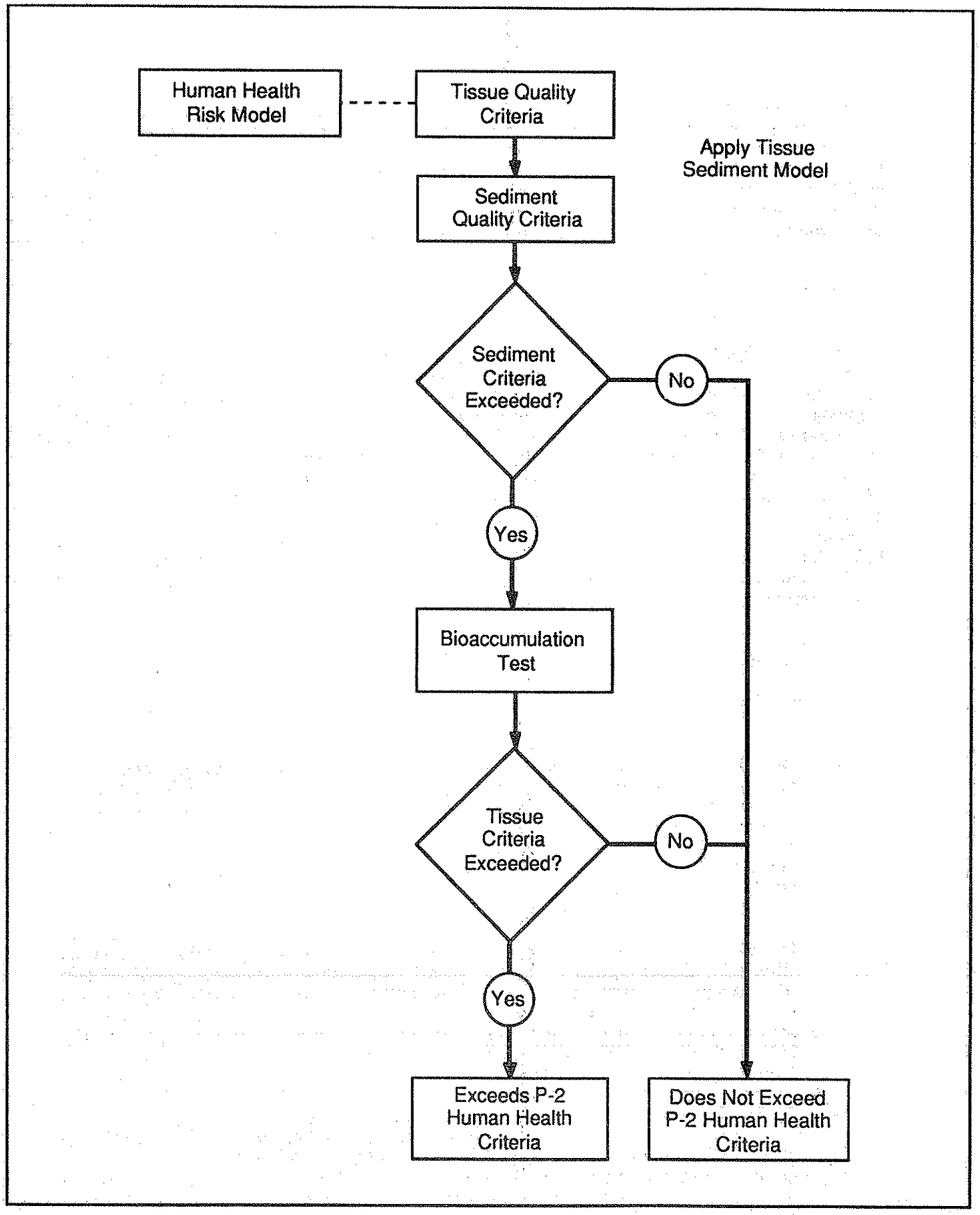


Figure 6. Recommended approach to human health assessment

- **Option 4: Human health guidelines as complementary to other criteria**—Under Option 4, all sediments would be evaluated relative to both the ecological and the human health guidelines. The results of both evaluations would be used to rank sites on the inventory.

The relationship of human health assessment to assessment of biological resources is shown in Figure 7 for each of the options.

Option 4 is recommended for incorporation of human health criteria into the proposed sediment quality standards. Option 4 would consistently provide the most information for classifying sites in Puget Sound. At the same time, Option 4 recognizes the uniqueness of both potential effects on biological resources and human health. Options 1-3 are limited by lack of consideration of potential differences between humans and aquatic organisms in their responses to chemical contamination, in their mechanisms of chemical uptake, and in their sensitivity to specific chemicals. Option 3 requires that some sediments be classified based on the results of the biological resources evaluation alone. Under Option 3, limited information would be available for classifying sites without the results of a human health risk assessment.

5.4. CONCLUSIONS AND SUMMARY OF RECOMMENDATIONS

The following conclusions are based on the evaluation of approaches to develop human health guidelines as part of the proposed sediment quality standards:

- Risk assessment approaches developed by EPA (e.g., U.S. EPA 1986a,b,c,d,e,f; Pastorok 1988) are potentially useful for developing human health guidelines for contaminated sediments

- The most important pathway for exposure of humans to sediment-associated contaminants is through the aquatic food chain

- Models to predict the concentrations of contaminants in tissues of fish and shellfish based on corresponding concentrations in sediments require further development

- Assessment of human health risks associated with complex mixtures of contaminants is essential; however, modeling approaches related to this problem are in an early state of development

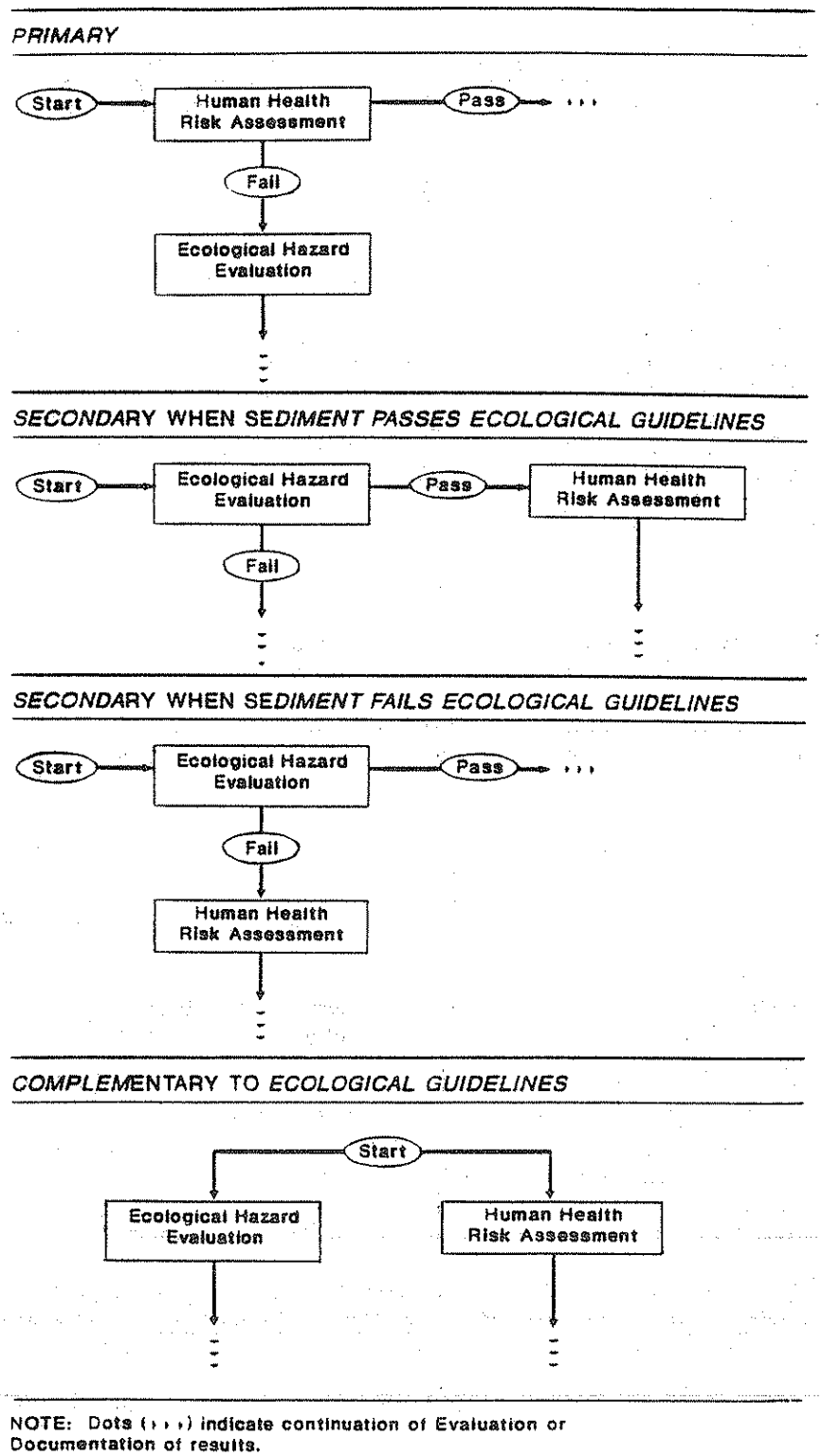
- Data from short-term genotoxicity tests (e.g., Ames test) of sediments or of tissues of marine organisms exposed to sediments can not be used directly to derive dose-response relationships to predict human cancer risk; however, such tests could supplement risk modeling efforts by assessing relative genotoxicity of complex mixtures of chemicals, including unidentified compounds

- Data on fish liver neoplasms are useful as an indicator of potentially carcinogenic substances in the environment, but can not be used to estimate human health risks or to develop dose-response relationships for contaminated sediments.

The recommended approach to human health guidelines for evaluation of sediment contamination is summarized below:

- Quantitative screening values for concentrations of key chemicals (e.g., PAH, PCBs, and DDT compounds) in sediments should be derived by extrapolation from tissue quality guidelines corresponding to tolerable risk levels

- Available risk assessment models should be used to derive the tissue quality guidelines (e.g., Appendix A) based on hypothetical (yet realistic) exposure scenarios that do not require actual exposure to be demonstrated at each site to be evaluated



NOTE: Dots (. . .) indicate continuation of Evaluation or Documentation of results.

FIGURE 7. Options for incorporating human health guidelines into sediment quality standards.

- When screening values are exceeded, a bioaccumulation test using clams (*Macoma* spp.) should be conducted to confirm that exposure of marine organisms could result in exceedance of the tissue quality guidelines

- Uncertainty associated with tissue-sediment extrapolations based on equilibrium partitioning should be assessed before application of the recommended approach

- Approaches to equilibrium partitioning models for metals (including organic forms of metals) should be evaluated further

- Tissue-sediment extrapolation models should eventually be field-validated

- The bioaccumulation potential of key contaminants should be determined in *Macoma* spp. relative to harvested fish and shellfish

- Short-term genotoxicity tests should be further developed to supplement risk assessment modeling approaches

- A state registry should be established for key toxicological data on chemicals of potential concern in human health risk assessment.

6. SUMMARY OF RECOMMENDED SEDIMENT REGULATORY LEVELS

This section summarizes the major elements of the recommended approaches for 1) classifying Puget Sound sediments according to their potential for having adverse chronic and acute biological effects and 2) developing human health criteria with respect to contaminated sediments. The assessment schemes for biological resources and human health are summarized in Figure 8. The details of these recommended approaches, including optional modifications or alternative approaches, have been described earlier in Sections 4 and 5. It is recommended that the two approaches be conducted independently, so that the results from one will not influence the results from the other. The independence of the two approaches is a reflection of their uniqueness.

6.1. BIOLOGICAL TESTING ASSESSMENT SCHEME

To develop a classification scheme for identifying and designating sediments having observable acute or chronic adverse biological effects, it is recommended that biological testing be integrated with chemical screening criteria in the manner presented in Figure 8. The key elements of the classification scheme are:

- Chemical screening criteria to identify sediments having a very low potential for causing adverse effects (i.e., NEC)

- Biological testing using acute and chronic sediment bioassays and effects on indigenous organisms to confirm that sediments with chemical concentrations above NEC have acute and/or chronic effects, and to classify the sediments in accordance with the observed biological results.

It is recommended that chemical screening criteria (i.e., NEC) be based on Puget Sound AET, which have been tested in this region. Because there is little information regarding chronic effects in Puget Sound, it is recommended that NEC initially be developed using the Puget Sound AET based on benthic macroinvertebrate assemblages (i.e., an indicator that includes both chronic and acute effects on indigenous organisms), and an ACR of 10 relative to the acute bioassays used to develop Puget Sound AET (i.e., amphipod mortality, oyster larvae abnormality, and Microtox tests). The 90th percentile of chemical concentrations from Puget Sound reference areas should set the lower limit for each NEC.

The proposed classification scheme will classify sediments in Puget Sound as having or not having adverse effects and identify those sediments with adverse effects according to whether the effects are chronic or acute. In doing so, the classification scheme will facilitate sediment management activities in the Sound. The use of chemical screening criteria as an integral part of the scheme will focus biological testing on the sediments having the greatest potential for causing adverse effects.

6.2. HUMAN HEALTH ASSESSMENT SCHEME

The recommended approach for human health assessment of a combination of:

- **Qualitative Hazard Assessment**—Evaluation of the potential toxicity of chemicals measured in sediments based on available data and literature reviews of health effects

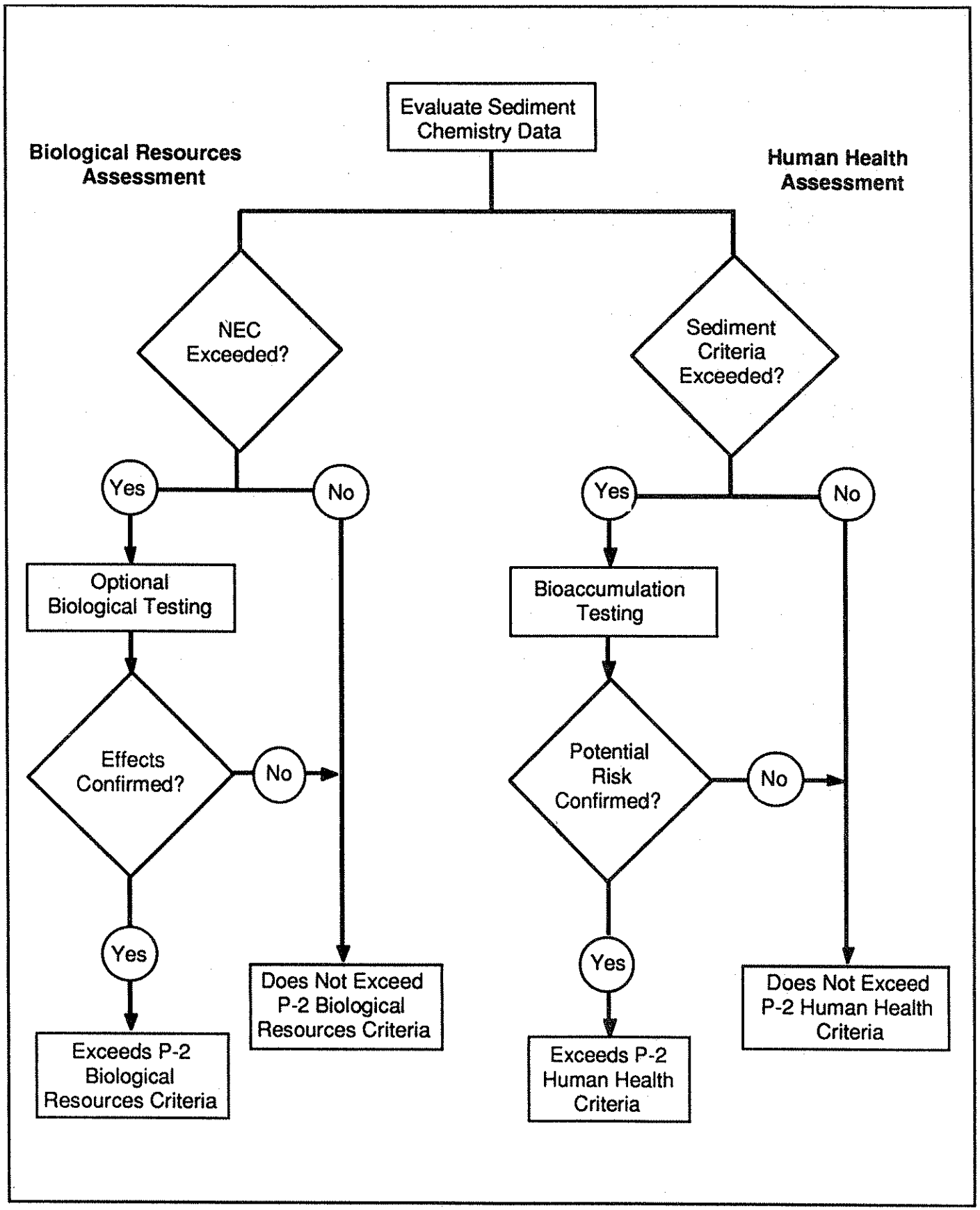


Figure 8. Integrated approach to biological resources and human health assessments

- **Quantitative Risk Assessment**—Modeling of potential human health risks resulting from consumption of fish or shellfish associated with contaminated sediments to derive sediment standards

- **Sediment-Specific Hazard Assessment**—Short-term bioassays (e.g., Ames test) of sediments or, preferably, tissues of organisms exposed to specific sediments.

Implementation of human health guidelines as part of the sediment quality standards (see Figure 6) requires further development of available hazard and risk assessment techniques. Thus, implementation of human health guidelines is not recommended at present. The developmental status and needs for each element of the recommended approach are summarized below.

Qualitative hazard assessment should constitute the first stage for any risk assessment process. Available information from EPA, IRIS, the National Toxicology Program, and the International Agency for Research on Cancer will be adequate to evaluate qualitatively the potential toxic effects of chemicals on exposed humans. Procedures for qualitative hazard assessment related to contaminated aquatic environments are described by U.S. EPA (1986f) and Pastorok (1988). A state registry (either electronic or paper) of toxicity profiles should be established for Puget Sound chemicals of concern as defined under the proposed sediment quality standards. Each toxicity profile would briefly describe the potential effects of a specific chemical on humans through sediment or food-chain exposures. Key variables for risk assessment (e.g., carcinogenic potency factors or RfD) would be included in the profile to provide support documentation for tissue quality guidelines or sediment standards related to human health concerns (e.g., Appendix A). For many chemicals, the chemical file from EPA IRIS will probably be sufficient to serve as the toxicity profile for the state registry. The registry should be updated at least quarterly.

A quantitative risk assessment model based on food-chain bioaccumulation of contaminants is recommended to address the question of how much risk may be associated with contaminated sediments. For example, by assuming that humans consume selected indicator species of seafood from Puget Sound at specified rates, the EPA risk assessment approach can be used to develop guidelines for chemical concentrations in edible tissues of fish or shellfish (e.g., Appendix A). Using a model of interactions between sediments and fish/shellfish, the tissue quality guidelines would then be extrapolated to develop sediment quality guidelines (i.e., screening values). These guidelines would represent the concentrations corresponding to tolerable levels of risk (or exposure) selected by regulatory management policy. In the near term, available data and equilibrium partitioning models may be used to develop very conservative (i.e., protective) screening values for concentrations of contaminants in sediments. When the sediment guidelines are exceeded, a potential for adverse human health effects should be confirmed by using a laboratory test of bioaccumulation of contaminants in clams. Protocols for bioaccumulation tests using sediments are being developed by EPA. The concentrations of contaminants in clam tissue would be compared directly with the tissue quality guidelines developed earlier. Exceedance of the tissue quality guidelines would lead to classification of the sediment as potentially having adverse human health effects.

Further refinement of quantitative approaches for setting human health guidelines is recommended. For example, models for relating sediment contamination to fish or shellfish contamination need to be developed further (Appendix A). The preferred approach involves development of empirical relationships between chemical concentrations in sediments and in fish/shellfish tissue. Extensive sampling and analysis may be required to obtain sufficiently precise relationships for a variety of chemicals in different sediment matrices (e.g., sediments differing in TOC concentration and grain size). Long-term development of equilibrium partitioning models to predict tissue concentrations of contaminants from corresponding concentrations in sediments is possible, but laboratory and field validation of models is needed for at least key organic contaminants (e.g., PAH, PCBs, DDT and related metabolites).

Finally, short-term bioassays (e.g., Ames test) of tissue extracts are recommended to evaluate the mutagenic and carcinogenic potential of chemical mixtures and single chemicals that are not analyzed for. These short-term tests complement the quantitative risk assessment modeling

approach. Methods are available for analysis of tissue extracts using the Ames test (e.g., Sparks et al. 1981; Pittinger et al. 1987), but problems in measuring the mutagenic potential of complex mixtures of contaminants require further research. Calibration of short-term bioassays to various concentrations of known mutagens in tissue samples is also needed. Consequently, the Ames test or another short-term bioassay will likely require substantial development before possibly being applied to the sediment quality standards. When data on the performance of the forthcoming Microtox mutagenicity bioassay become available, this mutagenicity test should be evaluated for use as part of the sediment quality standards.

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APPENDIX A

Issues and Approaches in Development of Sediment Criteria Based on Human Health Risk Assessment

INTRODUCTION

Options for incorporating guidelines for protection of human health into the proposed Element P-2 sediment standards are discussed in the main text of this report. In this appendix, some key issues in human health risk assessment and possible approaches to risk assessment of contaminated sediments are evaluated. The objectives of this appendix are listed below:

- Review available approaches to risk assessment that could be applied to regulation of contaminated sediments
- Evaluate human exposure pathways/routes
- Select a recommended exposure scenario(s), dose-response model, and model(s) for extrapolating sediment contamination to predict contaminant concentrations in fish/shellfish
- Compile toxicity data for chemicals of concern, including weight-of-evidence for health effects and dose-response indices or thresholds (i.e., Carcinogenic Potency Factors and RfDs)
- Develop guidelines on contaminant levels in edible tissues of aquatic organisms using the recommended risk assessment approach
- Evaluate the potential application of measurements of fish neoplasms to human health risk assessment for contaminated sediments.

Issues in risk assessment and available approaches are evaluated in the next section. Pathways and routes of exposure of humans to contaminants associated with sediments are evaluated next. A recommended approach to risk assessment incorporating the primary exposure pathway (i.e., transfer of contaminants from sediment to fish or shellfish to humans) is used with compiled toxicity data to develop interim human health guidelines. Finally, the role of data on fish liver pathology in assessing human health risks is evaluated.

EVALUATION OF RISK ASSESSMENT APPROACHES

Available approaches to chemical risk assessment are reviewed in this section. The review focuses on differences in the methods used by major regulatory agencies (e.g., EPA, FDA) and selected academic institutions in each stage of a risk assessment. Because scientific knowledge of the effects of toxic chemicals on humans is still rudimentary, many assumptions and uncertainties are inherent in evaluation of human health risk. Key issues in risk assessment of contaminated sediments are defined in the final subsection below.

ELEMENTS OF RISK ASSESSMENT

Risk assessment is a scientific procedure used to estimate the probability of adverse health effects that may result from exposure to a toxic agent. Assessing risks of human exposure to toxic chemicals consists of the following steps:

- **Hazard Identification** - Qualitative evaluation of the potential for a substance to cause adverse health effects (e.g., birth defects, cancer) in animals or in humans

- **Dose-response assessment** - Quantitative estimation of the relationship between the dose of a substance and the probability of an adverse health effect

- **Exposure assessment** - Characterization of the populations exposed to the toxic chemicals of concern; the environmental transport and fate pathways; and the magnitude, frequency, and duration of exposure

- **Risk characterization** - Estimation of risk for the health effect of concern based on information from the first three steps.

The concepts of exposure and dose, as defined below, are central to risk assessment:

- **Exposure** - Contact by an organism with a chemical or physical agent

- **Dose** - The amount of chemical uptake by an organism over a specified time as a consequence of exposure.

Note that these concepts are common to various risk assessment approaches. Potential approaches to risk assessment of Puget Sound sediments differ mainly in assumptions made in extrapolating sediment concentrations of contaminants to fish and in extrapolating health effects measured in animal bioassays to humans.

Some available approaches to risk assessment emphasize the early stages of the analysis (i.e., hazard assessment), without estimating quantitative estimates of risk. Such qualitative approaches are especially appropriate in the absence of adequate dose-response information for the effects of a specific chemical on humans or laboratory animals. When adequate dose-response data are available, a quantitative estimate of risk or an index of the dose-response curve may be derived. Some possible forms of

the results of a quantitative risk assessment are addressed in the Estimation of Risk section below.

Available approaches generally require some data on concentrations of contaminants in the environmental media of concern (e.g., water, sediment, tissue). The kind and amount of data required for risk assessment is a key element to be considered in the evaluation because of cost implications of data collection. The tradeoff between reduced uncertainty achieved by collecting site-specific data and cost of data collection influences the choice of a risk assessment approach.

HAZARD ASSESSMENT

The EPA approach to hazard assessment is a comprehensive procedure for evaluating biological and chemical information to determine the weight of evidence for causation of specific adverse health effects (e.g., U.S. EPA 1986a,c). The EPA approach considers the following kinds of information:

- Chemical properties
- Structure-activity relationships
- Metabolic/pharmacokinetic properties
- Toxicologic effects
- Short-term tests for mutagenicity/carcinogenicity
- Long-term animal studies
- Human studies
- Weight-of-evidence classification.

The results of the hazard assessment influence the nature and extent of subsequent steps in risk analysis. For example, the endpoint of concern to be used in dose-response assessment may be selected based on the most severe adverse effect identified in the hazard assessment. In the absence of quantitative data for other steps in the risk assessment process, the hazard assessment constitutes the final product for a qualitative evaluation of risk.

The major federal regulatory agencies [EPA, FDA, Occupational Safety and Health Administration (OSHA)] involved in risk assessment follow essentially similar approaches to evaluate hazards of chemicals. The primary issues in hazard assessment of contaminated sediments by Ecology concern access to and use of available toxicological information, development of or use of standard sediment tests (e.g., Ames test) to evaluate human health hazards, treatment of chemical mixtures, selection of critical endpoints, and development or adoption of classification schemes for chemicals (e.g., weight of evidence classification for carcinogenicity).

It is anticipated that Ecology will rely largely on existing databases and available toxicity profiles for hazard assessments based on interpretation of sediment chemistry

data. The EPA IRIS represents an excellent, easily accessible source of summary toxicological data used to support development of indices used in risk assessment (RfD and Carcinogenic Potency Factors). Other sources of toxicological data are discussed in a later section (see Development of Interim Guidelines for Allowable Risk, Compilation of Toxicity Data).

The most valuable information for hazard assessment of single chemicals in sediments will come from available databases on effects observed in human studies and in long-term animal bioassays (usually on rats or mice). However, site-specific data and direct characterization of dose-response for chemical mixtures in sediments could be obtained from short-term laboratory bioassays. Short-term tests are useful for qualitative prediction of carcinogenicity in long-term whole organism studies, but cannot be used to estimate potency of carcinogens in long-term tests or in humans (Purchase 1985; U.S. EPA 1985b). The results of short-term tests may have to be interpreted in light of responses to reference sediments. The results of the Ames test, for example, could be expressed as the number of mutations per gram dry weight of sediment (extract equivalent). The responses of short-term genotoxicity bioassays (including the Ames test and anaphase aberration test) to reference sediments are not well known. Also, it is noteworthy that none of the federal agency policies reviewed by the U.S. Office of Technology Assessment (OTA) (1987) supported use of short-term tests as the sole basis for identifying or regulating carcinogens.

Available tests for assessing genotoxicity of chemicals can be classified into four categories:

- Gene mutation
- Chromosome effects
- DNA damage
- Neoplastic transformation.

Within each category, there are several assays being used conventionally [U.S. Office of Science and Technology Policy (U.S. OSTP) 1985], although only two have been applied to contaminated sediments: the Ames mutagenicity test (Ames et al. 1973, 1982) and the anaphase aberration test for chromosomal abnormalities in trout cells (Kocan et al. 1982, 1985; Landolt and Kocan 1984). Assays of chromosomal abnormalities in echinoderms are under development (Hose 1985). Vouk et al. (1985) concluded that among the approximately 100 short-term tests for carcinogenicity, only the Ames test had been fully validated as an established test. Based on their present use as screening tests (U.S. OSTP 1985; Vouk et al. 1985; U.S. OTA 1987; Tennant et al. 1987), other promising tests for routine screening of carcinogenicity of sediments include induction of unscheduled DNA synthesis in mammalian cells and neoplastic transformation of mammalian cells. Few attempts have been made to apply short-term tests to extracts of contaminated tissues of aquatic organisms. Nevertheless, further development of this approach may be warranted because of the importance of bioaccumulation of contaminants in food chains as a potential pathway for exposure of humans to contaminants in sediments (see the Evaluation of Exposure Pathways/Routes section below).

A common approach to carcinogenicity testing is to use a suite of tests to avoid limitations inherent in any one test and to evaluate the weight of evidence for specific

chemical effects (U.S. EPA 1985b). However, the use of several tests in addition to the Ames test may not increase the ability to predict whether a chemical is carcinogenic in long-term animal bioassays (Tennant et al. 1987; U.S. OTA 1987).

Because of the importance of chemical mixtures in contaminated areas of Puget Sound, the assessment of multiple chemical effects is critical. Approaches to treatment of chemical mixtures in hazard and risk assessment are in a state of early development. Where complex mixtures of chemicals arise from a particular source (e.g., PAH mixtures from a creosote facility), information on the toxicity of the mixture itself may be available. More commonly, sediments contain various contaminants that originate from multiple sources. In this case, prediction of the mixture's toxicity based on knowledge of individual chemicals is difficult and highly uncertain. Most available hazard assessments used by regulatory agencies are based on single chemical tests. Available methods for hazard assessment of chemical mixtures rely largely on qualitative evaluation of multiple effects based on consideration of potential additive effects of individual chemicals. Despite the recognition of antagonistic and synergistic effects, conceptual approaches and practical regulatory tools for assessing hazards of chemical mixtures are severely limited. Consequently, use of bioassays for direct assessment of contaminated sediments has clear advantages over purely predictive techniques. Use of short-term tests to evaluate potential human health hazards from contaminated sediments would also be consistent with recommendations of U.S. EPA (1985b) that such tests be used in evaluating effluent discharges.

The endpoints of concern in short-term tests for evaluation of human health hazards are usually mutagenicity or carcinogenicity. Relatively inexpensive short-term tests for chronic systemic effects are unavailable (U.S. EPA 1985b). In predictive assessments or hazard evaluations based on literature data for long-term tests, the critical endpoints are diverse. However, the selection of endpoints for further evaluation and risk estimation is often focused on carcinogenicity or teratogenicity, although other developmental effects (e.g., U.S. EPA 1986c) are evaluated. Because of the limited dose-response data for promoters, cocarcinogens, and procarcinogens, regulatory criteria for chemicals with these properties are usually based on noncarcinogenic effects or treatment of the chemical as a complete carcinogen in risk modeling (Stara et al. 1983).

Several schemes have been developed to classify chemicals based on the weight of evidence for carcinogenicity. Available classification schemes include the International Agency for Research on Cancer (IARC) classification, the EPA classification, and the National Toxicology Program (NTP) classification. These agencies, individual states (e.g., California Safe Drinking Water Act of 1986), and individual researchers (e.g., Gold et al. 1984) have developed lists of carcinogens, and in some cases, lists of reproductive toxicants.

DOSE-RESPONSE ASSESSMENT

Current approaches to risk assessment differ greatly in their treatment of dose-response assessment, especially in the choice of model for extrapolation of animal bioassay data at high doses to low-dose exposures and in their treatment of uncertainty. Some of the issues in dose-response assessment include:

- The presence or absence of thresholds for carcinogenesis

- Choice of data from the most sensitive species vs. a variety of species

- Choice of low-dose extrapolation model [e.g., use of pharmacokinetic (mechanistic) models vs. tolerance distribution models]

- Use of relative body weight vs. relative surface area for dose conversion from animal bioassay results to humans

- Use of maximum likelihood estimates vs. upper confidence limits for dose-response curves and estimates of carcinogenetic potency

- Use of alternatives to dose-response modeling [e.g., relative potency indices such as ED_{50} (effective dose to 50 percent of test animals)].

It is anticipated that Ecology will not perform completely new dose-response assessments, but will rely on the results of previous assessments by other agencies (e.g., use of existing EPA or FDA Carcinogenic Potency Factors). In general, EPA, FDA, and other federal agencies use tolerance distribution models that assume a no-threshold, linear response for carcinogenicity at low doses characteristic of environmental exposures. Although some agencies have advocated the use of several models to generate a range of risk estimates, the FDA has stated that such an approach is unlikely to provide useful information. The use of a single model consistently, unless evidence supports use of an alternative model, offers advantages in its simplicity and value for relative risk comparisons. The EPA, FDA, and other federal agencies generally rely on upper confidence limits for estimating carcinogenetic potency, whereas some academic risk analysts have used maximum likelihood estimates with associated estimates of uncertainty. For most existing data sets from animal bioassays, the limited number of dose treatments leads to instability in maximum likelihood estimates. Consequently, use of estimates of upper confidence limits for carcinogenic potency is the only practical approach for many chemicals given present models. In final stages of the risk assessment, this approach results in a plausible upper-limit estimate of risk.

The most valuable database for summary information on dose-response parameters is the EPA IRIS database. An alternative approach would be to contact FDA directly for toxicological indices, although their Carcinogenic Potency Factors and RfDs are not easily accessible. The main difference between EPA and FDA risk assessment procedures concerns the methods for extrapolating the toxic potency of chemicals to small experimental animals (e.g., rats and mice) to estimate potential effects in humans. EPA's use of body surface area as a scaling factor for chemical dosages results in estimates of risk five to ten times higher than corresponding estimates derived by FDA using body weight as the scaling factor. The conversion of EPA's risk assessment parameters (e.g., Carcinogenic Potency Factors) to be consistent with FDA's approach to interspecies extrapolation using body weight scaling of dosages is described by Pastorok (1988).

The main alternative to dose-response extrapolation is the use of toxic potency indices that characterize a single point on the dose-response curve (e.g., the ED_{50}). This approach has been endorsed recently by Clayson et al. (1983) and Ames et al. (1987). Although such indices may be useful for limited screening analyses, they may offer little insight into low-dose risk (U.S. OSTP 1985). Moreover, they ignore relative differences in slope of the dose-response curve among chemicals.

EXPOSURE ASSESSMENT

Available strategies for exposure assessment are generally similar to one another. The primary differences among exposure assessments concern the pathways/routes considered, the specific values selected for uptake constants (e.g., human seafood consumption rate, gastrointestinal absorption coefficients), and treatment of uncertainty (e.g., use of worst-case analysis). The stages of exposure assessment include:

- Characterization of exposure pathways/routes
- Estimation of chemical concentrations
- Characterization of exposed populations
- Estimation of chemical intake (long-term average).

Approaches to exposure assessment often need to be site-specific.

The two broad categories of techniques applied to exposure assessment are modeling (especially transport and fate analysis) and direct biological monitoring (e.g., chemical analysis of human blood or tissue). It is anticipated that Ecology will rely on hypothetical exposure scenarios and estimation of exposure based on modeling. Available models for predicting transport and fate in aquatic environments were reviewed by Onishi (1985a,b). U.S. EPA (1986b) advocates development of realistic exposure scenarios, rather than the use of worst-case assessments. An evaluation of pathways and routes of exposure of humans to contaminants associated with sediments and a recommended exposure scenario are addressed in separate sections below.

In developing sediment criteria, the main issue concerning exposure assessment is whether selected exposure scenarios should be applied to all habitats regardless of the present condition of the habitat, its ability to support certain aquatic species, or actual site-specific potential for human exposure. For example, should sediment criteria derived from a hypothetical exposure scenario based on shellfish consumption be applied to all intertidal habitats even though some locations may not support harvestable shellfish? Should such criteria be applied to marginal or inaccessible shellfish habitats? The use of hypothetical exposure scenarios based on laboratory results is also questionable (e.g., application of risk assessment methods to evaluate contaminant levels in tissues of clams exposed to sediments in the laboratory).

ESTIMATION OF RISK

A major issue in use of risk assessment to evaluate contaminated sediments is whether the risk characterization should be quantitative. Quantitative carcinogenic risk assessment is used widely by regulatory agencies, although most recognize its limitations. Some academic researchers have recommended against using extrapolations of data on responses of laboratory animals in bioassays of chemicals to obtain quantitative estimates of human cancer risk (e.g., Ames et al. 1987). In the present context, quantitative estimates of human cancer risk from chemical exposure would be expressed as the probability that each exposed individual will experience a given adverse effect within his or her lifetime (usually assumed to be 70 years). In estimating risks, we are concerned with excess lifetime risk due only to contamination of sediments [and related

pathways (e.g., seafood)], not the total risk from all exposures to toxic chemicals (that is, through cigarette smoking, drinking water, diet other than seafood). Nevertheless, it is important to compare the potential risks from contaminated sediments with those from other exposures. Risk comparisons are valuable for providing perspective on tolerable risks. Also, the size of the potentially exposed population has generally been considered by EPA and other agencies in establishing environmental regulations (Travis et al. 1987).

Options for expressing carcinogenic risk estimates include the following:

- Mean risk
- Upper-limit risk
- ED_{50} or analogous index.

These approaches relate to the choice of measures of carcinogenic potency derived in the dose-response assessment as discussed above (see Dose-Response Assessment section). The EPA, FDA and most other regulatory agencies use the upper-limit estimate of carcinogenic risk.

Noncarcinogenic risk is generally determined by comparison of the estimated average lifetime exposure with a NOEL or a related guideline (e.g., EPA RfD). The main disadvantages of this approach are the lack of consideration of the slope of the dose-response curve and the use of somewhat arbitrary uncertainty factors (Dourson and Stara 1983).

Analogous to the treatment of chemical mixtures in hazard assessment, the typical approach to estimating risk of chemical mixtures is to calculate the sum of estimated risks for individual chemicals in the mixture. Although this approach has severe limitations, it is the only practical one at present (U.S. EPA 1986d).

For both carcinogens and noncarcinogens, development of sediment criteria would require back-calculation from a selected tolerable risk level (for carcinogens) or RfD (for noncarcinogens) to derive a target concentration in the media of concern (e.g., edible tissues of fish or shellfish). The target concentrations in tissue would then be related to a sediment concentration to obtain sediment criteria. Approaches to derive criteria for chemical mixtures are not available. Moreover, to justify use of the RfD for noncarcinogens it may be necessary to consider exposures through pathways other than those related to contaminated sediments (e.g., diet other than marine/estuarine fish and shellfish, drinking water, and soil ingestion by children).

SEDIMENT TO FISH EXTRAPOLATION

Derivation of quantitative guidelines for concentrations of chemicals in sediments based on human health risk models will likely involve generic rather than site-specific guidelines. Thus, a hypothetical exposure scenario would be defined and sediment quality guidelines would be derived by back-calculation from a tolerable risk level selected by regulatory policy. As discussed in later sections, bioaccumulation of contaminants in seafood species is the most important exposure pathway for potential effects of contaminated sediments on humans. Development of tissue (seafood) quality guidelines is therefore an intermediate step in the derivation of quantitative sediment quality values

(e.g., see below, Development of Interim Guidelines for Allowable Risk section). Another key step is extrapolation of tissue quality guidelines to estimate allowable concentrations of chemicals in sediments. Available models for relating contaminant concentrations in sediments to those in aquatic organisms include:

- Equilibrium partitioning
- Apparent effects threshold
- Regression models.

In the equilibrium partitioning approach, the sediment criteria value for a specific contaminant would be calculated from a selected human health guideline for edible tissue contamination in fish/shellfish (e.g., FDA action level or a guideline derived from EPA risk assessment models) by assuming thermodynamic equilibrium. A three-stage prediction is needed: 1) from chemical concentration in bulk sediment to concentration in interstitial water, 2) from chemical concentration in interstitial water to overlying water column, and 3) from water to fish or shellfish using bioconcentration factors. Using the AET approach, sediment criteria values would be established empirically by determining the sediment concentration of contaminant above which statistically significant ($P<0.05$) elevations of contaminant concentrations in tissue relative to a guideline are expected. The guideline for tissue contamination may be derived through the use of a risk assessment model (see below, Development of Interim Guidelines for Allowable Risk section). Both the equilibrium partitioning and AET approaches were evaluated by Beller et al. (1986) and are summarized in Section 2 of the main report. A third option is to use regression models to relate tissue concentrations of contaminants to sediment concentrations (e.g., Connor 1984). The sediment concentration of contaminant corresponding to a selected tissue contamination guideline (e.g., interim tissue quality guideline) would be designated as the sediment criterion. Advantages and disadvantages of these approaches are outlined below.

Equilibrium Partitioning

Equilibrium partitioning has the following advantages and disadvantages:

- **Advantages**
  - Well-developed theoretical basis
  - Uses available toxicological database (EPA WQC)
  - Some supportive empirical data for some chemicals and some organisms
  - Applies to a variety of sediment types (assuming equilibrium)
- **Disadvantages**
  - Limited to nonpolar, nonionic organic compounds
  - Assumes equilibrium among sediment, interstitial water, overlying water, and organisms

- Ignores chemical interactions that may influence partitioning
- For fish, assumes that individuals are exposed to sediments within a known, specified area.

Apparent Effect Thresholds

Apparent Effects Thresholds have the following advantages and disadvantages:

- **Advantages**
  - Can be applied to a wide variety of chemicals
  - Assumption of equilibrium is not required
  - Implicitly accounts for chemical interactions that may affect bioaccumulation
- **Disadvantages**
  - For fish, assumes that individuals are exposed to sediments within a known, specified area
  - No database available for shellfish
  - Limited database available for fish.

Regression Models

Regression models have the following advantages and disadvantages:

- **Advantages**
  - Empirical field-based approach
  - Can be applied to a wide variety of chemicals
- **Disadvantages**
  - High variability in fish-sediment regression relationships for some chemicals
  - Limited database available for fish and shellfish contamination in relation to sediment contamination
  - For fish, assumes that individuals are exposed to sediments within a known, specified area.

The equilibrium partitioning approach needs to be evaluated further to determine specific chemicals and classes of chemicals that can be modeled accurately. Some basic calculations could be done for available sediment chemistry data for Puget Sound and elsewhere to test the predictions of the equilibrium partitioning model. The results of ongoing work by the EPA Marine Science Center and be the EPA Criteria and Standards Program will be useful to evaluate equilibrium partitioning further. At present, the use of regression models is the most practical and cost-effective approach for the major contaminants of concern that cannot be addressed using the equilibrium partitioning approach.

SUMMARY OF KEY ISSUES

The most important issues in risk assessment of contaminated sediments are listed below:

1. Use of sensitive species data and protective assumptions for health risk assessment (i.e., combined effects of several worst-case assumptions on results)

2. Choice of model for low-dose extrapolation

3. Measurement vs. modeling of fish/shellfish contamination

4. Choice of model for sediment-fish extrapolation

5. Expression of risk as mean or upper-limit

6. Hypothetical characteristics of exposed population

7. Exposure pathways/routes considered

8. Use of hypothetical exposures based on contaminant bioaccumulation by clams in laboratory experiments.

EVALUATION OF EXPOSURE PATHWAYS/ROUTES

This section addresses how chemical contaminants move from sediments to exposed humans. Development of exposure scenarios involves the identification of important transport and fate processes. Potential exposure pathways and chemical uptake routes in humans include:

- **Water contact**
  - Dermal (skin) absorption of contaminants in seawater
  - Ingestion of seawater containing contaminants
  - Inhalation of contaminants that volatilize from seawater
- **Sediment contact**
  - Dermal absorption of contaminants in sediments
  - Ingestion of sediments containing contaminants
- **Food chain exposure**
  - Ingestion of fish, shellfish, or algae contaminated by chemicals transferred from sediments to aquatic organisms directly, from sediments to water to organisms, or from sediments to intermediate trophic level organisms to humans.

Potential exposure scenarios were evaluated by considering the behavior of chemicals in the environment (e.g., persistence and possible transformations), the likelihood of high concentrations in various environmental media, and the potential for human exposure. In the evaluation, pathways that potentially result in higher concentrations of persistent toxic chemicals with a higher potential for human contact were assigned a higher rating of relative importance.

The primary exposure pathways/routes are uptake of contaminants by dermal absorption or ingestion (by children) as a result of direct contact with sediments and transfer of contaminants from sediments to fish/shellfish to humans. Past studies have clearly demonstrated potential health risks associated with consumption of aquatic organisms from chemically contaminated water bodies (Humphrey 1983, 1987, 1988; Jacobson et al. 1985; Rogan et al. 1986). Contact with contaminated seawater or sediments may occur during swimming, diving, or wading or playing in intertidal areas. Exposure routes for "barefoot" fishing in the surf zone may be considered similar to those for wading. Dermal exposure is considered as a potential route for uptake of contaminants because body contact with household water contaminated by volatile organic compounds has been shown to lead to significant exposure (Brown et al. 1984). Inadvertent ingestion of small quantities of seawater is likely to occur during swimming or diving. Dermal absorption and inhalation are potentially important primarily for

volatile organic compounds. Ingestion of sediments is unlikely to be significant except possibly in children that exhibit pica behavior.

Because of the relatively infrequent exposure or the relatively low rate of contact with contaminated sediments or water, consumption of locally caught seafood is considered more important than other exposure pathways listed above. Because swimming, wading and diving in Puget Sound are infrequent activities, human exposure through these activities can be considered negligible for most contaminants. In general, contaminant concentrations in seawater and intertidal sediments of Puget Sound are probably low relative to concentrations that might lead to substantial risks from direct contact with or ingestion of sediments. However, adequate chemical data are lacking for many intertidal and shallow subtidal areas near storm drains and sewage overflow points, which are the primary areas of potential concern. Exposures to dioxin [2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD)] through contact with sediments could potentially be of concern. This issue requires further evaluation because of the limited database on dioxin in intertidal sediments of Puget Sound.

Stuart et al. (1988) evaluated human health risks associated with lead, PAH, and phthalic acid esters discharged from storm drains and combined sewer overflows and with existing sediment contamination in Lake Union. The estimated risks associated with eating crayfish or bottomfish in contact with sediments were considered significant in some cases, whereas risks associated with ingestion of water while swimming were negligible. Stuart et al. (1988) did not evaluate human health risks associated with dermal or inhalation exposures through contact with water or sediments.

Whitmyre et al. (1987) evaluated the human health risks associated with recreational exposure to surface (fresh) waters near 12 Superfund sites. Dermal, ingestion, and inhalation exposure routes were considered relative to fishing and swimming activities. Twenty-seven pollutants were selected for their risk assessment based on their frequency of occurrence at Superfund sites in general, their toxic action (cancer-causing substances were emphasized), and the availability of data for performing a risk assessment. The selected chemicals included PCBs, DDT and several other EPA priority pollutant pesticides, 13 volatile organic compounds, naphthalene (a commonly occurring component of petroleum oils), and five metals. Health risks were estimated for a hypothetical "Maximum Exposed Individual" who was assumed to engage in both swimming and fishing. Although Whitmyre et al. (1987) concluded that recreational risks were "significant" at some sites, only two sites had associated risks as high as approximately one in ten thousand. Risks at six of the sites were less than one in one million, a level that is commonly considered tolerable for environmental exposures (Travis et al. 1987). The median cancer risk from drinking contaminated groundwater was approximately 100 times greater than the median recreational risk. Moreover, preliminary calculations by Whitmyre et al. (1987) showed that the risk from consumption of recreationally caught fish near these Superfund sites would be 10 to 10,000 times higher than risks due to other recreational exposures (i.e., dermal contact, ingestion of water, and inhalation).

Whitmyre et al. (1987) noted several specific limitations of their analysis, including the following:

- Site-specific data on the frequency and duration of recreational use were generally unavailable

- Dermal permeability estimates were available for only six of the 27 pollutants (values were assumed for the other 21 pollutants)

- The degree to which adsorption of contaminants to suspended solids reduces dermal uptake of contaminants is unknown

- Data on the amount of water typically ingested during swimming were not available.

Burmaster et al. (1987) estimated total cancer and total noncancer risks for 53 "critical contaminants" at a Superfund site in Massachusetts. Their worst-case scenario included evaluation of chemical exposures from contact with or ingestion of soils or aquatic sediments (children 5-18 years), fish consumption (70-year lifetime), drinking water (70-year lifetime), and swimming (children 5-18 years). The results of Burmaster et al. (1987) showed that estimated cancer risks associated with contaminated fish consumption were 10-100 times higher than those associated with other recreational exposures (i.e., contact with or ingestion of aquatic sediments and swimming by children).

DEVELOPMENT OF INTERIM GUIDELINES FOR ALLOWABLE RISK

The primary exposure pathway involving bioaccumulation of contaminants by fish or shellfish and transfer to humans that consume seafood provides a basis for calculating interim guidelines for allowable human health risk under the proposed Element P-2 sediment standards. In this section, recommended scenarios for exposure of humans to contaminated sediments via consumption of fish/shellfish are summarized, selected toxicological data are compiled, and interim guidelines for contaminant concentrations in edible tissues of aquatic organisms are developed. Options for incorporating human health guidelines into the proposed Element P-2 sediment standards are discussed in the main text of this report.

RECOMMENDED EXPOSURE SCENARIOS

Because of the potential importance of bioconcentration and biomagnification of contaminants in aquatic food chains with humans as the ultimate receptor, the role of these processes in human exposure scenarios is summarized below. Contaminants with a high potential for bioconcentration and biomagnification are also identified. Recommended exposure scenarios are then summarized.

Role of Bioconcentration and Biomagnification

Uptake of contaminants from sediments or overlying water by aquatic organisms has been demonstrated in many laboratory and field studies (e.g., Young et al. 1980; Williams and Pastorok 1985). In a process called bioconcentration, contaminants may concentrate in edible tissues of aquatic organisms at higher levels than their original concentrations in sediments or water. Biomagnification occurs when persistent contaminants with a high bioaccumulation potential (e.g., mercury, DDT, and PCBs) are passed up a food chain, resulting in higher concentration in tissues of organisms at higher trophic levels. Consequently, these processes can lead to excessive accumulation of contaminants, particularly mercury and fat-soluble organic compounds, in tissues of humans who consume fish or shellfish on a regular basis (Swartz et al. 1983; Rogan et al. 1986; Humphrey 1988). Consumption of predatory fish with high fat content may present the greatest risk to humans (Humphrey 1988).

Bioconcentration factors relating concentrations of contaminants in sediments to concentrations in tissues of marine/estuarine fish and shellfish are not available for most chemicals of concern. However, the bioconcentration factor relating contaminant concentrations in water to tissue levels indicates the relative capacity for bioaccumulation of a chemical. Inorganic chemicals are listed in Table A-1 in descending order of bioaccumulation potential, according to their bioconcentration factors (from Pastorok 1988 as adapted from Williams and Pastorok 1985). Bioconcentration factors for transfer of organic chemicals from water to tissue can be predicted by the octanol-water partition coefficient (Williams and Pastorok 1985). Organic chemicals and selected pesticides are listed in Table A-2 in descending order of bioaccumulation potential, according to their

TABLE A-1. PSDDA AND PSEP INORGANIC CHEMICALS OF CONCERN, RANKED ACCORDING TO BIOCONCENTRATION FACTOR (BCF)

Priority Pollutant No.	PSDDA	PSEP	Substance	Log BCF[a]
123	X	X	Methylmercury	4.602
123	X	X	Phenylmercury	4.602
123	X	X	Mercuric acetate	3.447
120	X	X	Copper	3.073
128	X	X	Zinc	2.762
115	X	X	Arsenic	2.544
118	X	X	Cadmium	2.513
122	X	X	Lead	2.253
119			Chromium VI	2.190
119			Chromium III	2.104
123	X	X	Mercury	2.000
124	X	X	Nickel	1.699
127			Thallium	1.176
114	X	X	Antimony	ND
117			Beryllium	ND
121	X	X	Cyanide	ND
125			Selenium	ND
126	X	X	Silver	ND

[a] BCF = Bioconcentration Factor. The value shown is the geometric mean BCF among studies summarized by Williams and Pastorok (1985). U.S. EPA (1986g) provides additional information on BCF values for selected chemicals.

ND = No data.

TABLE A-2. PSDDA AND PSEP ORGANIC CHEMICALS OF CONCERN RANKED ACCORDING TO OCTANOL-WATER PARTITION COEFFICIENTS (K_{ow}) (updated from Callahan et al. 1979)

Priority Pollutant No.	PSDAA	PSEP	Substance	$\log(K_{ow})$	Reference
69	X		di-n-octyl phthalate	8.06	m
83	X	X	indeno(1,2,3-c,d)pyrene	7.66	
89	X	X	aldrin	7.40	o
79	X	X	benzo(g,h,i)perylene	7.05	i
111	X	X	PCB-1260	6.91	d
--[q]			mirex	6.89	b
75			benzo(k)fluoranthene	6.85	
74			benzo(b)fluoranthene	6.60	
82	X	X	dibenzo(a,h)anthracene	6.50	k
107	X	X	PCB-1254	6.48	d
73	X	X	benzo(a)pyrene	6.42	i
91	X		chlordane	6.42	i
92	X	X	4,4'-DDT	6.36	n
90	X	X	dieldrin	6.20	o
110	X	X	PCB-1248	6.11	d
129		X	TCDD (dioxin)	6.10	i
94	X	X	4,4'-DDD	6.02	i
106	X	X	PCB-1242	6.00	a
72	X	X	benzo(a)anthracene	5.91	j
112	X	X	PCB-1016	5.88	d
76	X	X	chrysene	5.79	j
93	X	X	4,4'-DDE	5.69	h
99			endrin aldehyde	5.60	
53			hexachlorocyclopentadiene	5.51	d
9	X	X	hexachlorobenzene	5.47	l
100	X	X	heptachlor	5.44	d
101			heptachlor expoxide	5.40	d
39	X	X	fluoranthene	5.22	j
84	X	X	pyrene	5.18	h
41			4-bromophenyl phenyl ether	5.08	g
64	X	X	pentachlorophenol	5.00	d
40			4-chlorophenyl phenyl ether	4.92	g
20			2-chloronaphthalene	4.72	g
81	X	X	phenanthrene	4.57	h
98			endrin	4.56	d
78	X	X	anthracene	4.54	h
109	X	X	PCB-1232	4.48	
80	X	X	fluorene	4.38	d
--[q]			methoxychlor	4.30	b
52	X	X	hexachlorobutadiene	4.28	f
66	X		bis(2-ethylhexyl)phthalate	4.20	d
68	X		di-n-butyl phthalate	4.13	m
77	X	X	acenaphthylene	4.07	
67	X		butyl benzyl phthalate	4.05	b
108	X	X	PCB-1221	4.00	

TABLE A-2. (Continued)

Priority Pollutant No.	PSDAA	PSEP	Substance	log(K_{ow})	Reference
8	X	X	1,2,4-trichlorobenzene	3.98	l
12	X	X	hexachloroethane	3.93	b
1	X	X	acenaphthene	3.92	b
102			alpha-HCH	3.85	p
103			beta-HCH	3.85	p
104			delta-hexachlorocyclohexane	3.85	h
--[r]			parathion	3.81	e
7			chlorobenzene	3.79	d
105	X	X	gamma-HCH	3.72	h
21			2,4,6-trichlorophenol	3.69	c
95			alpha-endosulfan	3.60	
96			beta-endosulfan	3.60	
97			endosulfan sulfate	3.60	
49			fluorotrichloromethane (removed)	3.53	c
26	X	X	1,3-dichlorobenzene	3.48	l
25	X	X	1,2-dichlorobenzene	3.38	l
27	X	X	1,4-dichlorobenzene	3.38	l
55	X	X	naphthalene	3.36	h
113			toxaphene	3.30	
38	X		ethylbenzene	3.15	
62	X	X	N-nitrosodiphenylamine	3.13	b
22			para-chloro-meta cresol	3.10	a
31	X	X	2,4-dichlorophenol	3.08	a
28			3,3'-dichlorobenzidine	3.02	
37			1,2-diphenylhydrazine	2.94	g
58			4-nitrophenol	2.91	d
--[r]			malathion	2.89	e
60			4,6-dinitro-o-cresol	2.85	
6			tetrachloromethane	2.64	d
42			bis(2-chloroisopropyl)ether	2.58	g
85	X	X	tetrachloroethene	2.53	b
11			1,1,1-trichloroethane	2.47	b
34	X		2,4-dimethylphenol	2.42	b
87	X	X	trichloroethene	2.42	b
15			1,1,2,2-tetrachloroethane	2.39	b
47			bromoform	2.30	
32			1,2-dichloropropane	2.28	
86			toluene	2.21	b
--[r]			guthion	2.18	
14			1,1,2-trichloroethane	2.18	
24			2-chlorophenol	2.16	b
50			dichlorodifluoromethane (removed)	2.16	c
4			benzene	2.11	d
51			chlorodibromomethane	2.08	
35			2,4-dinitrotoluene	2.00	
36			2,6-dinitrotoluene	2.00	
33			1,3-dichloropropene	1.98	
30			1,2-trans-dichloroethene	1.97	c

TABLE A-2. (Continued)

Priority Pollutant No.	PSDAA	PSEP	Substance	$\log(K_{ow})$	Reference
--[r]			demeton	1.93	
23		X	chloroform	1.90	b
48			dichlorobromomethane	1.88	
56			nitrobenzene	1.83	b
5			benzidine	1.81	g
13			1,1-dichloroethane	1.78	
57			2-nitrophenol	1.77	
54			isophorone	1.67	b
71	X		dimethyl phthalate	1.61	b
16			chloroethane	1.54	
59			2,4-dinitrophenol	1.53	
29			1,1-dichloroethene	1.48	
65	X	X	phenol	1.46	a
10			1,2-dichloroethane	1.45	b
70	X		diethyl phthalate	1.40	b
63			N-nitrosodipropylamine	1.31	
44			dichloromethane	1.30	
19			2-chloroethylvinylether	1.28	g
43			bis(2-chloroethoxy)methane	1.26	g
3			acrylonitrile	1.20	b
18			bis(2-chloroethyl)ether	1.12	b
46			bromomethane	1.00	
2			acrolein	0.90	b
45			chloromethane	0.90	
88			vinyl chloride	0.60	
61			N-nitrosodimethylamine	-0.58	g

[a] Veith et al. (1979a).

[b] Veith et al. (1980).

[c] Gossett et al. (1983).

[d] Veith et al. (1979b).

[e] Kenaga and Goring (1980).

[f] Leo, A., 20 November 1984, personal communication.

[g] U.S. EPA (1980).

[h] Karickhoff (1981).

[i] Rapaport and Eisenreich (1984).

[j] Miller et al. (1985).

[k] Means et al. (1980).

[l] Miller et al. (1984).

[m] McDuffie (1981).

[n] Chiou et al. (1981).

TABLE A-2. (Continued)

[o] Briggs (1981).

[p] Solubilities of the various isomers of HCH indicate that they will have similar log(K_{ow}) values.

[q] Chlorinated pesticides that are not on the priority pollutant list but are included in Section 301(h) (Clean Water Act) monitoring programs.

[r] Organophosphorus pesticides that are not on the priority pollutant list but are included in Section 301(h) (Clean Water Act) monitoring programs.

octanol-water partition coefficients (Pastorok 1988). Note that organic compounds with a log octanol-water partition coefficient greater than or equal to 2.3 were recommended by Williams and Pastorok (1985) for inclusion in EPA Section 301(h) (Clean Water Act) monitoring programs.

Bioaccumulation Exposure Scenario

The recommended exposure scenario focuses on transfer of contaminants from sediments to fish or shellfish to humans. Contaminant concentration guidelines developed below may be compared with corresponding concentrations in tissues of aquatic organisms exposed to test sediments in the laboratory (e.g., the clam *Macoma* spp.). A hypothetical exposure scenario is summarized below:

- Humans may potentially harvest fish from all subtidal areas of Puget Sound. English sole is recommended as an indicator species.

- Per capita consumption rate of sportfish is 11 grams/day (Option 1) or 20 grams/day (Option 2). It is assumed that only the skinned fillet is consumed.

- Shellfish may be harvested from intertidal and shallow subtidal areas. Native littleneck clam (*Protothaca staminea*) and butter clam (*Saxidomus giganteus*) are recommended as indicator species.

- Per capita consumption rate of shellfish is 1.3 grams/day. It is assumed that all of the soft parts of bivalves are consumed, including the gut.

- In the absence of site-specific field data on the recommended indicator species, the contaminant concentrations in the bivalve *Macoma* spp. (entire soft parts) after 30-day laboratory exposure to sediments may be used as a surrogate indicator to compare with human health guidelines.

- The average daily consumption rates cited above are assumed to apply continuously over a 70-year period.

Various fish species are harvested recreationally from Puget Sound (e.g., Landolt et al. 1985, 1987). English sole is recommended as an indicator species for assessment of bioaccumulation because of its association with fine, organically rich sediments, its tendency to accumulate many contaminants in edible muscle, and its wide geographic distribution. English sole is commonly consumed by shoreside anglers of Puget Sound (Landolt et al. 1987). Pastorok et al. (1985) pointed out the potential importance of PCB contamination of Puget Sound fish in light of human health concerns. Definitive studies of the relative degree of contamination among fish species in Puget Sound have not been performed. For example, there is a discrepancy between the results of two major studies (Gahler et al. 1982; Landolt et al. 1985) of the relative magnitude of PCB contamination in English sole and other species. Nevertheless, the average concentration of PCBs in English sole muscle was within an order of magnitude of that in other sportfish. Faigenblum (1988) presents some comparative data for several bivalve shellfish, including the native littleneck clam (*P. staminea*) and the butter clam (*S. giganteus*), the two species recommended here as candidates for the hypothetical bioaccumulation exposure scenario. Because these species live in the sediments and are relatively

sedentary, they should represent site-specific conditions. *Macoma* spp. are recommended as surrogate species for laboratory testing because it is a surface deposit-feeder that has been used widely for bioaccumulation studies in the laboratory and in the field. Because of its close association with the sediments, *Macoma* spp. are expected to show greater bioaccumulation of contaminants such as PCBs than are suspension feeders such as *P. staminea* or *S. giganteus*.

The assumed consumption rate estimate of 11 grams/day for fish is the average daily per capita consumption of all sportfish combined for shoreside anglers in Puget Sound (Landolt et al. 1987). This is likely to be an overestimate of the actual consumption rate because some fish species are actually harvested only during a limited season. Nevertheless, this is a plausible estimate for sportfish consumption. An estimate of 20 grams/day for fish consumption is proposed as an option. Average values for fish and shellfish consumption for the United States population generally range from 6.5 to 20.4 grams/day (Nash 1971; U.S. NMFS 1976, 1984; SRI 1980; U.S. DOA 1984). Most estimates include fish and shellfish (molluscs, crustaceans) in marine, estuarine, and fresh waters, but saltwater species form the bulk of consumed items. Most estimates also include commercially harvested fisheries products. Estimates of average United States consumption do not account for subpopulations in areas such as the Great Lakes that consume large quantities ($\geq$20 grams/day) of locally caught sport fish. Consumption rates for portions of the United States population (e.g., by region, age, race, and sex) show that average consumption of fisheries organisms may vary from about 6 to 100 grams/day (e.g., Suta 1978; SRI 1980; Puffer et al. 1982). Finch (1973) determined that approximately 0.1 percent (i.e., the 99.9th percentile) of the United States population consumes 165 grams/day of commercially harvested fish and shellfish. Pao et al. (1982) provided estimates of 48 grams/day for the average and 128 grams/day for the 95th percentile consumption rates by United States consumers of fish and shellfish.

Data on rates of shellfish consumption by Puget Sound harvesters were not available. An estimate of 1.3 grams/day is recommended for the hypothetical exposure scenario. This estimate corresponds to the estimated consumption of clams, oysters, and scallops combined from estuarine and fresh waters in the United States (U.S. EPA 1980).

COMPILATION OF TOXICITY DATA

Toxicity data used in developing the interim guidelines are summarized in this section. Sources of toxicity data relevant to human health risk assessment are also discussed.

The interim guidelines were derived from Carcinogenic Potency Factors and RfD shown in Tables A-3 and A-4. These values were compiled from the Superfund Public Health Evaluation Manual (U.S. EPA 1986f) and IRIS (U.S. EPA 1988). IRIS was used as the primary source of information. Data on classification of carcinogens are also available from NTP and IARC. Information on tributyltin, a potential contaminant of concern in Puget Sound was not available. Also, note that the Carcinogenic Potency Factor for 2,3,7,8-tetrachlorodibenzodioxin is presently under review by EPA.

The weight-of-evidence classification used by EPA to indicate confidence in the classification of a chemical as a carcinogen is summarized below (as extracted from U.S. EPA 1986a):

TABLE A-3. CARCINOGENIC POTENCY FACTORS AND WEIGHT OF EVIDENCE FOR CARCINOGENICITY OF PSDDA AND PSEP CHEMICALS OF CONCERN

Pollutant	PSDDA	PSEP	Potency[a] $(mg/kg/day)^{-1}$	Weight of Evidence[b]
Phenols				
phenol	X	X	--[c]	--
2,4-dimethylphenol	X		--	--
Substituted Phenols				
2,4,6-trichlorophenol			0.0199	B2*
para-chloro-meta cresol			--	--
2-chlorophenol			--	--
2,4-dichlorophenol			--	--
2-nitrophenol			--	--
4-nitrophenol			--	--
2,4-dinitrophenol			--	--
4,6-dinitro-o-cresol			--	--
pentachlorophenol	X	X	--	D
Organonitrogen Compounds				
benzidine			230 (I)	A
3,3'-dichlorobenzidine			1.69	B2*
2,4-dinitrotoluene			0.31	B2*
2,6-dinitrotoluene			--	C
1,2-diphenylhydrazine			0.77	B1*
nitrobenzene			--	--
N-nitrosodimethylamine			25.9 (B)	B2*
N-nitrosodiphenylamine	X	X	0.00492	--*
N-nitrosodipropylamine			--	B2
Low Molecular Weight Aromatic Hydrocarbons				
acenaphthene	X	X	--	--
naphthalene	X	X	--	--
acenaphthylene	X	X	--	--
anthracene	X	X	--	--
phenanthrene	X	X	--	--
fluorene	X	X	--	--
High Molecular Weight PAH				
fluoranthene	X	X	--	--
benzo(a)anthracene	X	X	--	--
benzo(a)pyrene	X	X	11.5	B2*
benzo(b)fluoranthene			d	B2
benzo(k)fluoranthene			--	D
chrysene	X	X	d	B2
benzo(g,h,i)perylene	X	X	--	--
dibenzo(a,h)anthracene	X	X	d	B2
indeno(1,2,3-cd)pyrene	X	X	d	C
pyrene	X	X	--	--
Chlorinated Aromatic Hydrocarbons				
1,2,4-trichlorobenzene	X	X	--	--
hexachlorobenzene	X	X	1.69	B2
2-chloronaphthalene			--	--
1,2-dichlorobenzene	X	X	--	--
1,3-dichlorobenzene	X	X	--	--
1,4-dichlorobenzene	X	X	--	--
Chlorinated Aliphatic Hydrocarbons				
hexachlorobutadiene	X	X	0.00775	C
hexachloroethane	X	X	0.0142	C
hexachlorocyclopentadiene			--	--

TABLE A-3. (Continued)

Pollutant	PSDDA	PSEP	Potency[a] $(mg/kg/day)^{-1}$	Weight of Evidence[b]
Halogenated Ethers				
bis(2-chloroethyl)ether			1.14	B2
4-chlorophenyl phenyl ether			--	--
4-bromophenyl phenyl ether			--	--
bis(2-chloroisopropyl)ether			--	--
bis(2-chloroethoxy)methane			--	--
Phthalates				
bis(2-ethylhexyl)phthalate	X		0.00068	B2
butyl benzyl phthalate	X		--	--
di-n-butyl phthalate	X		--	--
di-n-octyl phthalate	X		--	--
diethyl phthalate	X		--	--
dimethyl phthalate	X		--	--
PCBs				
PCBs			4.34	B2
Miscellaneous Oxygenated Compounds				
2,3,7,8-TCDD (dioxin)		X	156,000	B2
isophorone			--	--
Pesticides				
aldrin	X	X	11.4	B2
dieldrin	X	X	30	B2
chlordane	X		1.61	B2
4,4'-DDT	X	X	0.34	B2
4,4'-DDE			0.34	B2
4,4'-DDD			0.34	B2
alpha-endosulfan			--	--
beta-endosulfan			--	--
endosulfan sulfate			--	--
endrin			--	--
endrin aldehyde			--	--
heptachlor	X	X	3.37	B2
heptachlor epoxide			2.6	B2
alpha-HCH			11.12	B2
beta-HCH			1.84	C
delta-HCH			--	--
gamma-HCH	X	X	1.33	B2/C
toxaphene			1.13	B2
Volatile Halogenated Alkanes				
tetrachloromethane			0.13	B2
1,2-dichloroethane			0.091	B2
1,1,1-trichloroethane			--	--
1,1-dichloroethane			--	--
1,1,2-trichloroethane			0.0573	C
1,1,2,2-tetrachloroethane			0.2	C
chloroethane			--	--
chloroform		X	0.081	B2
1,2-dichloropropane			--	--
dichloromethane			0.00750	B2
chloromethane			--	--
bromomethane			--	--
bromoform			--	--
dichlorobromomethane			--	--
chlorodibromomethane			--	--

TABLE A-3. (Continued)

Pollutant	PSDDA	PSEP	Potency[a] $(mg/kg/day)^{-1}$	Weight of Evidence[b]
Volatile Halogenated Alkenes				
1,1-dichloroethene			0.58	C
			1.16 (I)	C
1,2-trans-dichloroethene			--	--
1,3-dichloropropene			--	--
tetrachloroethene	X	X	0.051	B2
			0.0017 (I)	B2
trichloroethene	X	X	0.011	B2
			0.0046 (I)	B2
vinyl chloride			2.3	A
			0.025 (I)	A
Volatile Aromatic Hydrocarbons				
benzene			0.052	A
			0.026 (I)	A
ethylbenzene	X		--	--
toluene			--	--
Volatile Chlorinated Aromatic Hydrocarbons				
chlorobenzene			--	--
Volatile Unsaturated Carbonyl Compounds				
acrolein			--	--
acrylonitrile			0.24 (W)	B1*
Volatile Ethers				
2-chloroethylvinyl ether			--	--
Metals				
antimony	X	X	--	--
arsenic	X	X	15 (H)	A
beryllium			2.6	B1*
			4.86 (I)	B1
cadmium	X	X	6.1 (I)	B1
chromium III			--	--
chromium VI			41 (I)	A
copper	X	X	--	--
lead	X	X	--	--
mercury	X	X	--	--
nickel (subsulfide, refinery dust)	X	X	1.05 (W)	A*
			1.19 (I)	A
selenium			--	--
silver	X	X	--	--
thallium			--	--
zinc	X	X	--	--
Miscellaneous				
cyanide	X	X	--	--

NOTE: Chromium (VI), cadmium, beryllium, and nickel are not considered to be carcinogenic via dietary exposure.

Reference: U.S. EPA (1985a, 1986f). Asterisk indicates potency value was available only in U.S. EPA (1985a).

[a] All slopes calculated as upper 95 percent confidence limit of slope (q_1*) based on animal oral data and linearized multistage model except:

(B) = slope calculated from 1-hit model
(W) = slope calculated from occupational exposure (including inhalation route)
(H) = slope calculated from human drinking water exposure
(I) = slope calculated from inhalation studies (including animal studies).

TABLE A-3. (Continued)

[b] U.S. EPA (1988) weight of evidence classification for carcinogenicity of chemicals:

Group A = Human carcinogen
Group B = Probable human carcinogen
Group B1 = Chemicals with limited epidemiological evidence for carcinogenicity
Group B2 = Chemicals with sufficient evidence of carcinogenicity from animal studies but inadequate or no evidence from epidemiological studies
Group C = Possible human carcinogen
Group D = Not classifiable
Group E = Evidence of noncarcinogenicity for humans.

[c] No data available.

[d] Potency factor for benzo(a)pyrene may be applied as an interim value to other carcinogenic PAH.

TABLE A-4. REFERENCE DOSE (RfD) VALUES FOR PSDDA AND PSEP CHEMICALS OF CONCERN[a]

Pollutant	PSDDA	PSEP	RfD (mg/kg/day) Oral Route
Phenols			
phenol	X	X	0.04
2,4-dimethylphenol	X		--[b]
Substituted Phenols			
2,4,6-trichlorophenol			--
para-chloro-meta cresol			--
2-chlorophenol			--
2,4-dichlorophenol			0.003
2-nitrophenol			--
4-nitrophenol			--
2,4-dinitrophenol			0.002[c]
4,6-dinitro-o-cresol			--
pentachlorophenol	X	X	0.03
Organonitrogen Compounds			
benzidine			--
3,3'-dichlorobenzidine			--
2,4-dinitrotoluene			--
2,6-dinitrotoluene			--
1,2-diphenylhydrazine			--
nitrobenzene			0.0005
N-nitrosodimethylamine			--
N-nitrosodiphenylamine	X	X	--
N-nitrosodipropylamine			--
Low Molecular Weight Aromatic Hydrocarbons			
acenaphthene	X	X	--
naphthalene	X	X	--
acenaphthylene	X	X	--
anthracene	X	X	--
phenanthrene	X	X	--
fluorene	X	X	--
High Molecular Weight PAH			
fluoranthene	X	X	--
benzo(a)anthracene	X	X	--
benzo(a)pyrene	X	X	--
benzo(b)fluoranthene			--
benzo(k)fluoranthene			--
chrysene	X	X	--
benzo(g,h,i)perylene	X	X	--
dibenzo(a,h)anthracene	X	X	--
indeno(1,2,3-c,d)pyrene	X	X	--
pyrene	X	X	--
Chlorinated Aromatic Hydrocarbons			
1,2,4-trichlorobenzene	X	X	0.02
hexachlorobenzene	X	X	--
2-chloronaphthalene			--
1,2-dichlorobenzene	X	X	--
1,3-dichlorobenzene	X	X	--
1,4-dichlorobenzene	X	X	--
Chlorinated Aliphatic Hydrocarbons			
hexachlorobutadiene	X	X	0.002
hexachloroethane	X	X	0.014
hexachlorocyclopentadiene			0.007

TABLE A-4. (Continued)

Pollutant	PSDDA	PSEP	RfD (mg/kg/day) Oral Route
Halogenated Ethers			
bis(2-chloroethyl)ether			--
4-chlorophenyl phenyl ether			--
4-bromophenyl phenyl ether			--
bis(2-chloroisopropyl)ether			--
bis(2-chloroethoxy)methane			--
Phthalates			
bis(2-ethylhexyl)phthalate	X		0.02
butyl benzyl phthalate	X		--
di-n-butyl phthalate	X		0.1
di-n-octyl phthalate	X		--
diethyl phthalate	X		--
dimethyl phthalate	X		--
PCBs			
PCBs			--
Miscellaneous Oxygenated Compounds			
2,3,7,8-TCDD (dioxin)		X	--
isophorone			0.15
Pesticides			
aldrin	X	X	0.00003[c]
dieldrin	X	X	--
chlordane	X		0.00005[c]
4,4'-DDT	X	X	0.0005
4,4'-DDE			--
4,4'-DDD			--
alpha-endosulfan			--
beta-endosulfan			--
endosulfan sulfate			--
endrin			--
endrin aldehyde			--
heptachlor	X	X	--
heptachlor epoxide			0.00003[c]
alpha-HCH			--
beta-HCH			--
delta-HCH			--
gamma-HCH	X	X	0.0003
toxaphene			--
Volatile Halogenated Alkanes			
tetrachloromethane			--
1,2-dichloroethane			--
1,1,1-trichloroethane			0.54[c]
1,1-dichloroethane			0.12[c]
1,1,2-trichloroethane			--
1,1,2,2-tetrachloroethane			--
chloroethane			--
chloroform		X	0.01
1,2-dichloropropane			--
dichloromethane			0.06[c]
chloromethane			--
bromomethane			0.0004[c]
bromoform			--
dichlorobromomethane			--
chlorodibromomethane			--

TABLE A-4. (Continued)

Pollutant	PSDDA	PSEP	RfD (mg/kg/day) Oral Route
Volatile Halogenated Alkenes			
1,1-dichloroethene			0.009
1,2-trans-dichloroethene			--
1,3-dichloropropene			--
tetrachloroethene	X	X	0.02
trichloroethene	X	X	--
vinyl chloride			--
Volatile Aromatic Hydrocarbons			
benzene			--
ethylbenzene	X	X	0.1
toluene			0.3
total xylenes	X		2
Volatile Chlorinated Aromatic Hydrocarbons			
chlorobenzene			0.027[c]
Volatile Unsaturated Carbonyl Compounds			
acrolein			--
acrylonitrile			--
Volatile Ethers			
2-chloroethylvinyl ether			--
Metals			
antimony	X	X	0.0004
arsenic	X	X	--
beryllium			0.005
cadmium	X	X	0.00029[c]
chromium III			1
chromium VI			0.005[c]
copper	X	X	0.037[c]
lead	X	X	0.0014[c]
mercury (inorganic)	X	X	0.002[c]
nickel	X	X	0.01[c]
selenium			0.003[c]
silver	X	X	0.003
thallium			0.0004[c]
zinc	X	X	0.21[c]
Miscellaneous			
cyanide	X	X	0.02

[a] RfD values taken from IRIS (U.S. EPA 1988a) except as noted.

[b] No data available.

[c] Data from Superfund Public Health Evaluation Manual (U.S. EPA 1986f), pp. 149-156.

- **Group A - Human Carcinogen:** This group is used only when there is sufficient evidence from epidemiologic studies to support a causal association between exposure to the agents and cancer.

- **Group B - Probable Human Carcinogen:** This group includes agents for which the weight of evidence of human carcinogenicity based on epidemiologic studies is "limited." It also includes agents for which the weight of evidence of carcinogenicity based on animal studies is "sufficient." The group is divided into two subgroups. Usually, Group B1 is reserved for agents for which there is limited evidence of carcinogenicity from epidemiologic studies. It is reasonable, for practical purposes, to regard an agent for which there is "sufficient" evidence of carcinogenicity in animals as presenting a carcinogenic risk to humans. Therefore, agents for which there is "sufficient" evidence from animal studies and for which there is "inadequate" evidence or "no data" from epidemiologic studies would usually be categorized under Group B2.

- **Group C - Possible Human Carcinogen:** This group is used for agents with limited evidence of carcinogenicity in animals in the absence of data on humans. It includes a wide variety of evidence [e.g., (a) a malignant tumor response in a single, well-conducted experiment that does not meet conditions for sufficient evidence; (b) tumor responses of marginal statistical significance in studies having inadequate design or reporting; (c) benign but not malignant tumors with an agent showing no response in a variety of short-term tests for mutagenicity; and (d) response of marginal statistical significance in a tissue known to have a high or variable background rate].

- **Group D - Not Classifiable as to Human Carcinogenicity:** This group is generally used for agents with inadequate human and animal evidence of carcinogenicity or for which no data are available.

- **Group E - Evidence of Noncarcinogenicity for Humans:** This group is used for agents that show no evidence for carcinogenicity in at least two adequate animal tests in different species or in both adequate epidemiologic and animal studies. The classification of an agent in Group E is based on the available evidence and should not be interpreted as a definitive conclusion that the agent is not a carcinogen under any circumstances.

In applying the interim guidelines and in developing final guidelines, the state will probably rely on toxicity profiles generated previously. IRIS (U.S. EPA 1988) is a key source of chemical toxicity data in support of EPA Carcinogenic Potency Factors and RfD. IRIS includes information from critical studies and weight-of-evidence classifications for carcinogens. Other primary sources of toxicological data include IARC, NTP, the Agency for Toxic Substances and Disease Registry, and the World Health Organization. Additional sources of toxicity data relevant to risk assessment related to human consumption of fish and shellfish are summarized by Pastorok (1988). U.S. OSTP (1985) described sources of FDA data for exposure assessments, including food consumption surveys and information on natural toxicants in foods.

INTERIM GUIDELINES BASED ON BIOACCUMULATION IN FOOD CHAINS

EPA risk assessment data and models were used to calculate interim guidelines for contaminant concentrations in edible tissues of fish and shellfish (Tables A-5 and A-6). The general procedure used to derive the guidelines is summarized in Pastorok et al. (1986). Key assumptions of the approach are presented in Table A-7. For noncarcinogens, RfD values from U.S. EPA (1988) were used to calculate the guidelines. For carcinogens, tolerable risk levels of 10^{-4}, 10^{-5}, and 10^{-6} were used. Note that a lifetime cancer risk of one in a million or less is generally considered tolerable, whereas a lifetime risk above approximately one in a thousand is generally considered unacceptable by federal regulatory agencies (Travis et al. 1987). Risk levels on the order of one in ten thousand to one in one-hundred thousand have often led to development of environmental regulations on chemical releases and exposure of humans. Tolerable risk levels defined by EPA under the Resource Conservation and Recovery Act range from 10^{-4} to 10^{-7}.

TABLE A-5. GUIDELINE CONCENTRATIONS OF CARCINOGENS IN FISH OR SHELLFISH AT SELECTED RISK LEVELS AND CONSUMPTION RATES

Pollutant	Concentration (mg/kg wet weight) Guidelines[a]								
	$Ri^*=10^{-4}$			$Rj^*=10^{-5}$			$Ri^*=10^{-6}$		
	I*=0.0013	I*=0.011 (kg/day)	I*=0.02	I*=0.0013	I*=0.011 (kg/day)	I*=0.02	I*=0.0013	I*=0.011 (kg/day)	I*=0.02
Substituted Phenols									
2,4,6-trichlorophenol	300	30	20	30	3	2	3	0.3	0.2
Organitrogen Compounds									
benzidine	0.02	0.003	0.002	0.002	0.0003	0.0002	0.0002	0.00003	0.00002
3,3'-dichlorobenzidine	3	0.4	0.2	0.3	0.04	0.02	0.03	0.004	0.002
2,4-dinitrotoluene	20	2	1	2	0.2	0.1	0.2	0.02	0.01
1,2-diphenylhydrazine	7	0.8	0.5	0.7	0.08	0.05	0.07	0.008	0.005
N-nitrosodimethylamine	0.2	0.02	0.01	0.02	0.002	0.001	0.002	0.0002	0.0001
N-nitrosodiphenylamine	1,000	100	70	100	10	7	10	1	0.7
High Molecular Weigh PAH									
benzo(a)pyrene[b]	0.5	0.06	0.03	0.05	0.006	0.003	0.005	0.0006	0.0003
Chlorinated Aromatic Hydrocarbons									
hexachlorobenzene	3	0.4	0.2	0.3	0.04	0.02	0.03	0.004	0.002
Chlorinated Aliphatic Hydrocarbons									
hexachlorobutadiene	700	80	50	70	8	5	7	0.8	0.5
hexachloroethane	400	40	20	40	4	2	4	0.4	0.2
Halogenated Ethers									
bis(2-chloroethyl)ether	5	0.6	0.3	0.5	0.06	0.03	0.05	0.006	0.003
Phthalates									
bis(2-ethylhexyl)phthalate	8,000	900	500	800	90	50	80	9	5
PCBs									
PCBs	1	0.1	0.08	0.1	0.01	0.008	0.01	0.001	0.0008
Miscellaneous Oxygenated Compounds									
2,3,7,8-TCDD (dioxin)	0.00003	0.000004	0.000002	0.000003	0.0000004	0.0000002	0.0000003	0.00000004	0.00000002
Pesticides									
aldrin	0.5	0.06	0.03	0.05	0.006	0.003	0.005	0.0006	0.0003
dieldrin	0.2	0.02	0.01	0.02	0.002	0.001	0.002	0.0002	0.0001
chlordane	3	0.4	0.2	0.3	0.04	0.02	0.03	0.004	0.002
4,4'-DDT	20	2	1	2	0.2	0.1	0.2	0.02	0.01
4,4'-DDE	20	2	1	2	0.2	0.1	0.2	0.02	0.01
4,4'-DDD	20	2	1	2	0.2	0.1	0.2	0.02	0.01
heptachlor	2	0.2	0.1	0.2	0.02	0.01	0.02	0.002	0.001
heptachlor epoxide	2	0.2	0.1	0.2	0.02	0.01	0.02	0.002	0.001
alpha-HCH	0.5	0.06	0.03	0.05	0.006	0.003	0.005	0.0006	0.0003
beta-HCH	3	0.3	0.2	0.3	0.03	0.02	0.03	0.003	0.002
gamma-HCH	4	0.5	0.3	0.4	0.05	0.03	0.04	0.005	0.003
toxaphene	5	0.6	0.3	0.5	0.06	0.03	0.05	0.006	0.003

TABLE A-5. (Continued)

Pollutant	Concentration (mg/kg wet weight) Guidelines[a]								
	$Ri^*=10^{-4}$			$Ri^*=10^{-5}$			$Ri^*=10^{-6}$		
	I*=0.0013	I*=0.011 (kg/day)	I*=0.02	I*=0.0013	I*=0.011 (kg/day)	I*=0.02	I*=0.0013	I*=0.011 (kg/day)	I*=0.02
Volatile Halogenated Alkanes									
tetrachloromethane	40	5	3	4	0.5	0.3	0.4	0.05	0.03
1,2-dichloroethane	60	7	4	6	0.7	0.4	0.6	0.07	0.04
1,1,2-trichloroethane	90	10	6	9	1	0.6	0.9	0.1	0.06
1,1,2,2-tetrachloroethane	30	3	2	3	0.3	0.2	0.3	0.03	0.02
chloroform	70	8	4	7	0.8	0.4	0.7	0.08	0.04
dichloromethane	700	80	50	70	8	5	7	0.8	0.5
Volatile Halogenated Alkenes									
1,1-dichloroethene	9	1	0.6	0.9	0.1	0.06	0.09	0.01	0.006
	5	0.5	0.3	0.5	0.05	0.03	0.05	0.005	0.003
tetrachloroethene	100	10	7	10	1	0.7	1	0.1	0.07
trichloroethene	500	60	30	50	6	3	5	0.6	0.3
vinyl chloride	2	0.3	0.2	0.2	0.03	0.02	0.02	0.003	0.002
	200	30	10	20	3	1	2	0.3	0.1
Volatile Aromatic Hydrocarbons									
benzene	100	10	7	10	1	0.7	1	0.1	0.07
Volatile Unsaturated Carbonyl Compounds									
acrylonitrile	20	3	1	2	0.3	0.1	0.2	0.03	0.01
Metals									
arsenic[c]	0.4	0.04	0.02	0.04	0.004	0.002	0.004	0.0004	0.0002

[a] R_i^* = Selected reference risk values.
I* = Assumed seafood consumption rates. See text for rationale for selected values shown in table.

Where two rows of values are shown for a single column, the first applies to the oral route of exposure and the second applies to the inhalation route. See Table A-3 for basis of carcinogenic potency factors.

[b] Values for benzo(a)pyrene may be applied to other carcinogenic PAH.

[c] Values apply to inorganic forms of arsenic only. On average, inorganic arsenic may account for approximately 0.12 percent of the total arsenic in fish and shellfish of Puget Sound (Crecelius and Apts 1985).

TABLE A-6. GUIDELINE CONCENTRATIONS OF CHEMICALS IN FISH OR SHELLFISH AT SELECTED CONSUMPTION RATES BASED ON REFERENCE DOSES (RfDs)[a]

	Concentration (mg/kg wet weight) Guidelines[b]		
	I*=0.0013	I*=0.011 (kg/day)	I*=0.02
Phenols			
phenol	2,000	300	100
Substituted Phenols			
2,4-dichlorophenol	200	20	10
2,4-dinitrophenol	100	10	7
pentachlorophenol	2,000	200	100
Organonitrogen Compounds			
nitrobenzene	30	3	2
Chlorinated Aromatic Hydrocarbons			
1,2,4-trichlorobenzene	1,000	100	70
Chlorinated Aliphatic Hydrocarbons			
hexachlorobutadiene	100	10	7
hexachloroethane	800	90	50
hexachlorocyclopentadiene	400	40	20
Phthalates			
bis(2-ethylhexyl)phthalate	1,000	100	70
di-n-butyl phthalate	5,000	600	400
Miscellaneous Oxygenated Compounds			
isophorone	8,000	1,000	500
Pesticides			
aldrin	2	0.2	0.1
chlordane	3	0.3	0.2
4,4'-DDT	30	3	2
heptachlor epoxide	2	0.2	0.1
gamma-HCH	20	2	1
Volatile Halogenated Alkanes			
1,1,1-trichloroethane	30,000	3,000	2,000
1,1-dichloroethane	6,000	800	400
chloroform	500	60	40
dichloromethane	3,000	400	200
bromomethane	20	3	1
Volatile Halogenated Alkenes			
1,1-dichloroethene	500	60	30
tetrachloroethene	1,000	100	70
Volatile Aromatic Hydrocarbons			
ethylbenzene	5,000	600	400
toluene	20,000	2,000	1,000
Volatile Chlorinated Aromatic Hydrocarbons			
chlorobenzene	1,000	200	90

TABLE A-6. (Continued)

	Concentration (mg/kg wet weight) Guidelines[b]		
	I*=0.0013	I*=0.011 (kg/day)	I*=0.02
Metals			
antimony	20	3	1
beryllium	300	30	20
cadmium	20	2	1
chromium III	50,000	6,000	4,000
chromium VI	300	30	20
copper	2,000	200	100
lead	80	9	5
mercury	100	10	7
nickel	500	60	40
selenium	200	20	10
silver	200	20	10
thallium	20	3	1
zinc	10,000	1,000	700
cyanide	1,000	100	70

[a] Based on oral RfD values.

[b] I* = Various assumed seafood consumption rates. See text for rationale for values shown.

TABLE A-7. SUMMARY OF ASSUMPTIONS AND NUMERICAL ESTIMATES USED IN RISK ASSESSMENT APPROACH

Parameter	Assumptions/Estimates	Reference
Exposure Assessment:		
Contaminant concentrations in tissues of indicator species.	No effect of cooking	Worst case for parent compounds. Net effect on risk is uncertain.
Average consumption rate[a]	11 g/day--fish (Option 1) 20 g/day--fish (Option 2) 1.3 g/day--shellfish	Representative values (see text)
Gastrointestinal absorption coefficient	1.0 - Assumes efficiency of absorption of contaminants is same for humans and bioassay animals	U.S. EPA 1980, 1986a,b
Exposure duration	70 years	U.S. EPA 1980, 1986a,b
Human body weight	70 kg (= average adult male)	U.S. EPA 1986a,b
Dose-Response Assessment		
Carcinogenic potency	Potency factors are based on low-dose extrapolation from animal bioassay data	U.S. EPA 1985a, 1986f
Reference Doses (RfDs)	RfDs represent a lifetime daily dose that is unlikely to result in substantial risk of a health effect.	U.S. EPA 1986f, 1988
Risk Characterization		
Carcinogenic risk model	Linearized multistage used to back calculate guidelines from selected risk value	Pastorok et al. 1986 Pastorok 1988
Noncarcinogenic risk	Guidelines calculated from RfDs for noncarcinogenic effects	Pastorok et al. 1986 Pastorok 1988

[a] Estimates of consumption for local population should be used in place of values shown for U.S. population whenever possible.

RELATIONSHIP OF LIVER NEOPLASMS IN FISH TO HUMAN CANCER RISK

Techniques for assessing human health risks from low-level exposure to environmental chemicals rely on extrapolation of chemical effects in other animals (usually rats or mice) to humans. Thus, it is logical to consider the possibility of using information on diseases in Puget Sound fish to assess chemical hazards or risks to humans. Studies of disease in fish or humans are concerned primarily with deviations from normal structure, physiology, biochemistry, and cellular and molecular biology. Most field studies of fish liver histopathology have focused primarily on the morphologic changes that occur in response to harmful environmental stimuli. Many of these changes are relatively stable and amenable to some form of quantification.

Adverse effects of neoplasms on the health of affected fish have not been documented, although they also have not been well studied. Mix (1986) noted that there have been no systematic studies of the ecological effects of neoplasms on fish populations. In contrast, the adverse effects of human cancers, including death for many kinds of cancer, are well known. In this regard, it is important to distinguish the terms "neoplasm" and "cancer". A neoplasm is a swelling of the tissue (i.e., tumor) due to new growth of abnormal tissue associated with abnormally rapid cellular growth. Strictly speaking, the term cancer denotes a malignant neoplasm (i.e., one that has the capacity to spread to distant sites in the body). In human health risk assessment, the distinction between cancerous tumors and benign neoplasms is often blurred because both benign and malignant tumors are considered equally as severe adverse effects.

Issues surrounding the use of information on fish liver neoplasms (i.e., cancerous tumors) to assess human health hazards associated with contaminated sediments are addressed in this section. The relationship between the occurrence of fish liver neoplasms and contaminated sediments is described. The primary objective of this section is to compare the characteristics of neoplasms, their development, and factors that potentially influence their incidence in fish with those in mammals, including humans. Finally, differences in exposure factors between fish and humans are described. These exposure-related differences preclude the application of dose-response data on wild-fish to quantitative assessment of human health risks. A key conclusion of this evaluation is that data on fish liver neoplasms in Puget Sound and other aquatic environments can be used as an indicator of potential carcinogens in the environment, but they can not be used directly to estimate human health risks.

FISH LIVER NEOPLASMS AND CONTAMINATED SEDIMENTS

Malins et al. (1984), Mix (1986), and Becker and Grieb (1987) reviewed previous studies of liver disease in fishes of Puget Sound. Liver lesions, including neoplasms, have been documented in four species of bottomfish from polluted areas of Puget Sound: English sole (*Parophrys vetulus*), starry flounder (*Platichthys stellatus*), rock sole (*Lepidopsetta bilineata*), and Pacific staghorn sculpin (*Leptocottus armatus*). In general, highest prevalences of most liver abnormalities were found in major urbanized areas for all four fishes. Lowest prevalences generally were found in nonurban areas.

Using multivariate and bivariate statistical analyses, Malins et al. (1984) found positive associations between sediment concentrations of aromatic hydrocarbons and certain liver lesions in English sole and Pacific staghorn sculpin, and between sediment concentrations of metals and certain liver lesions in English sole.

Myers et al. (1987) described the rationale for the hypothesis that liver neoplasms in Puget Sound fish, particularly English sole, may be caused by exposure to carcinogenic chemicals (particularly PAH) in sediment. They cite the following evidence in support of this hypothesis:

- Statistically significant associations between concentrations of aromatic hydrocarbons (a group of chemicals that includes known carcinogens) and the prevalences of neoplasms and other liver lesions in English sole
- Uptake and metabolism of potentially carcinogenic aromatic hydrocarbons by feral English sole
- Statistically significant associations between the concentrations of metabolites of aromatic hydrocarbons in the bile of English sole and prevalences of neoplasms and other liver lesions
- The similarity of the histopathology of neoplasms and other liver lesions in English sole from polluted areas of Puget Sound to liver lesions found in rats, mice and fish exposed to liver carcinogens in the laboratory.

Mix (1986) and Becker and Grieb (1987) reviewed other field studies throughout the United States that suggest an association of liver neoplasms in certain fish species with sediments contaminated by potentially carcinogenic chemicals.

Determining the specific causes of pathological liver conditions in feral fishes rarely is possible because these organisms generally are exposed to an unknown diversity of potentially harmful stimuli (e.g., infectious, nutritional, chemical, physical). Possible interactions among stimuli that modify their individual effects (e.g., synergism, antagonism) further complicate causal determinations.

Although most field studies of fish liver histopathology are limited to observing morphologic changes, laboratory studies frequently consider the etiology, pathogenesis, or functional changes related to the morphologic changes. In the majority of laboratory studies, fishes are exposed to a single stimulus under carefully controlled conditions. The pathological conditions that result can thus be attributed with reasonable confidence to the effects of the test stimulus. In addition, by monitoring the test organisms over time, the pathogenesis and functional changes involved with a particular condition often can be observed.

Meyers and Hendricks (1982), Couch and Harshbarger (1985), and Becker and Grieb (1987) reviewed the evidence for induction of liver lesions in fishes upon exposure to potentially carcinogenic chemicals in controlled laboratory studies. Approximately 90 chemicals have been found to induce lesions in the liver of fishes. Of these, approximately 28 induced neoplasms.

Because of the difficulty and expense of testing the long-term effects (e.g., cancer induction) of contaminated sediments on fish in the laboratory, it is anticipated

that field data on fish liver neoplasms and other lesions would be considered in assessments of human health risk. Field data are also needed to address effects of complex mixtures in the natural environment. The remainder of this report focuses on issues regarding extrapolation of field data on fish liver neoplasms to humans. Nevertheless, it is recognized that any use of fish neoplasm data from field studies to assess carcinogenic potential in humans would need to be supported by laboratory verification studies.

COMPARATIVE ASPECTS OF FISH AND HUMAN NEOPLASIA

This evaluation of the relationship of fish liver neoplasms to human cancer risk is based primarily on comparison of each of the following factors between fish and humans (or other mammals):

- Stages in development of neoplasia
- Modifying factors that affect quantitative extrapolation (e.g., differential sensitivity due to differences in metabolism, diet, lifespan, and environmental agents other than chemicals)
- Exposure pathways.

This is not intended to be a detailed evaluation of neoplasia induction in fish and humans. Rather, the following discussion focuses on key issues that may affect use of fish pathology data in regulation of contaminated sediments based on human health concerns.

Comparison of Neoplasia in Fish and Humans

Detailed comparative studies of the characteristics of neoplasia in humans and fish were not found during a limited literature search. Most of the available information on mechanisms of carcinogenesis and stages in the development of neoplasms in humans is based on studies of rats and mice as models of humans. It is generally accepted that cellular and subcellular mechanisms of carcinogenesis are common to diverse species of animals (although the details of those mechanisms are unknown). Available information on comparisons of neoplasia in fish and rats or mice indicates that stages in the development of liver neoplasms in fish are similar to an established series of liver lesions induced in rodents by exposure to chemical carcinogens in the laboratory. Myers et al. (1987) provide comprehensive documentation of close morphological similarities between liver lesions in feral fish and the well known sequence of lesions in laboratory rats and mice.

There is strong experimental evidence that neoplasm formation is a progressive process involving multiple steps and multiple exposures to stimuli. It is therefore possible that neoplasms may be induced by simultaneous or sequential exposure to several different carcinogens.

All chemical carcinogens fall into one of two groups. The first group is termed direct-acting (or activation-independent) carcinogens. These chemicals do not require any kind of modification to exert their carcinogenic effect. However, they sometimes can be chemically or enzymatically inactivated.

The second group of carcinogens is termed procarcinogens. These chemicals require some form of metabolic conversion to produce metabolites capable of inducing neoplasms. Procarcinogens are often called parent compounds, whereas their carcinogenic metabolites are called ultimate carcinogens. Many procarcinogens are activated by liver enzymes. Although procarcinogens require metabolic activation to be carcinogenic, they can also be metabolized to noncarcinogenic end-products (i.e., detoxified). Procarcinogens include potent carcinogens such as PAH, nitrosamines, and aflatoxins.

There is strong evidence that many chemical carcinogens induce tumors by interacting with DNA, indicating they are mutagenic. However, tumors could also be induced by the interaction of carcinogens with RNA and proteins. Chemical carcinogenesis involves at least two stages: initiation and promotion. Initiation can theoretically result from a single interaction of a carcinogenic chemical with DNA. An initiated cell is altered permanently, making it more likely to give rise to a neoplasm. Initiation alone cannot induce neoplasms, but must be followed by promotion. Promotion increases the probability that an initiated cell will ultimately result in a neoplasm. Because initiation is irreversible, promotion does not have to follow it immediately. Most promoters do not induce tumors by themselves. However, some chemicals can act as both initiators and promoters, and are thus called complete carcinogens.

The relative sensitivity of fish and humans (or other mammals) to carcinogens has not been well studied. Janardan et al. (1984) generally found high ($r>0.7$) correlations between the acute toxicity of organic EPA priority pollutants to fish (bluegills and fathead minnows) and the acute toxicity to rats. These authors suggested that their results indicated a similar mode of toxic action among species. However, these results may have little bearing on relative sensitivity to carcinogens. Trout are known to be particularly sensitive to known human carcinogens, but the relative sensitivity of English sole and other Puget Sound flatfish is unknown. It is likely that sensitivity to carcinogens differs greatly among fish species and between certain fish species and humans. Differential sensitivity is thought to be related to metabolic rate. Crouch and Wilson (1979) demonstrated that humans are more sensitive to carcinogens [based on similar dose rates per unit body weight (e.g., mg carcinogen ingested per kg body weight per day)] than rats and mice are. Corrections for differential sensitivity of rodents and humans are made by regulatory agencies that perform human health risk assessment. Similar corrections could be made between fish and humans based on knowledge of differences in body surface area, body weight, or metabolic rate.

Comparative aspects of metabolism of toxic chemicals (e.g., conversion of procarcinogens to an active form, detoxification and elimination of toxic chemicals), target site in the body, average lifespan of individuals, and other factors that affect the validity of quantitative extrapolation of data on fish neoplasms to humans also are poorly known. The particular site of appearance of neoplasia in the body depends on the chemical, route of exposure, and species-specific metabolism. The similarity in metabolic functioning between fish species and mammals (Guarino 1987) and the limited observations available on chemical causes of liver neoplasms in feral fish (Myers et al. 1987) suggests that chemicals that induce neoplasms in the livers of fish may also be capable of inducing liver neoplasms in humans. The rate of appearance of spontaneous neoplasms (i.e., not induced by chemicals or other applied agents) in fish and mammals must also be considered. Spontaneous neoplasms are common in certain laboratory strains of rats and mice. Although hybrids of certain fish species are susceptible to spontaneous neoplasms in the laboratory, naturally occurring fish species may have a low background incidence of

neoplasms. This appears to be true for rainbow trout (Hendricks et al. 1984) and English sole (Becker and Grieb 1987).

Modifying factors that may affect the incidence of neoplasms in fish or human populations exposed to chemical carcinogens include the following:

- Diet
- Viruses
- Radiant energy
- Water quality variables such as salinity/temperature (fish only)
- Genetic composition.

As discussed below, the general role of these modifying factors is similar in fish and humans (or other mammals as models for humans). However, the quantitative influence of each factor on the prevalence of liver neoplasms in fish is unknown.

Comparative studies of the influence of modifying factors on neoplasia in fish and mammals have not been conducted. Relevant fish studies include documentation of the influence of diet (Bailey et al. 1984), viruses (studies summarized by Black 1984), temperature (Kyono-Hamaguchi 1984; Hendricks et al. 1984), and genetic susceptibility (Cooper and Keller 1969). Based on mammalian studies, radiant energy in the form of ultraviolet rays, x-rays, gamma rays, and ionizing particles (alpha particles, beta particles, protons, neutrons) can induce neoplasms. Radiant energy can damage DNA and cellular membranes, alter proteins, and inactivate enzymes. However, the exact event responsible for producing neoplastic cells is unknown. Much of the evidence suggests that radiant energy exerts its carcinogenicity through interactions with DNA, indicating a mutagenic pathway.

Exposure Factors

Differences in potential exposure to toxic chemicals in sediments also limits extrapolation of data on fish liver neoplasms to predict human health effects. Guarino (1987) compared fish and humans (and laboratory models) with respect to exposure and absorption of toxic chemicals. He noted the general lack of information on the potential for gastrointestinal absorption of toxic chemicals by fish and the need for studies that quantify chemical uptake rates via gills, gut, and skin separately. As noted earlier, the primary human exposure route for chemicals in sediments is through the food chain. Fish may be exposed through direct contact with sediments, contact with overlying water, or ingestion of contaminated prey. The obvious differences between the aquatic habitat of fish and the terrestrial habitat of humans has major implications for the pathways, routes, magnitude, and duration of exposure to toxic chemicals associated with contaminated sediments. Moreover, because fish are part of the food chain linking contaminated sediments to human exposure, the differential uptake of various chemicals by fish and their subsequent transformation must be considered. For example, fish may absorb PAH originally associated with sediments and subsequently transform or eliminate them (e.g., Varanasi et al. 1985). Although fish liver neoplasms may be induced by

such processes (Myers et al. 1987), the contaminants are generally not found in edible tissues of the fish and therefore would not be available to fish eaters, including humans.

Although the exposure factors just discussed preclude quantitative extrapolation of data on fish liver neoplasms to humans, they do not limit the use of such data to assess qualitative or relative human health hazards from contaminated sediments. One consideration in interpreting the exposure of fish to contaminated sediments that bears on qualitative extrapolations is the issue of fish movements, including long-distance migrations. If fish travel great distances, then association of observed neoplasms with specific areas of contaminated sediments is severely limited. Because English sole have been well studied, because they appear to be sensitive to chemical carcinogens, and because they tend to associate with fine-grained, organically rich habitats which have a high potential for contamination, this species is best suited for assessment of contaminated sediments. Although little is known about movements of English sole, it is clear from data on the spatial distribution of liver lesions and contaminants in tissue that individuals of this species exhibit somewhat limited movements along shorelines. Based on the distribution of lesions observed in the Elliott Bay and Commencement Bay studies (Becker et al. 1987; Beller et al. 1988), it may be speculated that the "home territory" of English sole covers on the order of 1-2 km of shoreline. Thus, the use of liver lesions in this species as an indicator of potentially contaminated sediments is appropriate for areas on the order of the East or West Waterways of the Duwamish River system or the individual waterways of the Commencement Bay system.

APPLICATION OF FISH PATHOLOGY DATA TO HUMAN HEALTH RISK ASSESSMENT

Quantitative extrapolation of information on fish liver neoplasms to assess potential human health effects from contaminated sediments is limited by lack of comparative data on dose-response relationships and by differential exposure factors just discussed. Prediction of human health risks or dose-response relationships to contaminated sediments is clearly not warranted at present. Nevertheless, the use of data on fish liver neoplasms as an indicator of potentially carcinogenic material in the environment is promising. Therefore, it is recommended that relationships between prevalence of liver lesions in English sole and contaminated sediments be developed further. In the long-term (e.g., over the next 5-10 years), it may be possible to develop guidelines for unacceptable levels of contaminants in sediments based on relationships between fish neoplasms and sediment chemistry. These guidelines could be used in conjunction with short-term tests for mutagenicity/carcinogenicity to assess potential human health hazards and to rank sediments in terms of priority for regulatory action. However, the limitations of using fish data to assess health hazards, even in a qualitative manner, must be recognized. Moreover, the relevance of toxicological endpoints other than neoplasia in fish-human food chains (e.g., Rogan et al. 1986; Humphrey 1988) warrants further development of comparative interspecies data on birth defects, embryo and fetal toxicity, and corresponding short-term tests to assess contaminated sediments.

APPENDIX B

Background Material on Sediment Quality Values Data Treatment

INTRODUCTION

A preliminary list of Puget Sound stations that exceed the recommended Element P-2 sediment standards for biological effects (Option 2 in Section 4 of this report) is provided in Table B-1. Alternative sediment quality values and the data sets used to test the reliability of sediment quality values proposed for developing Element P-2 sediment standards are also tabulated in this appendix. In addition, data treatment options implemented for determining adverse biological effects and anomalous chemical concentrations during the development of AET values are summarized (see Barrick et al. 1988 for further discussion).

ALTERNATIVE SEDIMENT QUALITY VALUES

1988 AET values (dry weight) for Puget Sound that were used in reliability analyses are tabulated in Table B-2. Selected AET values based on chemical data normalized to total organic carbon are presented in Table B-3. Alternative sediment quality criteria values generated using the SLC approach, the Triad approach, and the sediment-water equilibrium partitioning approach, are tabulated in Table B-4.

DATA SETS USED FOR ANALYSES

The 1988 Puget Sound AET are based on data collected in 11 Puget Sound surveys (Table B-5) in 13 embayments. These data sets include various reconnaissance and dredging surveys, studies of contamination and biological effects in Elliott Bay and the central basin of Puget Sound by the Municipality of Metropolitan Seattle (Metro), state or federal Superfund investigations in Commencement Bay and Eagle Harbor, Urban Bay Action programs in Elliott Bay and Everett Harbor, and associated reference area surveys.

Contaminated sediments in Commencement Bay waterways and along the Ruston-Pt. Defiance Shoreline were characterized during a remedial investigation that resulted in the identification of eleven high priority problem areas and multiple potential sources of contamination. Approximately 50 samples were collected for chemical and biological measurements. The Eagle Harbor biological effects data were collected during a preliminary investigation conducted for Ecology's state Superfund program. PSEP sponsored the collection of bioassay and benthic infauna data for 10 stations sampled during the investigation, primarily to provide data for evaluation of sediment quality values at a site known to be heavily contaminated with PAH. The Elliott Bay and Everett Harbor data (approximately 150 stations) were collected as part of Urban Bay Toxics Action Programs conducted by PSEP in these embayments located adjacent to the cites of Seattle and Everett, respectively. Contamination in each area is attributed to a variety of industrial, commercial, and residential sources.

A summary of stations and biological test types for all surveys that were used in this project is provided in Table B-6. Location maps of all stations are given in Figures B-1 through B-20. Reliability analyses (presented in Section 3 of this report)

for these stations were conducted using a menu-driven program (SEDQUAL) developed for the Puget Sound sediment quality values database. A detailed description of the database and menu-driven features is given in the SEDQUAL users manual (Nielsen 1988).

TREATMENT OF DATA USED TO GENERATE 1988 AET

Options have been developed and discussed by Barrick et al. (1988) to address statistical treatment of biological data, chemical data qualifiers, and anomalous chemical concentrations in data sets used to establish or verify sediment quality values. A summary of options and recommendations for treating anomalous data is presented in Table B-7. Such anomalous values are not necessarily incorrect or unreliable, but may not be representative of Puget Sound conditions. Of special concern are stations that do not appear to have biological effects, but are chemically contaminated at concentrations that are well above those at other nonimpacted stations. The inclusion of such anomalous stations may increase AET values as a result of highly localized, nonrepresentative conditions. Alternatively, such stations may simply indicate the need for additional data to confirm an increase in AET values that is more representative of Puget Sound conditions.

Implementation of all of the procedures recommended in Table B-7 to all AET data types has not been completed (a summary of the status of the implemented options is also shown in Table B-7). The methods for treatment of biological and chemical data used to generate 1988 AET presented in this report are described in the following sections. Implementation of the following biological and chemical guidelines for anomalies resulted in a net increase of 8 percent in the sensitivity of 1988 amphipod AET (from 50 to 58 percent) and a net increase of 4 percent in the sensitivity of 1988 benthic infauna AET (from 71 to 75 percent), when used to predict impacts in the expanded database. Because all Puget Sound data have been used to generate these 1988 AET, their efficiency is by definition 100 percent for the Puget Sound database of biological effect stations that pass both biological and chemical guidelines for anomalies.

Bioassay Data

In generating 1988 AET, several modifications to procedures used previously (e.g., Beller et al. 1986) were made for the inclusion of new data for the amphipod mortality bioassay. The recommendations summarized in Table B-7 were made to improve the consistency of the results among the various pooled studies. Similar modifications have not been made for the oyster larvae abnormality bioassay and the Microtox bioassay because each of these indicators is represented by only a single study area in the SEDQUAL database.

The first modification addressed the level of significance at which pairwise comparisons between impacted and reference sites were judged significant. A level of $P \leq 0.05$ comparisonwise was consistently used for all comparisons between stations, instead of using an experimentwise error rate that would result in variable *alpha* levels for pairwise station comparisons depending on the sample size of different studies (see Appendix C of Barrick et al. 1988).

The second modification considered screening criteria and power analyses for the amphipod tests conducted in the Elliott Bay and Everett Harbor PSEP surveys. A detailed discussion of these analyses is presented in Appendix C of Barrick et al. 1988. This modification resulted in the exclusion of seven potentially nonimpacted stations for which there was inadequate statistical power to distinguish significant effects relative to reference conditions. The excluded stations are summarized in Table B-8.

By applying the guidelines in Table B-7 to all amphipod bioassay stations in the SEDQUAL database, impacted stations were those that exhibited statistically significant mortality ($P \leq 0.05$) relative to reference conditions <u>and</u> exceeded 25 percent mortality (see Appendix C of Barrick et al. 1988 for discussion of this criterion as a minimum level of concern). Twenty stations exhibited statistically significant mortality ($P \leq 0.05$), but were classified as nonimpacted because mean mortality was ≤ 25 percent. These stations were subsequently included in the data set used to calculate AET.

Benthic Infauna Data

Several modifications were also made for treatment of benthic infauna data to improve consistency of the results among the various pooled studies. As with the amphipod bioassay data, a comparisonwise significance level of $P \leq 0.05$ was used for all pairwise benthic comparisons. Power analyses were not conducted for benthic infauna because they were beyond the scope of this project. In lieu of a power analysis, and analogous to the amphipod bioassay guideline, a guideline was developed to ensure that significant benthic effects were of sufficient magnitude to be of concern as adverse impacts and to be discriminated statistically in most cases. Thus, only significant effects ($P \leq 0.05$) that also exceeded a 50 percent reduction in major taxa abundance were considered impacts. Five stations [(Station AP-01 from the Alki Point survey, Stations CI-20, RS-14, and SI-15 from the Commencement Bay survey, and Station WP-16 from the Metro Toxic Pretreatment Planning Study (Romberg et al. 1984; see Appendix B)] exhibited significant benthic effects ($P \leq 0.05$) but were considered nonimpacted because depressions in abundance were ≤ 50 percent. These stations were subsequently included in the data set used to calculate AET.

This guideline was derived partly from consideration of the natural variability of benthic infauna in relatively undisturbed environments of Puget Sound. Based on a summary of data from Lie (1968), Nichols (1975), and Word et al. (1984) in Tetra Tech (1987), the abundances of selected major taxa (Polychaeta, Mollusca, Crustacea) and total infauna may vary seasonally by roughly a factor of 2 (i.e., lowest mean abundances are roughly 50 percent of the highest mean abundances). In most cases, ≥ 50 percent reductions in mean abundance can be detected statistically ($P \leq 0.05$), whereas ≤ 30 percent reductions cannot be detected ($P > 0.05$). Finally, the guideline of 50 percent reduction in benthic infauna abundances provides a recommended level of environmental protectiveness for regulatory application with a reasonable balance between underprotection due to tolerance of major effects and overprotectiveness due to misclassification of nonimpacted sites as impacted. Nevertheless, a criterion based on 50 percent cannot be considered as sensitive or protective as a criterion based on a lower value (e.g., 30 percent depression).

Chemical Data

All detected chemical data entered in SEDQUAL after quality assurance review were included in AET calculations. These calculations were based on biological effect stations that passed the biological screening criteria summarized in the previous section.

A guideline was adopted for this report to address the concern over anomalously high chemical concentrations at nonimpacted stations (e.g., because of unusual matrix effects or low bioavailability). From a technical perspective (i.e., representativeness of data used in AET generation), the AET database was screened for biologically nonimpacted sediments exhibiting chemical concentrations that were anomalously high by a factor of 3 from that at the nonimpacted station exhibiting the next highest concentration (see Appendix C of Barrick et al. 1988 for additional details). From a management perspective, this guideline generates more protective (sensitive) sediment quality standards that may also be less efficient in only identifying problem sediments.

The purpose of this guideline was to acknowledge potentially nonrepresentative data for exceptional chemical matrices (e.g., slag, coal), or unusual biological conditions (e.g., extremely tolerant species under localized conditions). Only a limited number of stations were identified using this guideline. For the amphipod bioassay, this guideline affected 8 of 295 stations that passed biological guidelines discussed in the previous section and resulted in changes to the AET for nine chemicals where the ratio of the anomalous station to the nonimpacted station with the next highest concentration ranged from 3.2 to 14.3 (Table B-9). Amphipod bioassay stations excluded according to this criterion are summarized in Table B-8.

For benthic infauna, this procedure resulted in the exclusion of 4 of the 205 stations (Table B-8). The procedure also resulted in changes to the AET for seven chemicals [including high molecular weight HPAH as a class] where the ratio of the anomalous station to the nonimpacted station with the next highest concentration ranged from 3.0 to 20.0 (see Table B-9). Data currently excluded as anomalous (Table B-8) are intended to be restored if and when confirming data become available.

01/09/89

TABLE B-1. PRELIMINARY LIST OF STATIONS EXCEEDING NEC DEFINED IN OPTION 2 IN SECTION 4.5.2; SUCH STATIONS WOULD COMPRISE AN INITIAL INVENTORY OF SEDIMENTS UNDER ELEMENT P-2 WHICH WILL BE MAPPED IN A FORTHCOMING REPORT

SURVEY	STN_ID	DATE	SRV_SAMPID	TEST_TYPE	FIELD_REP	SUBSAMPLE	NUM_CHEMS	BIO_EFFECT			
ALKI	AP-04	05/25/84	AP-04	LAET			1		BENA*		
CBBLAIR	B02	06/01/84	B02	LAET			1				
CBBLAIR	B03	06/01/84	B03	LAET			7	AMPT	BENA		OYST
CBBLAIR	B04	06/01/84	B04	LAET			19	AMPT	BENA		OYST
CBBLAIR	B07	06/01/84	B07	LAET			3				
CBBLAIR	B09	06/01/84	B09	LAET			6	AMPT	BENA		OYST
CBBLAIR	B10	06/01/84	B10	LAET			3	AMPT	BENA		OYST*
CBBLAIR	B11	06/01/84	B11	LAET			3				
CBBLAIR	B12	06/01/84	B12	LAET			3	AMPT	BENA		OYST
CBBLAIR	B14	06/01/84	B14	LAET			6				
CBBLAIR	B15	06/01/84	B15	LAET			6	AMPT*	BENA		OYST
CBBLAIR	B17	06/01/84	B17	LAET			2				
CBBLAIR	B18	06/01/84	B18	LAET			2				
CBMSQS	BL-11	01/01/84	BL-11	LAET			4	AMPT	BENA	MICB	OYST
CBMSQS	BL-12	01/01/84	BL-12	LAET			8				
CBMSQS	BL-13	01/01/84	BL-13	LAET			8	AMPT	BENA	MICB*	OYST
CBMSQS	BL-14	01/01/84	BL-14	LAET			21				
CBMSQS	BL-15	01/01/84	BL-15	LAET			2				
CBMSQS	BL-16	01/01/84	BL-16	LAET			16				
CBMSQS	BL-17	01/01/84	BL-17	LAET			5				
CBMSQS	BL-18	01/01/84	BL-18	LAET			8				
CBMSQS	BL-19	01/01/84	BL-19	LAET			4				
CBMSQS	BL-20	01/01/84	BL-20	LAET			7				
CBMSQS	BL-21	01/01/84	BL-21	LAET			8	AMPT	BENA	MICB	OYST
CBMSQS	BL-22	01/01/84	BL-22	LAET			12				
CBMSQS	BL-23	01/01/84	BL-23	LAET			17				
CBMSQS	BL-24	01/01/84	BL-24	LAET			14				
CBMSQS	BL-25	01/01/84	BL-25	LAET			8	AMPT*	BENA	MICB	OYST
CBMSQS	BL-26	01/01/84	BL-26	LAET			18				
CBMSQS	BL-27	01/01/84	BL-27	LAET			3				
CBMSQS	BL-28	01/01/84	BL-28	LAET			8	AMPT	BENA	MICB	OYST
CBMSQS	BL-29	01/01/84	BL-29	LAET			20				
CBMSQS	BL-30	01/01/84	BL-30	LAET			7				

CBMSQS	BL-31	01/01/84	BL-31	LAET	10	AMPT	BENA	MICB*	OYST
CBMSQS	BL-32	01/01/84	BL-32	LAET	19				
CBMSQS	CB-11	01/01/84	CB-11	LAET	16				
CBMSQS	CB-12	01/01/84	CB-12	LAET	5				
CBMSQS	CB-13	01/01/84	CB-13	LAET	8				
CBMSQS	CB-14	01/01/84	CB-14	LAET	8				
CBMSQS	CI-11	01/01/84	CI-11	LAET	34	AMPT*	BENA*	MICB*	OYST*
CBMSQS	CI-12	01/01/84	CI-12	LAET	47				
CBMSQS	CI-13	01/01/84	CI-13	LAET	34	AMPT	BENA*	MICB*	OYST*
CBMSQS	CI-14	01/01/84	CI-14	LAET	25				
CBMSQS	CI-15	01/01/84	CI-15	LAET	41				
CBMSQS	CI-16	01/01/84	CI-16	LAET	28		BENA*	MICB*	OYST*
CBMSQS	CI-17	01/01/84	CI-17	LAET	34	AMPT	BENA*	MICB*	OYST
CBMSQS	CI-18	01/01/84	CI-18	LAET	27				
CBMSQS	CI-19	01/01/84	CI-19	LAET	25				
CBMSQS	CI-20	01/01/84	CI-20	LAET	26	AMPT*	BENA	MICB	OYST*
CBMSQS	CI-21	01/01/84	CI-21	LAET	33				
CBMSQS	CI-22	01/01/84	CI-22	LAET	25	AMPT	BENA	MICB	OYST
CBMSQS	CR-11	01/01/84	CR-11	LAET	2	AMPT	BENA	MICB	OYST
CBMSQS	CR-12	01/01/84	CR-12	LAET	1	AMPT	BENA	MICB	OYST
CBMSQS	CR-13	01/01/84	CR-13	LAET	2	AMPT	BENA	MICB	OYST
CBMSQS	CR-14	01/01/84	CR-14	LAET	5	AMPT	BENA	MICB	OYST
CBMSQS	HY-11	01/01/84	HY-11	LAET	16				
CBMSQS	HY-12	01/01/84	HY-12	LAET	26		BENA	MICB*	OYST*
CBMSQS	HY-13	01/01/84	HY-13	LAET	19				
CBMSQS	HY-14	01/01/84	HY-14	LAET	21	AMPT	BENA*	MICB*	OYST
CBMSQS	HY-15	01/01/84	HY-15	LAET	27				
CBMSQS	HY-16	01/01/84	HY-16	LAET	31				
CBMSQS	HY-17	01/01/84	HY-17	LAET	34	AMPT	BENA*	MICB*	OYST*
CBMSQS	HY-18	01/01/84	HY-18	LAET	26				
CBMSQS	HY-19	01/01/84	HY-19	LAET	33				
CBMSQS	HY-20	01/01/84	HY-20	LAET	33				
CBMSQS	HY-21	01/01/84	HY-21	LAET	36				
CBMSQS	HY-22	01/01/84	HY-22	LAET	50	AMPT*	BENA*	MICB*	OYST*
CBMSQS	HY-23	01/01/84	HY-23	LAET	34	AMPT*	BENA*	MICB*	OYST*
CBMSQS	HY-24	01/01/84	HY-24	LAET	34	AMPT	BENA*	MICB*	OYST
CBMSQS	HY-25	01/01/84	HY-25	LAET	31				
CBMSQS	HY-26	01/01/84	HY-26	LAET	21				
CBMSQS	HY-27	01/01/84	HY-27	LAET	23				
CBMSQS	HY-28	01/01/84	HY-28	LAET	30	AMPT	BENA*	MICB	OYST
CBMSQS	HY-29	01/01/84	HY-29	LAET	26				

CBMSQS	HY-30	01/01/84	HY-30	LAET	10				
CBMSQS	HY-31	01/01/84	HY-31	LAET	16				
CBMSQS	HY-32	01/01/84	HY-32	LAET	22	AMPT	BENA*	MICB	OYST
CBMSQS	HY-33	01/01/84	HY-33	LAET	27				
CBMSQS	HY-34	01/01/84	HY-34	LAET	25				
CBMSQS	HY-35	01/01/84	HY-35	LAET	33				
CBMSQS	HY-36	01/01/84	HY-36	LAET	36				
CBMSQS	HY-37	01/01/84	HY-37	LAET	29	AMPT	BENA*	MICB*	OYST
CBMSQS	HY-38	01/01/84	HY-38	LAET	30				
CBMSQS	HY-39	01/01/84	HY-39	LAET	28				
CBMSQS	HY-40	01/01/84	HY-40	LAET	29				
CBMSQS	HY-41	01/01/84	HY-41	LAET	31				
CBMSQS	HY-42	01/01/84	HY-42	LAET	33	AMPT*	BENA*	MICB*	OYST
CBMSQS	HY-43	01/01/84	HY-43	LAET	29	AMPT	BENA*	MICB*	OYST
CBMSQS	HY-44	01/01/84	HY-44	LAET	5	AMPT	BENA	MICB	OYST
CBMSQS	HY-45	01/01/84	HY-45	LAET	22				
CBMSQS	HY-46	01/01/84	HY-46	LAET	27				
CBMSQS	HY-47	01/01/84	HY-47	LAET	23	AMPT*	BENA*	MICB*	OYST*
CBMSQS	HY-48	01/01/84	HY-48	LAET	24				
CBMSQS	HY-49	01/01/84	HY-49	LAET	6				
CBMSQS	HY-50	01/01/84	HY-50	LAET	7	AMPT	BENA	MICB*	OYST
CBMSQS	HY-51	01/01/84	HY-51	LAET	4				
CBMSQS	MD-11	01/01/84	MD-11	LAET	38				
CBMSQS	MD-12	01/01/84	MD-12	LAET	38	AMPT	BENA*	MICB	OYST
CBMSQS	MD-13	01/01/84	MD-13	LAET	27				
CBMSQS	MI-11	01/01/84	MI-11	LAET	23	AMPT*	BENA	MICB	OYST
CBMSQS	MI-12	01/01/84	MI-12	LAET	16				
CBMSQS	MI-13	01/01/84	MI-13	LAET	21	AMPT	BENA	MICB*	OYST
CBMSQS	MI-14	01/01/84	MI-14	LAET	12				
CBMSQS	MI-15	01/01/84	MI-15	LAET	12	AMPT*	BENA	MICB*	OYST
CBMSQS	RS-11	01/01/84	RS-11	LAET	29				
CBMSQS	RS-12	01/01/84	RS-12	LAET	18	AMPT	BENA	MICB	OYST
CBMSQS	RS-13	01/01/84	RS-13	LAET	31	AMPT*	BENA	MICB	OYST*
CBMSQS	RS-14	01/01/84	RS-14	LAET	30	AMPT	BENA	MICB	OYST
CBMSQS	RS-15	01/01/84	RS-15	LAET	2				
CBMSQS	RS-16	01/01/84	RS-16	LAET	34				
CBMSQS	RS-17	01/01/84	RS-17	LAET	26				
CBMSQS	RS-18	01/01/84	RS-18	LAET	39	AMPT*	BENA*	MICB*	OYST*
CBMSQS	RS-19	01/01/84	RS-19	LAET	13	AMPT*	BENA*	MICB*	OYST*
CBMSQS	RS-20	01/01/84	RS-20	LAET	6	AMPT	BENA*	MICB*	OYST
CBMSQS	RS-21	01/01/84	RS-21	LAET	37				

CBMSQS	RS-22	01/01/84	RS-22	LAET	5	AMPT		MICB	OYST
CBMSQS	RS-24	01/01/84	RS-24	LAET	11	AMPT*		MICB	OYST
CBMSQS	SI-11	01/01/84	SI-11	LAET	27	AMPT	BENA*	MICB*	OYST
CBMSQS	SI-12	01/01/84	SI-12	LAET	20	AMPT*	BENA*	MICB*	OYST
CBMSQS	SI-13	01/01/84	SI-13	LAET	14				
CBMSQS	SI-14	01/01/84	SI-14	LAET	31				
CBMSQS	SI-15	01/01/84	SI-15	LAET	17	AMPT*	BENA	MICB	OYST
CBMSQS	SP-11	01/01/84	SP-11	LAET	12	AMPT	BENA*	MICB*	OYST
CBMSQS	SP-12	01/01/84	SP-12	LAET	16	AMPT	BENA	MICB*	OYST*
CBMSQS	SP-13	01/01/84	SP-13	LAET	30				
CBMSQS	SP-14	01/01/84	SP-14	LAET	19	AMPT*	BENA*	MICB*	OYST*
CBMSQS	SP-15	01/01/84	SP-15	LAET	9	AMPT*	BENA*	MICB*	OYST*
CBMSQS	SP-16	01/01/84	SP-16	LAET	10	AMPT*	BENA*	MICB*	OYST*
DUWRIV1	DR-03	04/15/85	DR-03	LAET	2	AMPT			
DUWRIV1	DR-04	04/15/85	DR-04	LAET	1	AMPT			
DUWRIV1	DR-06	04/15/85	DR-06	LAET	1	AMPT			
DUWRIV1	DR-08	04/15/85	DR-08	LAET	14	AMPT*			
DUWRIV2	DR-10	07/01/85	CA1	LAET	15	AMPT*			
DUWRIV2	DR-11	07/01/85	CA2	LAET	3	AMPT*			
DUWRIV2	DR-13	07/01/85	CB1	LAET	2	AMPT			
DUWRIV2	DR-14	07/01/85	CB2	LAET	7	AMPT			
DUWRIV2	DR-19	07/01/85	CC2	LAET	3	AMPT			
DUWRIV2	DR-23	07/01/85	CD1	LAET	7	AMPT			
DUWRIV2	DR-25	07/01/85	CE1	LAET	7	AMPT*			
DUWRIV2	DR-26	07/01/85	CE2	LAET	4	AMPT*			
DUWRIV2	DR-27	07/01/85	CE3	LAET	5	AMPT*			
DUWRIV2	DR-28	07/01/85	CF1	LAET	14	AMPT			
DUWRIV2	DR-29	07/01/85	CF2	LAET	7	AMPT*			
DUWRIV2	DR-30	07/01/85	CF3	LAET	2	AMPT			
DUWRIV2	DR-31	07/01/85	CF4	LAET	2	AMPT			
DUWRIV2	DR-33	07/01/85	CG1	LAET	4	AMPT			
DUWRIV2	DR-34	07/01/85	CG2	LAET	3	AMPT			
DUWRIV2	DR-35	07/01/85	CG3	LAET	2	AMPT			
DUWRIV2	DR-36	07/01/85	CG4	LAET	3	AMPT			
DUWRIV2	DR-38	07/01/85	CH1	LAET	3	AMPT			
DUWRIV2	DR-39	07/01/85	CW/A1	LAET	3	AMPT			
EBCHEM	AB-01	09/26/85	AB-01	LAET	32	AMPT*			
EBCHEM	AB-02	09/26/85	AB-02	LAET	21	AMPT	BENA		
EBCHEM	DR-01	09/30/85	DR-01	LAET	1	AMPT			
EBCHEM	DR-02	09/30/85	DR-02	LAET	20	AMPT*			
EBCHEM	DR-03	09/30/85	DR-03	LAET	3	AMPT			

EBCHEM	DR-04	09/30/85	DR-04	LAET		10	AMPT	
EBCHEM	DR-05	09/30/85	DR-05	LAET		3		
EBCHEM	DR-06	10/09/85	DR-06	LAET		3	AMPT	
EBCHEM	DR-07	09/30/85	DR-07	LAET		2	AMPT	
EBCHEM	DR-08	09/30/85	DR-08	LAET		24		
EBCHEM	DR-09	09/30/85	DR-09	LAET		8	AMPT	
EBCHEM	DR-10	09/30/85	DR-10	LAET		12		
EBCHEM	DR-11	09/30/85	DR-11	LAET		17	AMPT	
EBCHEM	DR-12	09/30/85	DR-12	LAET		19		
EBCHEM	DR-13	09/30/85	DR-13	LAET		9	AMPT*	
EBCHEM	DR-14	09/30/85	DR-14	LAET		11	AMPT*	
EBCHEM	DR-15	09/30/85	DR-15	LAET		18	AMPT*	
EBCHEM	DR-16	09/30/85	DR-16	LAET		31	AMPT*	
EBCHEM	DR-17	09/30/85	DR-17	LAET		14	AMPT	
EBCHEM	DR-25	10/10/85	DR-25	LAET		10	AMPT*	
EBCHEM	EW-01	10/09/85	EW-01	LAET		3	AMPT	
EBCHEM	EW-02	10/04/85	EW-02	LAET		36	AMPT*	BENA*
EBCHEM	EW-03	10/04/85	EW-03	LAET		22		BENA
EBCHEM	EW-04	10/14/85	EW-04	LAET		26	AMPT*	BENA*
EBCHEM	EW-05	10/14/85	EW-05	LAET	1	25	AMPT*	BENA*
EBCHEM	EW-05	10/14/85	EW-05	LAET	2	18	AMPT*	BENA*
EBCHEM	EW-06	10/04/85	EW-06	LAET		35	AMPT*	BENA*
EBCHEM	EW-07	10/14/85	EW-07	LAET		19	AMPT*	BENA*
EBCHEM	EW-08	10/14/85	EW-08	LAET		18	AMPT*	BENA*
EBCHEM	EW-09	10/14/85	EW-09	LAET		22	AMPT*	BENA*
EBCHEM	EW-10	10/14/85	EW-10	LAET		18	AMPT*	BENA
EBCHEM	EW-11	10/14/85	EW-11	LAET		26	AMPT*	BENA*
EBCHEM	EW-12	10/15/85	EW-12	LAET		18	AMPT	BENA
EBCHEM	EW-13	10/15/85	EW-13	LAET		14	AMPT	BENA
EBCHEM	EW-14	10/15/85	EW-14	LAET		34	AMPT	BENA
EBCHEM	EW-15	10/15/85	EW-15	LAET		21	AMPT	BENA
EBCHEM	EW-16	10/15/85	EW-16	LAET		3	AMPT*	BENA
EBCHEM	KG-01	09/25/85	KG-01	LAET		25	AMPT	BENA*
EBCHEM	KG-02	10/09/85	KG-02	LAET		3	AMPT*	
EBCHEM	KG-03	09/25/85	KG-03	LAET		18	AMPT*	BENA
EBCHEM	KG-04	10/09/85	KG-04	LAET		6	AMPT	
EBCHEM	KG-05	09/30/85	KG-05	LAET		12	AMPT*	BENA*
EBCHEM	KG-06	09/30/85	KG-06	LAET		17	AMPT	BENA*
EBCHEM	KG-07	09/30/85	KG-07	LAET		13	AMPT	BENA*
EBCHEM	KG-08	10/01/85	KG-08	LAET		2	AMPT	BENA*
EBCHEM	KG-09	10/01/85	KG-09	LAET		20	AMPT*	BENA

EBCHEM	KG-10	10/08/85	KG-10	LAET		10	AMPT*	
EBCHEM	KG-11	10/01/85	KG-11	LAET		3	AMPT*	BENA*
EBCHEM	MG-01	09/26/85	MG-01	LAET		1	AMPT	BENA*
EBCHEM	NH-01	10/15/85	NH-01	LAET		18	AMPT	BENA*
EBCHEM	NH-02	10/15/85	NH-02	LAET		19	AMPT*	BENA*
EBCHEM	NH-03	10/16/85	NH-03	LAET		39	AMPT*	BENA*
EBCHEM	NH-04	10/15/85	NH-04	LAET		35	AMPT*	BENA*
EBCHEM	NH-05	10/15/85	NH-05	LAET		28	AMPT*	BENA*
EBCHEM	NH-06	10/16/85	NH-06	LAET		37	AMPT*	BENA*
EBCHEM	NH-08	10/16/85	NH-08	LAET		35	AMPT*	BENA*
EBCHEM	NH-09	10/16/85	NH-09	LAET		3	AMPT*	BENA
EBCHEM	NH-10	10/08/85	NH-10	LAET		3	AMPT	
EBCHEM	NH-11	10/15/85	NH-11	LAET		4	AMPT*	BENA*
EBCHEM	NS-01	10/08/85	NS-01	LAET		6	AMPT*	
EBCHEM	NS-02	09/27/85	NS-02	LAET		6	AMPT	BENA
EBCHEM	NS-03	10/04/85	NS-03	LAET		3	AMPT	BENA
EBCHEM	NS-04	10/08/85	NS-04	LAET		6		
EBCHEM	NS-05	10/04/85	NS-05	LAET		1	AMPT	BENA
EBCHEM	NS-06	09/27/85	NS-06	LAET		4		BENA*
EBCHEM	NS-07	10/04/85	NS-07	LAET		29	AMPT*	BENA
EBCHEM	NS-08	09/26/85	NS-08	LAET		22	AMPT*	BENA*
EBCHEM	PS-02	10/12/85	PS-02	LAET		1	AMPT	BENA
EBCHEM	PS-05	10/15/85	PS-05	LAET		18		
EBCHEM	SS-03	10/04/85	SS-03	LAET		36	AMPT*	BENA*
EBCHEM	SS-04	10/04/85	SS-04	LAET		41		BENA*
EBCHEM	SS-05	10/03/85	SS-05	LAET	SS18	35	AMPT	BENA
EBCHEM	SS-05	10/03/85	SS-05	LAET	SS19	28	AMPT	BENA
EBCHEM	SS-06	10/03/85	SS-06	LAET		33	AMPT*	BENA*
EBCHEM	SS-07	10/03/85	SS-07	LAET		34	AMPT*	BENA
EBCHEM	SS-08	09/27/85	SS-08	LAET		36	AMPT*	BENA*
EBCHEM	SS-09	09/27/85	SS-09	LAET		46	AMPT*	BENA*
EBCHEM	SS-10	09/27/85	SS-10	LAET		28		
EBCHEM	SS-11	09/27/85	SS-11	LAET		25	AMPT	BENA
EBCHEM	SS-12	09/27/85	SS-12	LAET		12	AMPT	BENA
EBCHEM	WW-01	10/01/85	WW-01	LAET		2	AMPT	BENA*
EBCHEM	WW-02	10/09/85	WW-02	LAET		6	AMPT*	
EBCHEM	WW-03	10/01/85	WW-03	LAET		2	AMPT	BENA*
EBCHEM	WW-04	10/01/85	WW-04	LAET		31	AMPT	BENA*
EBCHEM	WW-05	10/01/85	WW-05	LAET		4	AMPT	BENA*
EBCHEM	WW-06	10/01/85	WW-06	LAET	1	21	AMPT	BENA*
EBCHEM	WW-06	10/01/85	WW-06	LAET	2	24	AMPT	BENA*

EBCHEM	WW-08	10/01/85	WW-08	LAET	20	AMPT*	BENA*
EBCHEM	WW-09	10/02/85	WW-09	LAET	33	AMPT*	BENA*
EBCHEM	WW-10	10/02/85	WW-10	LAET	20	AMPT	BENA*
EBCHEM	WW-11	10/02/85	WW-11	LAET	23	AMPT*	BENA*
EBCHEM	WW-12	10/02/85	WW-12	LAET	24	AMPT*	BENA*
EBCHEM	WW-13	10/02/85	WW-13	LAET	17	AMPT	BENA
EBCHEM	WW-14	10/02/85	WW-14	LAET	23		BENA*
EBCHEM	WW-15	10/08/85	WW-15	LAET	4	AMPT	
EBCHEM	WW-16	10/02/85	WW-16	LAET	17	AMPT	BENA*
EBCHEM	WW-17	10/03/85	WW-17	LAET	25	AMPT	BENA*
EBCHEM	WW-18	10/03/85	WW-18	LAET	20	AMPT	BENA*
EBCHEM	WW-19	10/03/85	WW-19	LAET	21	AMPT	BENA*
EBCHEM	WW-20	10/03/85	WW-20	LAET	14	AMPT	BENA*
EHCHEM	BH-01	06/01/85	B1	LAET	2	AMPT	BENA
EHCHEM	BH-02	06/01/85	B1	LAET	3	AMPT	BENA
EHCHEM	EH-01	06/01/85	B1	LAET	1	AMPT	BENA
EHCHEM	EH-02	06/01/85	B1	LAET	3	AMPT	BENA*
EHCHEM	EH-04	06/01/85	V1	LAET	21		
EHCHEM	EH-05	06/01/85	V1	LAET	21	AMPT	BENA*
EHCHEM	EH-06	06/01/85	V6	LAET	17	AMPT	BENA*
EHCHEM	EH-07	06/01/85	V1	LAET	18		
EHCHEM	EH-08	06/01/85	B1	LAET	23		
EHCHEM	EH-08	06/01/85	V1	LAET	21	AMPT*	BENA*
EHCHEM	EH-10	06/01/85	V1	LAET	22		
EHCHEM	EH-11	06/01/85	V1	LAET	21		
EHCHEM	EH-12	06/01/85	B1	LAET	13		
EHCHEM	EH-13	06/01/85	V1	LAET	11		
EHCHEM	EH-14	06/01/85	V1	LAET	16		
EHCHEM	EH-15	06/01/85	V1	LAET	24	AMPT	BENA*
EHCHEM	EH-16	06/01/85	B1	LAET	11		
EHCHEM	EH-16	06/01/85	V1	LAET	21	AMPT	BENA*
EHCHEM	EH-17	06/01/85	V1	LAET	15		
EHCHEM	EH-18	06/01/85	V1	LAET	22		
EHCHEM	EH-19	06/01/85	V1	LAET	25		
EHCHEM	EH-20	06/01/85	V1	LAET	5		
EHCHEM	EH-21	06/01/85	V1	LAET	11		
EHCHEM	EH-22	06/01/85	B1	LAET	14		
EHCHEM	EH-23	06/01/85	V1	LAET	6		
EHCHEM	EH-24	06/01/85	V1	LAET	7		
EHCHEM	RB-01	06/01/85	V1	LAET	1		
EHCHEM	RB-02	06/01/85	V1	LAET	3		

EHCHEM	RB-03	06/01/85	V1	LAET	2	
EHCHEM	RB-04	06/01/85	V1	LAET	1	
EHCHEM	RB-06	06/01/85	B1	LAET	1	
EHCHEM	RB-07	06/01/85	V1	LAET	1	
EHCHEM	WP-01	06/01/85	V1	LAET	2	
EIGHTBAY	BH-03	01/01/82	BH-03	LAET	4	AMPT
EIGHTBAY	BH-04	01/01/82	BH-04	LAET	5	AMPT
EIGHTBAY	BH-05	01/01/82	BH-05	LAET	3	AMPT*
EIGHTBAY	BH-07	01/01/82	BH-07	LAET	8	AMPT
EIGHTBAY	BH-11	01/01/82	BH-11	LAET	1	AMPT
EIGHTBAY	BH-12	01/01/82	BH-12	LAET	2	AMPT
EIGHTBAY	BH-23	01/01/82	BH-23	LAET	2	AMPT*
EIGHTBAY	BH-24	01/01/82	BH-24	LAET	1	AMPT
EIGHTBAY	CS-01	01/01/82	CS-01	LAET	1	AMPT*
EIGHTBAY	CS-15	01/01/82	CS-15	LAET	2	AMPT*
EIGHTBAY	CS-17	01/01/82	CS-17	LAET	3	AMPT*
EIGHTBAY	DB-07	01/01/82	DB-07	LAET	1	AMPT*
EIGHTBAY	DB-15	01/01/82	DB-15	LAET	2	AMPT*
EIGHTBAY	EL-09	01/01/82	EL-09	LAET	6	AMPT
EIGHTBAY	EL-10	01/01/82	EL-10	LAET	17	AMPT
EIGHTBAY	EL-12	01/01/82	EL-12	LAET	2	AMPT
EIGHTBAY	EL-17	01/01/82	EL-17	LAET	5	AMPT
EIGHTBAY	EL-20	01/01/82	EL-20	LAET	7	AMPT
EIGHTBAY	EL-22	01/01/82	EL-22	LAET	3	AMPT
EIGHTBAY	EL-24	01/01/82	EL-24	LAET	1	AMPT
EIGHTBAY	EV-01	01/01/82	EV-01	LAET	14	AMPT*
EIGHTBAY	EV-02	01/01/82	EV-02	LAET	4	AMPT*
EIGHTBAY	EV-03	01/01/82	EV-03	LAET	10	AMPT*
EIGHTBAY	EV-04	01/01/82	EV-04	LAET	17	AMPT*
EIGHTBAY	EV-05	01/01/82	EV-05	LAET	10	AMPT*
EIGHTBAY	EV-06	01/01/82	EV-06	LAET	3	AMPT
EIGHTBAY	EV-07	01/01/82	EV-07	LAET	10	AMPT
EIGHTBAY	EV-11	01/01/82	EV-11	LAET	1	AMPT*
EIGHTBAY	SC-06	01/01/82	SC-06	LAET	9	AMPT*
EIGHTBAY	SC-07	01/01/82	SC-07	LAET	14	AMPT
EIGHTBAY	SC-08	01/01/82	SC-08	LAET	11	AMPT*
EIGHTBAY	SC-14	01/01/82	SC-14	LAET	11	AMPT*
EIGHTBAY	SC-17	01/01/82	SC-17	LAET	11	AMPT*
EIGHTBAY	SC-18	01/01/82	SC-18	LAET	8	AMPT*
EIGHTBAY	SC-19	01/01/82	SC-19	LAET	7	AMPT
EIGHTBAY	SC-20	01/01/82	SC-20	LAET	22	AMPT*

EIGHTBAY	SM-01	01/01/82	SM-01	LAET		2	AMPT*	
EIGHTBAY	SM-03	01/01/82	SM-03	LAET		1	AMPT*	
EIGHTBAY	SM-20	01/01/82	SM-20	LAET		1	AMPT*	
EIGHTBAY	SQ-17	01/01/82	SQ-17	LAET		2	AMPT	
EIGHTBAY	SQ-18	01/01/82	SQ-18	LAET		2	AMPT	
EVCHEM	ES-01	10/06/86	ES-01G	LAET		3	AMPT	
EVCHEM	ES-02	10/06/86	ES-02G	LAET		6	AMPT	
EVCHEM	ES-03	10/06/86	ES-03G	LAET		10	AMPT	
EVCHEM	EW-01	10/07/86	EW-01G	LAET	EW01	27	AMPT*	BENA*
EVCHEM	EW-02	09/30/86	EW-02G	LAET		10		
EVCHEM	EW-03	09/30/86	EW-03G	LAET		9		
EVCHEM	EW-04	09/30/86	EW-04G	LAET	EW04	51	AMPT*	BENA*
EVCHEM	EW-05	09/30/86	EW-05G	LAET		9		
EVCHEM	EW-06	09/30/86	EW-06G	LAET		8		
EVCHEM	EW-07	09/30/86	EW-07G	LAET	EW07	40	AMPT*	BENA*
EVCHEM	EW-08	09/30/86	EW-08G	LAET		10		
EVCHEM	EW-09	09/30/86	EW-09G	LAET		8		
EVCHEM	EW-10	10/01/86	EW-10G	LAET	EW10	29	AMPT*	BENA*
EVCHEM	EW-11	09/30/86	EW-11G	LAET		26		
EVCHEM	EW-12	10/01/86	EW-12G	LAET	EW12	14	AMPT	BENA*
EVCHEM	EW-13	10/01/86	EW-13G	LAET		44		
EVCHEM	EW-14	10/01/86	EW-14G	LAET	EW14	45		BENA*
EVCHEM	EW-15	10/01/86	EW-15G	LAET		28		
EVCHEM	NG-01	10/02/86	NG-01G	LAET	NG01	6	AMPT	BENA
EVCHEM	NG-02	10/02/86	NG-02G	LAET	NG02	7	AMPT	BENA*
EVCHEM	NG-03	10/02/86	NG-03G	LAET	NG03	4	AMPT	BENA*
EVCHEM	NG-04	10/02/86	NG-04G	LAET	NG04	5	AMPT*	BENA*
EVCHEM	NG-05	10/02/86	NG-05G	LAET		6		
EVCHEM	NG-06	10/08/86	NG-06G	LAET	NG06	6	AMPT*	BENA
EVCHEM	NG-07	10/02/86	NG-07G	LAET		3		
EVCHEM	NG-08	10/02/86	NG-08G	LAET		6		
EVCHEM	NG-09	10/03/86	NG-09G	LAET		28		
EVCHEM	NG-10	10/03/86	NG-10G	LAET	NG10	6	AMPT	BENA
EVCHEM	NG-11	10/03/86	NG-11G	LAET		21		
EVCHEM	NG-12	10/15/86	NG-12G	LAET		1	AMPT	
EVCHEM	NG-13	10/15/86	NG-13G	LAET		3	AMPT	
EVCHEM	NG-14	10/15/86	NG-14G	LAET		5	AMPT	
EVCHEM	NG-15	10/15/86	NG-15G	LAET		4	AMPT	
EVCHEM	OG-01	10/09/86	OG-01G	LAET		14		
EVCHEM	OG-02	10/09/86	OG-02G	LAET		15		
EVCHEM	OG-03	10/09/86	OG-03G	LAET	OG03	19	AMPT*	

EVCHEM	OG-04	10/09/86	OG-04G	LAET		19		
EVCHEM	OG-05	10/09/86	OG-05G	LAET		14		
EVCHEM	OG-06	10/09/86	OG-06G	LAET		21		
EVCHEM	OG-07	10/09/86	OG-07G	LAET		18		
EVCHEM	PS-02	10/13/86	PS-02G	LAET	PS02	3	AMPT	BENA
EVCHEM	PS-03	10/13/86	PS-03G	LAET	PS03	2	AMPT	BENA
EVCHEM	PS-04	10/10/86	PS-04G	LAET	PS04	2	AMPT	BENA
EVCHEM	SD-01	10/09/86	SD-01G	LAET	SD01	2	AMPT	BENA*
EVCHEM	SD-02	10/07/86	SD-02G	LAET	SD02	8		BENA
EVCHEM	SD-03	10/02/86	SD-03G	LAET		10		
EVCHEM	SR-01	10/06/86	SR-01G	LAET		2	AMPT	
EVCHEM	SR-02	10/06/86	SR-02G	LAET		1	AMPT	
EVCHEM	SR-03	10/06/86	SR-03G	LAET		6		
EVCHEM	SR-04	10/06/86	SR-04G	LAET		19	AMPT	
EVCHEM	SR-05	10/06/86	SR-05G	LAET		15		
EVCHEM	SR-06	10/06/86	SR-06G	LAET		3		
EVCHEM	SR-07	10/03/86	SR-07G	LAET	SR07	13		BENA*
EVCHEM	SR-08	10/03/86	SR-08G	LAET	SR08	11	AMPT	BENA*
EVCHEM	SS-01	10/06/86	SS-01G	LAET		2	AMPT	
EVCHEM	SS-02	10/06/86	SS-02G	LAET		11		
EVCHEM	SS-03	10/06/86	SS-03G	LAET		11	AMPT	
EVCHEM	SS-04	10/06/86	SS-04G	LAET		1		
EVCHEM	SS-05	10/06/86	SS-05G	LAET		4		
EVCHEM	SS-06	10/06/86	SS-06G	LAET		5		
EVERETT1	EV-20	01/01/85	EV-20	LAET		6	AMPT	
EVERETT1	EV-21	01/01/85	EV-21	LAET		1	AMPT	
EVERETT1	EV-22	01/01/85	EV-22	LAET		1	AMPT	
EVERETT1	EV-24	01/01/85	EV-24	LAET		3	AMPT*	
EVERETT1	EV-25	01/01/85	EV-25	LAET		3	AMPT	
TPPS3AB	EB-30	03/15/82	1824	LAET		11		
TPPS3AB	EB-30	07/15/82	2081	LAET		9		
TPPS3AB	EB-31	03/15/82	1818	LAET		15		
TPPS3AB	EB-31	07/15/82	2078	LAET		2		
TPPS3AB	EB-32	03/15/82	1825	LAET		4		
TPPS3AB	EB-32	07/15/82	2077	LAET		6		
TPPS3AB	EB-33	03/15/82	1779	LAET		14		BENA*
TPPS3AB	EB-33	07/15/82	2080	LAET		11		BENA*
TPPS3AB	EB-34	03/15/82	1780	LAET		8		
TPPS3AB	EB-34	07/15/82	2071	LAET		15		
TPPS3AB	EB-35	03/15/82	1775	LAET		18		BENA*
TPPS3AB	EB-35	07/15/82	2079	LAET		22		BENA*

TPPS3AB	EB-36	03/15/82	1776	LAET	14	BENA*
TPPS3AB	EB-36	07/15/82	2072	LAET	9	BENA*
TPPS3AB	EB-37	03/15/82	1777	LAET	14	
TPPS3AB	EB-37	07/15/82	2073	LAET	17	
TPPS3AB	EB-38	03/15/82	1778	LAET	12	BENA
TPPS3AB	EB-38	07/15/82	2074	LAET	15	BENA*
TPPS3AB	EB-39	03/15/82	1814	LAET	16	
TPPS3AB	EB-39	07/15/82	2075	LAET	14	
TPPS3AB	WP-01	03/15/82	1806	LAET	5	
TPPS3AB	WP-01	07/15/82	2088	LAET	1	BENA
TPPS3AB	WP-02	03/15/82	1807	LAET	13	
TPPS3AB	WP-02	07/15/82	2089	LAET	11	BENA
TPPS3AB	WP-03	03/15/82	1788	LAET	1	
TPPS3AB	WP-03	07/15/82	2090	LAET	2	BENA*
TPPS3AB	WP-04	03/15/82	1809	LAET	3	
TPPS3AB	WP-04	07/15/82	2091	LAET	4	BENA
TPPS3AB	WP-05	03/15/82	1810	LAET	2	
TPPS3AB	WP-05	07/15/82	2092	LAET	3	BENA
TPPS3AB	WP-06	03/15/82	1811	LAET	4	
TPPS3AB	WP-06	07/15/82	2084	LAET	4	BENA
TPPS3AB	WP-07	03/15/82	1812	LAET	9	
TPPS3AB	WP-07	07/15/82	2093	LAET	10	BENA
TPPS3AB	WP-08	03/15/82	1813	LAET	5	
TPPS3AB	WP-08	07/15/82	2083	LAET	3	BENA*
TPPS3AB	WP-09	03/15/82	1815	LAET	4	
TPPS3AB	WP-09	07/15/82	2082	LAET	2	BENA
TPPS3AB	WP-10	03/15/82	1787	LAET	10	
TPPS3AB	WP-10	07/15/82	2076	LAET	7	BENA
TPPS3AB	WP-11	03/15/82	1789	LAET	20	
TPPS3AB	WP-11	07/15/82	2087	LAET	16	
TPPS3AB	WP-12	03/15/82	1786	LAET	7	BENA
TPPS3AB	WP-12	07/15/82	2069	LAET	4	BENA
TPPS3AB	WP-13	03/15/82	1784	LAET	3	BENA
TPPS3AB	WP-13	07/15/82	2070	LAET	5	BENA
TPPS3AB	WP-14	03/15/82	1785	LAET	2	BENA
TPPS3AB	WP-14	07/15/82	2085	LAET	7	
TPPS3AB	WP-15	03/15/82	1817	LAET	12	BENA
TPPS3AB	WP-15	07/15/82	2094	LAET	8	BENA
TPPS3AB	WP-16	03/15/82	1816	LAET	7	BENA*
TPPS3AB	WP-16	07/15/82	2086	LAET	2	BENA

* NEC = 1/10* HAET for acute bioassays (amphipod, oyster larvae, or Microtox) or the benthic AET, whichever is lower, and ≥90th percentile concentration for Puget Sound reference areas.

TABLE B-2. 1988 PUGET SOUND AET USED IN RELIABILITY TESTS*

Test types listed are:

- AMPT Amphipod toxicity AET.
- BENA Benthic infauna abundance AET.
- HAET Highest of AMPT, BENA, MICB, and OYST.
- LAET Lowest of AMPT, BENA, MICB, and OYST.
- MICB Microtox bioassay AET.
- OYST Oyster larvae toxicity AET.

*NOTE: This listing does not include preliminary AET (i.e., AET defined by "greater than" values because these AET are not used in determining sensitivity and efficiency.

== Group 88PSAET ==

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Value type: Original values, no modification or transformation.

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Chemical	AMPT	BENA	MICB	OYST
1-METHYLPYRENE		85.00	280.00	280.00
1,2-DICHLOROBENZENE		50.00	35.00	50.00
1,2-DICHLOROETHENE	0.80	0.80		
12-CHLORODEHYDROABIETIC ACID	61.00			
1,2,4-TRICHLOROBENZENE	51.00		31.00	64.00
1,4-DICHLOROBENZENE	120.00	110.00	110.00	120.00
HEXADECANOIC ACID	350.00	180.00		
HEXADECANOIC ACID, METHYL ESTER	1100.00	990.00		
HEXADECENOIC ACID METHYL ESTER	2200.00	2200.00		
1-METHYL PHENANTHRENE	1300.00	1300.00	370.00	370.00
2-CHLOROPHENOL	8.00	36.00		
2-METHYLNAPHTHALENE	1900.00	1400.00	670.00	670.00
2-METHYLPHENOL	63.00	72.00		63.00
2-METHYLPYRENE	410.00	510.00	510.00	410.00
2-METHOXYPHENOL	930.00	580.00	930.00	930.00
2,3,5 TRIMETHYL NAPTHALENE	54.00			
2,4-DICHLOROPHENOL	5.00	9.00		
2,4-DIMETHYL PHENOL	72.00	210.00	29.00	29.00
2,4,5-TRICHLOROPHENOL	29.00	3.00		
2,4,6-TRICHLOROPHENOL	25.00	6.00		
DIBENZO(A,H)ANTHRACENE	540.00	970.00	230.00	230.00
2-METHYL PHENANTHRENE	1500.00	1500.00	490.00	470.00
DI-N-OCTYL PHTHALATE		6200.00	25.00	
TRICHLOROETHENE	0.80			
3-METHYL PHENANTHRENE	1400.00	1400.00		
4-METHYL PHENOL	3600.00	1800.00	670.00	670.00
TETRACHLOROETHYLENE		57.00	140.00	140.00
PENTACHLOROCYCLOPENTANE		11.00	25.00	
PENTACHLOROPHENOL	360.00	690.00		
HEXACHLOROBENZENE	130.00	22.00	70.00	230.00
HEXACHLOROBUTADIENE	180.00	11.00	120.00	270.00

HEXACHLOROETHANE	140.00			140.00
ABIETIC ACID	450.00			
ACENAPHTHENE	2000.00	730.00	500.00	500.00
ACENAPHTHYLENE	1300.00	1300.00		
ALDRIN (PESTICIDE)		0.44		
UNIDENTIFIED ALKANOL	580.00	350.00		
ANTHRACENE	13000.00	4400.00	960.00	960.00
ANTIMONY (SB)	200.00	150.00	26.00	26.00
ARSENIC	93.00	57.00	700.00	700.00
BIS(2-ETHYLHEXYL)PHTHALATE		1300.00	1900.00	1900.00
BENZO(A)ANTHRACENE	5100.00	5100.00	1300.00	1600.00
BENZO(A)PYRENE	3000.00	3600.00	1600.00	1600.00
BARIUM	48.00	50.00		
BENZO(B)FLUORANTHENE	3500.00	4900.00		
BENZOIC ACID	760.00	650.00	650.00	650.00
BENZYL ALCOHOL	870.00	870.00	57.00	73.00
BERYLLIUM			0.36	
BENZO(G,H,I)PERYLENE	1400.00	2600.00	670.00	720.00
BIPHENYL	310.00	300.00	270.00	260.00
BENZO(K)FLUORANTHENE	4300.00	5000.00		
BASE PEAK M/Z 181, ISOMER 1	380.00	160.00		
BASE PEAK M/Z 181,ISOMER 2	200.00	87.00		
BUTYL BENZYL PHTHALATE	900.00	900.00	63.00	
CADMIUM	6.70	5.10	9.60	9.60
CAMPESTEROL	73.00			
CARBAZOLE	3600.00	970.00		
CHOLESTEROL (CHOLEST-5-EN-3[BETA]-OL)	260.00	160.00		
CHROMIUM TOTAL	270.00	260.00	27.00	
CHRYSENE	9200.00	9200.00	1400.00	2800.00
TOTAL CHLORINATED BENZENES	90.00			
COPPER	1300.00	530.00	390.00	390.00
COPROSTANOL		160.00	140.00	
CYMENE (UNSPECIFIED ISOMER)	2800.00	600.00	1100.00	2300.00
DIETHYL PHTHALATE		200.00		
DEHYDROABIETIC ACID	640.00	150.00		
DIBENZOFURAN	1700.00	700.00	540.00	540.00
DIBENZOTHIOPHENE	950.00	570.00	250.00	240.00
DI-N-BUTYL PHTHALATE	1400.00		1400.00	1400.00
DIMETHYL PHTHALATE			71.00	160.00
DITERPENOID HYDRCARBN (DEHYDROABIETANE?)	200.00	77.00		
DITERPENOID ALCOHOL (TOTAROL?)	130.00	10.00		
ETHYLBENZENE		10.00	33.00	37.00
FLUORANTHENE	30000.00	24000.00	1700.00	2500.00
FLUORENE	3600.00	1000.00	540.00	540.00
POLYCYCLIC AROMATIC HYDROCARBON-HEAVY	69000.00	69000.00	12000.00	17000.00
INDENO(1,2,3-CD)PYRENE	1800.00	2600.00	600.00	690.00
ISOPIMARADIENE	1500.00	1500.00	1400.00	1500.00
ISOPIMARIC ACID	170.00			
KAUR-16-ENE	2000.00	2000.00	2000.00	2100.00
LEAD	660.00	450.00	530.00	660.00
POLYCYCLIC AROMATIC HYDROCARBON-LIGHT	24000.00	13000.00	5200.00	5200.00
MERCURY	2.10	2.10	0.41	0.59
NAPHTHALENE	2400.00	2700.00	2100.00	2100.00
NICKEL				
TOTAL NITROGEN (NO3+NO3+NH4)	0.29	0.28	0.28	0.28
N-NITROSO DIPHENYLAMINE	48.00	28.00	40.00	130.00

OIL AND GREASE	4500.00	4600.00	1100.00	4300.00
POLYCHLORINATED BIPHENYLS	3100.00	1000.00	130.00	1100.00
PENTACHLOROBUTADIENE	2800.00	95.00	770.00	
PHENANTHRENE	6900.00	5400.00	1500.00	1500.00
PHENOL	1200.00	1200.00	1200.00	420.00
P,P'-DDD	43.00	16.00		
P,P'-DDE	15.00	9.00		
P,P'-DDT		34.00		
PYRENE	16000.00	16000.00	2600.00	3300.00
RETENE	1700.00	2000.00		
SELENIUM	1.00			
SILVER	6.10			
SULFIDES	540.00	630.00	45.00	45.00
TOTAL BENZOFLUORANTHENES (B + K)	7800.00	9900.00	3200.00	3600.00
TETRACHLOROBUTADIENE		380.00	3100.00	
TETRACHLOROGUAIACOL		4.00		
THALLIUM	0.40	0.24	0.24	0.24
TOTAL ORGANIC CARBON	15.10	15.10	15.10	15.10
TOTAL PHTHALATES		3300.00		
TRICHLOROBUTADIENE		820.00	5800.00	
1,2,4-TRITHIOLANE	82.00	4.30		
TOTAL VOLATILE SOLIDS	26.93	22.20	22.20	22.20
TOTAL XYLENE		40.00	100.00	120.00
ZINC	960.00	410.00	1600.00	1600.00

TABLE B-3. 1988 PUGET SOUND AET FOR SELECTED CHEMICALS (normalized to total organic carbon)[a]

Chemical	Amphipod AET[b]	Oyster AET[c]	Benthic AET[d]	Microtox AET[e]
Nonionic Organic Compounds (mg/kg organic carbon; ppm)				
Low molecular weight PAH	2,200	370	780	>530
Naphthalene	220	99	170	>170
Acenaphthylene	66	>27	66	>27
Acenaphthene	200	16	57	>57
Fluorene	360	23	79	>71
Phenanthrene	690	120	480	>160
Anthracene	1,200	>79	220	>79
2-Methylnaphthalene	>120	---	64	---
High molecular weight PAH	5,300	960	7,600	1,500
Fluoranthene	3,000	160	1,200	>190
Pyrene	1,000	>210	1,400	>210
Benz(a)anthracene	270	110	650	>160
Chrysene	460	110	850	>200
Benzofluoranthenes	450	230	1,500	>430
Benzo(a)pyrene	210	99	>1,300	>140
Indeno(1,2,3-c,d)pyrene	88	33	900	>87
Dibenzo(a,h)anthracene	47	120	89	33
Benzo(g,h,i)perylene	78	31	>1,200	>67
Chlorinated benzenes				
1,3-Dichlorobenzene	>15	>15	>15	>15
1,4-Dichlorobenzene	9	3.1	16	>16
1,2-Dichlorobenzene	>5.8	2.3	2.3	2.3
1,2,4-Trichlorobenzene	1.8	2.7	---	0.81
Hexachlorobenzene (HCB)	4.5	9.6	0.38	2.3
Total PCBs	190	>46	65	12
Phthalates				
Dimethyl phthalate	53	>22	53	>19
Diethyl phthalate	>110	>5.3	61	>5.3
Di-n-butyl phthalate	260	260	1,700	220
Butyl benzyl phthalate	42	>9.2	64	4.9
Bis(2-ethylhexyl)phthalate	78	60	60	47
Di-n-octyl phthalate	58	>57	4,500	--

TABLE B-3. (Continued)

Chemical	Amphipod AET[b]	Oyster AET[c]	Benthic AET[d]	Microtox AET[e]
Miscellaneous Extractables				
Dibenzofuran	>170	15	58	>58
Hexachlorobutadiene	6.2	11	6.9	3.9
N-nitrosodiphenylamine	>11	>11	11	>11
Volatile Organics				
Tetrachloroethene	>22	>22	>22	>22
Ethylbenzene	>3.8	>3.8	>3.8	>3.8
Total xylenes	>12	>12	>12	>12
Pesticides				
p,p'-DDE	0.81	--	0.31	--
p,p'-DDD	2.2	--	1.0	--
p,p'-DDT	>16	--	3.7	--
Ionizable Organic Compounds (mg/kg organic carbon; ppm)				
Phenols and Miscellaneous Extractables				
Phenol	440	>39	>140	33
2-Methylphenol	3.1	3.1	10	>10
4-Methylphenol	780	37	250	81
2,4-Dimethyl phenol	6.5	>1.3	2.6	0.63
Pentachlorophenol	24	>11	66	>11
Benzyl alcohol	73	5.0	>73	5.0
Benzoic acid	>170	>170	>170	>170
Metals (mg/kg organic carbon; ppm)				
Antimony	>55,000	3,300	5,500	3,300
Arsenic	32,000	88,000	4,400	88,000
Cadmium	1,100	1,200	580	1,200
Chromium	>150,000	---	65,000	---
Copper	100,000	49,000	13,000	48,000
Lead	110,000	66,000	18,000	66,000
Mercury	210	210	120	77
Nickel	>41,000	---	31,000	---
Silver	170	>100	490	100
Zinc	210,000	>200,000	48,000	>200,000

TABLE B-3. (Continued)

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[a] ">" indicates that a defined AET could not be established because there were no "effects" stations with chemical concentrations above the highest concentration among "no effects" stations (normalized to TOC). "--" indicates AET data not available.

[b] Based on 287 stations (including recent surveys in Eagle Harbor, Elliott Bay, and Everett Harbor not included in the previous generation of 1986 AET).

[c] Based on 56 stations (all from Commencement Bay Remedial Investigation and Blair Waterway dredging study); no additional stations added since 1986.

[d] Based on 201 stations (including recent surveys in Eagle Harbor, Elliott Bay, and Everett Harbor not included in the previous generation of 1986 AET).

[e] Based on 50 stations (all from Commencement Bay Remedial Investigation); no additional stations added since 1986.

TABLE B-4. ALTERNATIVE SEDIMENT QUALITY CRITERIA VALUES (SLC, TRIAD, AND EQUILIBRIUM PARTITIONING APPROACH)

Chemical	Screening Level Concentrations	Sediment-Water Equilibrium Partitioning	Triad Approach
Acenapthalene	4.74 (2)		
Aniline		0.248 (3)	
Anthracene	16.3 (2)		
Benzo(a)pyrene	39.7 (2)		
Benz(a)anthracene	26.1 (2)		
Chlorpyrifos		0.44 (3)	
Chrysene	38.4 (2)		
DDT	50.5 (2)		
Dieldrin		5.77 (3)	
Endrin		0.215 (3)	
Fluoranthene	64.4 (2)		
Fluorene	10.1 (2)		
Heptachlor		0.104 (3*)	
Naphthalene	41.4 (2)		
PAH	7.6 (1)		3.8 (1)
Pb			50 (1)
PCBs	3.66 (2)		
PCBs	0.06 (1)		0.1 (1)
PCB (1254)		41.8 (3*)	
Phenanthrene	36.8 (2)	102 (3)	
Pyrene	66.5 (2)		

(1) Chapman et al. 1987. Values are in ug/g sediment dry weight.
(2) Neff et al. 1988. Values are in ug/g sediment organic carbon.
(3) "Interim Sediment Criteria Values for Nonpolar Hydrophobic Organic Contaminants" (EPA 1988b). Saltwater Final Chronic Value unless otherwise noted. Values are in ug/g sediment organic carbon.
* Value is a Saltwater Final Residual Value.

TABLE B-5. BIOLOGICAL EFFECTS STATIONS USED IN RELIABILITY TESTS; SUCH STATIONS PROVIDE A BASIS FOR ESTABLISHING AN INVENTORY OF SEDIMENTS UNDER ELEMENT P-2 WHICH WILL BE MAPPED IN A FORTHCOMING REPORT

The biological effects listed are:

AMPT Amphipod toxicity AET.
BENA Benthic infauna abundance AET.
MICB Microtox bioassay AET.
OYST Oyster larvae toxicity AET.

Survey	Station	Date	Sample	AMPT	BENA	MICB	OYST
ALKI	AP-01	05/25/84	AP-01		NO HITa		
ALKI	AP-02	05/25/84	AP-02		NO HIT		
ALKI	AP-03	05/25/84	AP-03		NO HIT		
ALKI	AP-04	05/25/84	AP-04		HIT		
ALKI	AP-05	05/25/84	AP-05		NO HIT		
ALKI	AP-06	05/25/84	AP-06		NO HIT		
ALKI	AP-07	05/25/84	AP-07		NO HIT		
ALKI	PW-01	05/26/84	PW-01		NO HIT		
ALKI	PW-02	05/26/84	PW-02		NO HIT		
ALKI	PW-03	05/26/84	PW-03		NO HIT		
ALKI	PW-04	05/26/84	PW-04		NO HIT		

*Benthic infauna abundance was the only biological effect tested in the Alki survey.

CBBLAIR	B03	06/01/84	B03	NO HIT	NO HIT		NO HIT
CBBLAIR	B04	06/01/84	B04	NO HIT	NO HIT		NO HIT
CBBLAIR	B09	06/01/84	B09	NO HITa	NO HIT		NO HIT
CBBLAIR	B10	06/01/84	B10	NO HITa	NO HIT		HIT
CBBLAIR	B12	06/01/84	B12	NO HITa	NO HIT		NO HIT
CBBLAIR	B15	06/01/84	B15	HIT	NO HIT		NO HIT

*Benthic infauna abundance, and the amphipod and oyster larvae toxicity tests were the only biological effects tested in the CBBLAIR survey.

CBMSQS	BL-11	01/01/84	BL-11	NO HIT	NO HIT	NO HIT	NO HIT
CBMSQS	BL-13	01/01/84	BL-13	NO HITa	NO HIT	HIT	NO HIT
CBMSQS	BL-21	01/01/84	BL-21	NO HIT	NO HIT	NO HIT	NO HIT
CBMSQS	BL-25	01/01/84	BL-25	HIT	NO HIT	NO HIT	NO HIT
CBMSQS	BL-28	01/01/84	BL-28	NO HIT	NO HIT	NO HIT	NO HIT
CBMSQS	BL-31	01/01/84	BL-31	NO HITa	NO HIT	HIT	NO HIT
CBMSQS	CI-11	01/01/84	CI-11	HIT	HIT	HIT	HIT
CBMSQS	CI-13	01/01/84	CI-13	NO HITa	HIT	HIT	HIT
CBMSQS	CI-16	01/01/84	CI-16	b	HIT	HIT	HIT
CBMSQS	CI-17	01/01/84	CI-17	NO HITa	HIT	HIT	NO HIT
CBMSQS	CI-20	01/01/84	CI-20	HIT	NO HITa	NO HIT	HIT
CBMSQS	CI-22	01/01/84	CI-22	NO HITa	NO HIT	NO HIT	NO HIT
CBMSQS	CR-11	01/01/84	CR-11	NO HIT	NO HIT	NO HIT	NO HIT
CBMSQS	CR-12	01/01/84	CR-12	NO HIT	NO HIT	NO HIT	NO HIT
CBMSQS	CR-13	01/01/84	CR-13	NO HIT	NO HIT	NO HIT	NO HIT
CBMSQS	CR-14	01/01/84	CR-14	NO HIT	NO HIT	NO HIT	NO HIT
CBMSQS	HY-12	01/01/84	HY-12	b	NO HIT	HIT	HIT
CBMSQS	HY-14	01/01/84	HY-14	NO HIT	HIT	HIT	NO HIT

CBMSQS	HY-17	01/01/84	HY-17	NO HIT	HIT	HIT	HIT
CBMSQS	HY-22	01/01/84	HY-22	HIT	HIT	HIT	HIT
CBMSQS	HY-23	01/01/84	HY-23	HIT	HIT	HIT	HIT
CBMSQS	HY-24	01/01/84	HY-24	NO HITa	HIT	HIT	NO HIT
CBMSQS	HY-28	01/01/84	HY-28	NO HIT	HIT	NO HIT	NO HIT
CBMSQS	HY-32	01/01/84	HY-32	NO HITa	HIT	NO HIT	NO HIT
CBMSQS	HY-37	01/01/84	HY-37	NO HITa	HIT	HIT	NO HIT
CBMSQS	HY-42	01/01/84	HY-42	HIT	HIT	HIT	NO HIT
CBMSQS	HY-43	01/01/84	HY-43	NO HITa	HIT	HIT	NO HIT
CBMSQS	HY-44	01/01/84	HY-44	NO HITa	NO HIT	NO HIT	NO HIT
CBMSQS	HY-47	01/01/84	HY-47	HIT	HIT	HIT	HIT
CBMSQS	HY-50	01/01/84	HY-50	NO HITa	NO HIT	HIT	NO HIT
CBMSQS	MD-12	01/01/84	MD-12	NO HIT	HIT	NO HIT	NO HIT
CBMSQS	MI-11	01/01/84	MI-11	HIT	NO HIT	NO HIT	NO HIT
CBMSQS	MI-13	01/01/84	MI-13	NO HIT	NO HIT	HIT	NO HIT
CBMSQS	MI-15	01/01/84	MI-15	HIT	NO HIT	HIT	NO HIT
CBMSQS	RS-12	01/01/84	RS-12	NO HIT	NO HIT	NO HIT	NO HIT
CBMSQS	RS-13	01/01/84	RS-13	HIT	NO HIT	NO HIT	HIT
CBMSQS	RS-14	01/01/84	RS-14	NO HITa	NO HITa	NO HIT	NO HIT
CBMSQS	RS-18	01/01/84	RS-18	HIT	HIT	HIT	HIT
CBMSQS	RS-19	01/01/84	RS-19	HIT	HIT	HIT	HIT
CBMSQS	RS-20	01/01/84	RS-20	NO HIT	HIT	HIT	NO HIT
CBMSQS	RS-22	01/01/84	RS-22	NO HIT		NO HIT	NO HIT
CBMSQS	RS-24	01/01/84	RS-24	HIT		NO HIT	NO HIT
CBMSQS	SI-11	01/01/84	SI-11	NO HITa	HIT	HIT	NO HIT
CBMSQS	SI-12	01/01/84	SI-12	HIT	HIT	HIT	NO HIT
CBMSQS	SI-15	01/01/84	SI-15	HIT	NO HITa	NO HIT	NO HIT
CBMSQS	SP-11	01/01/84	SP-11	NO HIT	HIT	HIT	NO HIT
CBMSQS	SP-12	01/01/84	SP-12	NO HITa	NO HIT	HIT	HIT
CBMSQS	SP-14	01/01/84	SP-14	HIT	HIT	HIT	HIT
CBMSQS	SP-15	01/01/84	SP-15	HIT	HIT	HIT	HIT
CBMSQS	SP-16	01/01/84	SP-16	HIT	HIT	HIT	HIT
DUWRIV1	DR-01	04/15/85	DR-01	NO HIT			
DUWRIV1	DR-02	04/15/85	DR-02	NO HIT			
DUWRIV1	DR-03	04/15/85	DR-03	NO HIT			
DUWRIV1	DR-04	04/15/85	DR-04	NO HIT			
DUWRIV1	DR-05	04/15/85	DR-05	NO HIT			
DUWRIV1	DR-06	04/15/85	DR-06	NO HIT			
DUWRIV1	DR-07	04/15/85	DR-07	HIT			
DUWRIV1	DR-08	04/15/85	DR-08	HIT			
DUWRIV1	SQ-09	04/19/85	SQ-09	NO HIT			
DUWRIV2	DR-10	07/01/85	CA1	HIT			
DUWRIV2	DR-11	07/01/85	CA2	HIT			
DUWRIV2	DR-12	07/01/85	CA3	NO HIT			
DUWRIV2	DR-13	07/01/85	CB1	NO HIT			
DUWRIV2	DR-14	07/01/85	CB2	NO HIT			
DUWRIV2	DR-15	07/01/85	CB3	NO HIT			
DUWRIV2	DR-16	07/01/85	CB4	HIT			
DUWRIV2	DR-17	07/01/85	CB5	NO HIT			
DUWRIV2	DR-18	07/01/85	CC1	NO HIT			
DUWRIV2	DR-19	07/01/85	CC2	NO HIT			
DUWRIV2	DR-20	07/01/85	CC3	NO HIT			
DUWRIV2	DR-21	07/01/85	CC4	NO HIT			
DUWRIV2	DR-22	07/01/85	CC5	NO HIT			

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DUWRIV2   DR-23     07/01/85  CD1             NO HIT
DUWRIV2   DR-24     07/01/85  CD2             NO HIT
DUWRIV2   DR-25     07/01/85  CE1             HIT
DUWRIV2   DR-26     07/01/85  CE2             HIT
DUWRIV2   DR-27     07/01/85  CE3             HIT
DUWRIV2   DR-28     07/01/85  CF1             NO HIT
DUWRIV2   DR-29     07/01/85  CF2             HIT
DUWRIV2   DR-30     07/01/85  CF3             NO HIT
DUWRIV2   DR-31     07/01/85  CF4             NO HIT
DUWRIV2   DR-32     07/01/85  CF5             NO HIT
DUWRIV2   DR-33     07/01/85  CG1             NO HIT
DUWRIV2   DR-34     07/01/85  CG2             NO HIT
DUWRIV2   DR-35     07/01/85  CG3             NO HITa
DUWRIV2   DR-36     07/01/85  CG4             NO HIT
DUWRIV2   DR-37     07/01/85  CG5             NO HIT
DUWRIV2   DR-38     07/01/85  CH1             NO HIT
DUWRIV2   DR-39     07/01/85  CW/A1           NO HIT
DUWRIV2   SQ-21     07/01/85  SEQUIM          NO HIT

    *Amphipod toxicity was the only biological effect tested in the DUWRIV1 and DUWRIV2 surveys.

EBCHEM    AB-01     09/26/85  AB-01           HIT          b
EBCHEM    AB-02     09/26/85  AB-02           NO HIT    NO HIT
EBCHEM    AB-03     09/26/85  AB-03           NO HIT    HIT
EBCHEM    AB-04     09/26/85  AB-04           NO HIT    NO HIT
EBCHEM    DR-01     09/30/85  DR-01           NO HIT
EBCHEM    DR-02     09/30/85  DR-02           HIT
EBCHEM    DR-03     09/30/85  DR-03           NO HIT
EBCHEM    DR-04     09/30/85  DR-04           NO HIT
Amphipod toxicity was the only biological effect tested in the DUWRIV1 and DUWRIV2 surveys.

EBCHEM    AB-01     09/26/85  AB-01           HIT          b
EBCHEM    AB-02     09/26/85  AB-02           NO HIT    NO HIT
EBCHEM    AB-03     09/26/85  AB-03           NO HIT    HIT
EBCHEM    AB-04     09/26/85  AB-04           NO HIT    NO HIT
EBCHEM    DR-01     09/30/85  DR-01           NO HIT
EBCHEM    DR-02     09/30/85  DR-02           HIT
EBCHEM    DR-03     09/30/85  DR-03           NO HIT
Amphipod toxicity was the only biological effect tested in the DUWRIV1 and DUWRIV2 surveys.

EBCHEM    AB-01     09/26/85  AB-01           HIT          b
EBCHEM    AB-02     09/26/85  AB-02           NO HIT    NO HIT
EBCHEM    AB-03     09/26/85  AB-03           NO HIT    HIT
EBCHEM    AB-04     09/26/85  AB-04           NO HIT    NO HIT
EBCHEM    DR-01     09/30/85  DR-01           NO HIT
EBCHEM    DR-02     09/30/85  DR-02           HIT
EBCHEM    DR-03     09/30/85  DR-03           NO HIT
Amphipod toxicity was the only biological effect tested in the DUWRIV1 and DUWRIV2 surveys.

EBCHEM    AB-01     09/26/85  AB-01           HIT          b
EBCHEM    AB-02     09/26/85  AB-02           NO HIT    NO HIT
EBCHEM    AB-03     09/26/85  AB-03           NO HIT    HIT
EBCHEM    AB-04     09/26/85  AB-04           NO HIT    NO HIT
EBCHEM    DR-01     09/30/85  DR-01           NO HIT
EBCHEM    DR-02     09/30/85  DR-02           HIT
EBCHEM    DR-03     09/30/85  DR-03           NO HIT
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EBCHEM	DR-04	09/30/85	DR-04	NO HIT	
EBCHEM	DR-05	09/30/85	DR-05	c	
EBCHEM	DR-06	10/09/85	DR-06	NO HIT	
EBCHEM	DR-07	09/30/85	DR-07	NO HIT	
EBCHEM	DR-08	09/30/85	DR-08	c	
EBCHEM	DR-09	09/30/85	DR-09	NO HIT	
EBCHEM	DR-10	09/30/85	DR-10	b	
EBCHEM	DR-11	09/30/85	DR-11	NO HIT	
EBCHEM	DR-12	09/30/85	DR-12	b	
EBCHEM	DR-13	09/30/85	DR-13	HIT	
EBCHEM	DR-14	09/30/85	DR-14	HIT	
EBCHEM	DR-15	09/30/85	DR-15	HIT	
EBCHEM	DR-16	09/30/85	DR-16	HIT	
EBCHEM	DR-17	09/30/85	DR-17	NO HIT	
EBCHEM	DR-25	10/10/85	DR-25	HIT	
EBCHEM	EW-01	10/09/85	EW-01	NO HIT	
EBCHEM	EW-02	10/04/85	EW-02	HIT	HIT
EBCHEM	EW-03	10/04/85	EW-03	c	NO HIT
EBCHEM	EW-04	10/14/85	EW-04	HIT	HIT
EBCHEM	EW-05	10/14/85	EW-05	HIT	HIT
EBCHEM	EW-05		EW-05	HIT	HIT
EBCHEM	EW-06	10/04/85	EW-06	HIT	HIT
EBCHEM	EW-07	10/14/85	EW-07	HIT	HIT
EBCHEM	EW-08	10/14/85	EW-08	HIT	HIT
EBCHEM	EW-09	10/14/85	EW-09	HIT	HIT
EBCHEM	EW-10	10/14/85	EW-10	HIT	NO HIT
EBCHEM	EW-11	10/14/85	EW-11	HIT	HIT
EBCHEM	EW-12	10/15/85	EW-12	NO HIT	NO HIT
EBCHEM	EW-13	10/15/85	EW-13	NO HIT	NO HIT
EBCHEM	EW-14	10/15/85	EW-14	NO HIT	NO HIT
EBCHEM	EW-15	10/15/85	EW-15	NO HIT	NO HIT
EBCHEM	EW-16	10/15/85	EW-16	HIT	NO HIT
EBCHEM	KG-01	09/25/85	KG-01	NO HIT	HIT
EBCHEM	KG-02	10/09/85	KG-02	HIT	
EBCHEM	KG-03	09/25/85	KG-03	HIT	NO HIT
EBCHEM	KG-04	10/09/85	KG-04	NO HIT	
EBCHEM	KG-05	09/30/85	KG-05	HIT	HIT
EBCHEM	KG-06	09/30/85	KG-06	NO HIT	HIT
EBCHEM	KG-07	09/30/85	KG-07	NO HIT	HIT
EBCHEM	KG-08	10/01/85	KG-08	NO HIT	HIT
EBCHEM	KG-09	10/01/85	KG-09	HIT	NO HIT
EBCHEM	KG-10	10/08/85	KG-10	HIT	
EBCHEM	KG-11	10/01/85	KG-11	HIT	HIT
EBCHEM	MG-01	09/26/85	MG-01	NO HIT	HIT
EBCHEM	MG-02	09/26/85	MG-02	NO HIT	HIT
EBCHEM	MG-03	09/26/85	MG-03	NO HIT	HIT
EBCHEM	MG-04	09/26/85	MG-04	NO HIT	HIT
EBCHEM	NH-01	10/15/85	NH-01	NO HIT	HIT
EBCHEM	NH-02	10/15/85	NH-02	HIT	HIT
EBCHEM	NH-03	10/16/85	NH-03	HIT	HIT
EBCHEM	NH-04	10/15/85	NH-04	HIT	HIT
EBCHEM	NH-05	10/15/85	NH-05	HIT	HIT
EBCHEM	NH-06	10/16/85	NH-06	HIT	HIT
EBCHEM	NH-07	10/09/85	NH-07	NO HIT	
EBCHEM	NH-08	10/16/85	NH-08	HIT	HIT
EBCHEM	NH-09	10/16/85	NH-09	HIT	NO HIT

EBCHEM	NH-10	10/08/85	NH-10	NO HIT	
EBCHEM	NH-11	10/15/85	NH-11	HIT	HIT
EBCHEM	NS-01	10/08/85	NS-01	HIT	
EBCHEM	NS-02	09/27/85	NS-02	NO HIT	NO HIT
EBCHEM	NS-03	10/04/85	NS-03	NO HIT	NO HIT
EBCHEM	NS-04	10/04/85	NS-04	c	
EBCHEM	NS-05	10/04/85	NS-05	NO HIT	NO HIT
EBCHEM	NS-06	09/27/85	NS-06	b	HIT
EBCHEM	NS-07	10/04/85	NS-07	HIT	NO HIT
EBCHEM	NS-08	09/26/85	NS-08	HIT	HIT
EBCHEM	PS-01	10/12/85	PS-01	NO HIT	NO HIT
EBCHEM	PS-01		PS-01	NO HIT	NO HIT
EBCHEM	PS-02	10/12/85	PS-02	NO HIT	NO HIT
EBCHEM	PS-03	10/12/85	PS-03	NO HIT	NO HIT
EBCHEM	PS-04	10/12/85	PS-04	NO HIT	NO HIT
EBCHEM	SS-01	10/16/85	SS-01	NO HIT	NO HIT
EBCHEM	SS-03	10/04/85	SS-03	HIT	HIT
EBCHEM	SS-04	10/04/85	SS-04	b	HIT
EBCHEM	SS-05	10/03/85	SS-05	NO HIT	NO HIT
EBCHEM	SS-05	10/03/85	SS-05	NO HIT	NO HIT
EBCHEM	SS-06	10/03/85	SS-06	HIT	HIT
EBCHEM	SS-07	10/03/85	SS-07	HIT	NO HIT
EBCHEM	SS-08	09/27/85	SS-08	HIT	HIT
EBCHEM	SS-09	09/27/85	SS-09	HIT	HIT
EBCHEM	SS-10	09/27/85	SS-10	b	b
EBCHEM	SS-11	09/27/85	SS-11	NO HIT	NO HIT
EBCHEM	SS-12	09/27/85	SS-12	NO HIT	NO HIT
EBCHEM	WW-01	10/01/85	WW-01	NO HIT	HIT
EBCHEM	WW-02	10/09/85	WW-02	HIT	
EBCHEM	WW-03	10/01/85	WW-03	NO HIT	HIT
EBCHEM	WW-04	10/01/85	WW-04	NO HIT	HIT
EBCHEM	WW-05	10/01/85	WW-05	NO HIT	HIT
EBCHEM	WW-06	10/01/85	WW-06	NO HIT	HIT
EBCHEM	WW-06	10/01/85	WW-06	NO HIT	HIT
EBCHEM	WW-08	10/01/85	WW-08	HIT	HIT
EBCHEM	WW-09	10/02/85	WW-09	HIT	HIT
EBCHEM	WW-10	10/02/85	WW-10	NO HIT	HIT
EBCHEM	WW-11	10/02/85	WW-11	HIT	HIT
EBCHEM	WW-12	10/02/85	WW-12	HIT	HIT
EBCHEM	WW-13	10/02/85	WW-13	NO HIT	NO HIT
EBCHEM	WW-14	10/02/85	WW-14	b	HIT
EBCHEM	WW-15	10/08/85	WW-15	NO HIT	
EBCHEM	WW-16	10/02/85	WW-16	NO HIT	HIT
EBCHEM	WW-17	10/03/85	WW-17	NO HIT	HIT
EBCHEM	WW-18	10/03/85	WW-18	NO HIT	HIT
EBCHEM	WW-19	10/03/85	WW-19	NO HIT	HIT
EBCHEM	WW-20	10/03/85	WW-20	NO HIT	HIT
EHCHEM	BH-01	06/01/85	B1	NO HIT	NO HIT
EHCHEM	BH-02	06/01/85	B1	NO HIT	NO HIT
EHCHEM	EH-01	06/01/85	B1	NO HIT	NO HIT
EHCHEM	EH-02	06/01/85	B1	NO HIT	HIT
EHCHEM	EH-03	06/01/85	V6	NO HIT	HIT
EHCHEM	EH-05	06/01/85	V1	NO HIT	HIT
EHCHEM	EH-06	06/01/85	V6	NO HIT	HIT
EHCHEM	EH-08	06/01/85	V1	HIT	HIT

EHCHEM	EH-15	06/01/85	V1	NO HIT	HIT
EHCHEM	EH-16	06/01/85	V1	NO HIT	HIT

*Amphipod toxicity and benthic infauna abundance were the only biological effects tested in the EBCHEM and EHCHEM surveys.

EIGHTBAY	BH-03	01/01/82	BH-03	NO HIT
EIGHTBAY	BH-04	01/01/82	BH-04	NO HIT
EIGHTBAY	BH-05	01/01/82	BH-05	HIT
EIGHTBAY	BH-07	01/01/82	BH-07	NO HIT
EIGHTBAY	BH-11	01/01/82	BH-11	NO HIT
EIGHTBAY	BH-12	01/01/82	BH-12	NO HIT
EIGHTBAY	BH-23	01/01/82	BH-23	HIT
EIGHTBAY	BH-24	01/01/82	BH-24	NO HIT
EIGHTBAY	CS-01	01/01/82	CS-01	HIT
EIGHTBAY	CS-11	01/01/82	CS-11	HIT
EIGHTBAY	CS-15	01/01/82	CS-15	HIT
EIGHTBAY	CS-17	01/01/82	CS-17	HIT
EIGHTBAY	DB-01	01/01/82	DB-01	NO HIT
EIGHTBAY	DB-05	01/01/82	DB-05	NO HIT
EIGHTBAY	DB-07	01/01/82	DB-07	HIT
EIGHTBAY	DB-15	01/01/82	DB-15	HIT
EIGHTBAY	EL-09	01/01/82	EL-09	NO HIT
EIGHTBAY	EL-10	01/01/82	EL-10	NO HIT
EIGHTBAY	EL-12	01/01/82	EL-12	NO HIT
EIGHTBAY	EL-17	01/01/82	EL-17	NO HIT
EIGHTBAY	EL-20	01/01/82	EL-20	NO HIT
EIGHTBAY	EL-22	01/01/82	EL-22	NO HIT
EIGHTBAY	EL-23	01/01/82	EL-23	NO HIT
EIGHTBAY	EL-24	01/01/82	EL-24	NO HIT
EIGHTBAY	EV-01	01/01/82	EV-01	HIT
EIGHTBAY	EV-02	01/01/82	EV-02	HIT
EIGHTBAY	EV-03	01/01/82	EV-03	HIT
EIGHTBAY	EV-04	01/01/82	EV-04	HIT
EIGHTBAY	EV-05	01/01/82	EV-05	HIT
EIGHTBAY	EV-06	01/01/82	EV-06	NO HIT
EIGHTBAY	EV-07	01/01/82	EV-07	NO HIT
EIGHTBAY	EV-11	01/01/82	EV-11	HIT
EIGHTBAY	SC-06	01/01/82	SC-06	HIT
EIGHTBAY	SC-07	01/01/82	SC-07	NO HIT
EIGHTBAY	SC-08	01/01/82	SC-08	HIT
EIGHTBAY	SC-14	01/01/82	SC-14	HIT
EIGHTBAY	SC-17	01/01/82	SC-17	HIT
EIGHTBAY	SC-18	01/01/82	SC-18	HIT
EIGHTBAY	SC-19	01/01/82	SC-19	NO HITa
EIGHTBAY	SC-20	01/01/82	SC-20	HIT
EIGHTBAY	SM-01	01/01/82	SM-01	HIT
EIGHTBAY	SM-03	01/01/82	SM-03	HIT
EIGHTBAY	SM-07	01/01/82	SM-07	HIT
EIGHTBAY	SM-20	01/01/82	SM-20	HIT
EIGHTBAY	SQ-14	01/01/82	SQ-14	NO HIT
EIGHTBAY	SQ-17	01/01/82	SQ-17	NO HIT
EIGHTBAY	SQ-18	01/01/82	SQ-18	NO HIT
EIGHTBAY	SQ-20	01/01/82	SQ-20	NO HIT

*Amphipod toxicity was the only biological effect tested in the EIGHTBAY survey.

EVCHEM	ES-01	10/06/86	ES-01G	NO HIT	
EVCHEM	ES-02	10/06/86	ES-02G	NO HIT	
EVCHEM	ES-03	10/06/86	ES-03G	NO HIT	
EVCHEM	EW-01	10/07/86	EW-01G	HIT	HIT
EVCHEM	EW-04	09/30/86	EW-04G	HIT	HIT
EVCHEM	EW-07	09/30/86	EW-07G	HIT	HIT
EVCHEM	EW-10	10/01/86	EW-10G	HIT	HIT
EVCHEM	EW-12	10/01/86	EW-12G	NO HIT	HIT
EVCHEM	EW-14	10/01/86	EW-14G	c	HIT
EVCHEM	NG-01	10/02/86	NG-01G	NO HIT	NO HIT
EVCHEM	NG-02	10/02/86	NG-02G	NO HIT	HIT
EVCHEM	NG-03	10/02/86	NG-03G	NO HIT	HIT
EVCHEM	NG-04	10/02/86	NG-04G	HIT	HIT
EVCHEM	NG-06	10/08/86	NG-06G	HIT	NO HIT
EVCHEM	NG-10	10/03/86	NG-10G	NO HIT	NO HIT
EVCHEM	NG-12	10/15/86	NG-12G	NO HIT	
EVCHEM	NG-13	10/15/86	NG-13G	NO HIT	
EVCHEM	NG-14	10/15/86	NG-14G	NO HIT	
EVCHEM	NG-15	10/15/86	NG-15G	NO HIT	
EVCHEM	OG-03	10/09/86	OG-03G	HIT	
EVCHEM	PS-02	10/13/86	PS-02G	NO HIT	NO HIT
EVCHEM	PS-03	10/13/86	PS-03G	NO HIT	NO HIT
EVCHEM	PS-04	10/10/86	PS-04G	NO HIT	NO HIT
EVCHEM	SD-01	10/09/86	SD-01G	NO HIT	HIT
EVCHEM	SD-02	10/07/86	SD-02G	c	NO HIT
EVCHEM	SR-01	10/06/86	SR-01G	NO HIT	
EVCHEM	SR-02	10/06/86	SR-02G	NO HIT	
EVCHEM	SR-04	10/06/86	SR-04G	NO HIT	
EVCHEM	SR-07	10/03/86	SR-07G	c	HIT
EVCHEM	SR-08	10/03/86	SR-08G	NO HIT	HIT
EVCHEM	SS-01	10/06/86	SS-01G	NO HIT	

*Amphipod toxicity and benthic infauna abundance were the only biological effects tested in the EVCHEM survey.

EVERETT1	EV-20	01/01/85	EV-20	NO HIT
EVERETT1	EV-21	01/01/85	EV-21	NO HIT
EVERETT1	EV-22	01/01/85	EV-22	NO HIT
EVERETT1	EV-23	01/01/85	EV-23	NO HIT
EVERETT1	EV-24	01/01/85	EV-24	HIT
EVERETT1	EV-25	01/01/85	EV-25	NO HIT

*Amphipod toxicity was the only biological effect tested in the EVERETT1 survey.

TPPS3AB	EB-33	03/15/82	1779	HIT
TPPS3AB	EB-33	07/15/82	2080	HIT
TPPS3AB	EB-35	03/15/82	1775	HIT
TPPS3AB	EB-35	07/15/82	2079	HIT
TPPS3AB	EB-36	03/15/82	1776	HIT
TPPS3AB	EB-36	07/15/82	2072	HIT
TPPS3AB	EB-38	03/15/82	1778	HIT
TPPS3AB	EB-38	07/15/82	2074	HIT
TPPS3AB	WP-01	07/15/82	2088	NO HIT
TPPS3AB	WP-02	07/15/82	2089	NO HIT
TPPS3AB	WP-03	07/15/82	2090	HIT

TPPS3AB	WP-04	07/15/82	2091	NO HIT
TPPS3AB	WP-05	07/15/82	2092	NO HIT
TPPS3AB	WP-06	07/15/82	2084	NO HIT
TPPS3AB	WP-07	07/15/82	2093	NO HIT
TPPS3AB	WP-08	07/15/82	2083	HIT
TPPS3AB	WP-09	07/15/82	2082	NO HIT
TPPS3AB	WP-10	07/15/82	2076	NO HIT
TPPS3AB	WP-11	03/15/82	1789	b
TPPS3AB	WP-12	03/15/82	1786	NO HIT
TPPS3AB	WP-12	07/15/82	2069	NO HIT
TPPS3AB	WP-13	03/15/82	1784	NO HIT
TPPS3AB	WP-13	07/15/82	2070	NO HIT
TPPS3AB	WP-14	03/15/82	1785	NO HIT
TPPS3AB	WP-14	07/15/82	2085	b
TPPS3AB	WP-15	03/15/82	1817	NO HIT
TPPS3AB	WP-15	07/15/82	2094	NO HIT
TPPS3AB	WP-16	03/15/82	1816	HIT
TPPS3AB	WP-16	07/15/82	2086	NO HITa

*Benthic infauna abundance was the only biological effect tested in the TPPS3AB survey.

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a Although the indicated biological effect was statistically significant ($P<0.05$) it is not classified as a "hit" because mortality was <25% for AMPT or abundance depression was <50% for BENA.

b The indicated biological effect was excluded because of a chemical anomaly.

c The indicated biological effect was excluded as inconclusive because of inadequate statistical power.

TABLE B-6. SUMMARY OF DATA SETS USED TO EVALUATE PUGET SOUND AET

Embayment	Survey Code[a]	Number/Kind of Bioeffect Samples[b]	Chemical Analyses Conducted[c]							
			Acid	Base	Neut.	PCB	Pest.	VOA	Metal	Misc
Bellingham Bay	EIGHTBAY	8 / A	x	x	x	x	x	x	x	
Carr Inlet	CBMSQS	4 / BAOM	x	x	x	x	x	x	x	x
Case Inlet	EIGHTBAY	4 / A	x	x	x	x	x	x	x	
Central Puget Sound Basin	ALKI	4 / B	x	x	x	x	x	x	x	
	EHCHEM	2 / BA	x	x	x	x	x	x	x	x
Commencement Bay	CBBLAIR	6 / BAO	x	x	x	x	x	x	x	x
	CBMSQS	42 / BAOM	x	x	x	x	x	x	x	x
		2 / AOM								
		2 / B OM								
Dabob Bay	EIGHTBAY	4 / A	x	x	x	x	x	x	x	
Eagle Harbor	EHCHEM	8 / BA	x	x	x	x	x	x	x	x
Elliott Bay	EBCHEM	71 / BA	x	x	x	x	x	x	x	x
		24 / A								
		4 / B								
	ALKI	7 / B	x	x	x	x	x	x	x	
	TPPS3AB	27 / B	x	x	x	x	x	x	x	
	DUWRIV1	8 / A			x	x	x	x	x	
	DUWRIV2	30 / A			x	x	x	x	x	
	EIGHTBAY	8 / A	x	x	x	x	x	x	x	
Everett Harbor	EVCHEM	13 / BA	x	x	x	x	x	x	x	x
		13 / A								
		3 / B								
	EVERETT1	6 / A		x	x		x	x		
	EIGHTBAY	8 / A	x	x	x	x	x	x	x	
Port Susan	EBCHEM	5 / BA	x	x	x	x	x	x	x	x
	EVCHEM	3 / BA	x	x	x	x	x	x	x	x
Samish Bay	EIGHTBAY	4 / A	x	x	x	x	x	x	x	
Sequim Bay	EIGHTBAY	4 / A	x	x	x	x	x	x	x	
	DUWRIV1	1 / A			x	x	x	x	x	
	DUWRIV2	1 / A			x	x	x	x	x	
Sinclair Inlet	EIGHTBAY	8 / A	x	x	x	x	x	x	x	

TABLE B-6. (Continued)

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[a] Puget Sound samples are derived from multiple surveys, which provided data for varying numbers of chemicals and biological indicators. The surveys include:

ALKI	Metro survey of Alki Point, Seattle (Osborn et al. 1985; Trial and Michaud 1985)
CBBLAIR	Port of Tacoma dredging survey of Blair Waterway in Commencement Bay (see Barrick et al. 1985)
CBMSQS	Commencement Bay Nearshore/Tideflats Superfund project; Carr Inlet reference area (Barrick et al. 1985)
DUWRIV1	PSDDA dredging study in the Duwamish River, Seattle (Phase I); Sequim Bay reference area (Chan et al. 1985)
DUWRIV2	PSDDA dredging study in the Duwamish River, Seattle (Phase II); Sequim Bay reference area (Chan et al. 1986)
EBCHEM	PSEP survey of Elliott Bay; Port Susan reference area (Beller et al. 1988)
EHCHEM	Ecology preliminary investigation of Eagle Harbor; Blakely Harbor reference area in Central Puget Sound (Barrick et al. 1986)
EIGHTBAY	EPA survey of eight urban and nonurban embayments in Puget Sound (Battelle 1985)
EVCHEM	PSEP survey of Everett Harbor; Port Susan reference area (Pastorok et al. 1988)
EVERETT1	U.S. Navy preliminary dredging study in Everett Harbor (U.S. Navy 1985)
TPPS3AB	Toxic Pretreatment Planning Study conducted in Central Puget Sound and Elliott Bay by Metro (Romberg et al. 1984).

Station locations for each survey are summarized in Appendix B of Barrick et al. (1988).

[b] 334 distinct samples (including 12 repeated samplings) at a total of 322 locations: (B) 201 benthic infaunal analyses; (A) 287 amphipod mortality bioassays; (O) 56 oyster larvae abnormality bioassays; (M) 50 Microtox (saline extract) bioassays. The seven amphipod bioassay stations excluded as biological anomalies and the three benthic infauna and eight amphipod bioassay stations excluded as chemical anomolies (see text) are <u>not</u> included in these totals.

[c] Chemical analyses conducted for EPA priority pollutant acid, base, neutral, PCB, pesticide, and volatile organic compounds, metals, and miscellaneous compounds not recognized as EPA priority pollutants (e.g., resin acid compound data for the EVCHEM survey, and tentatively identified organic compounds).

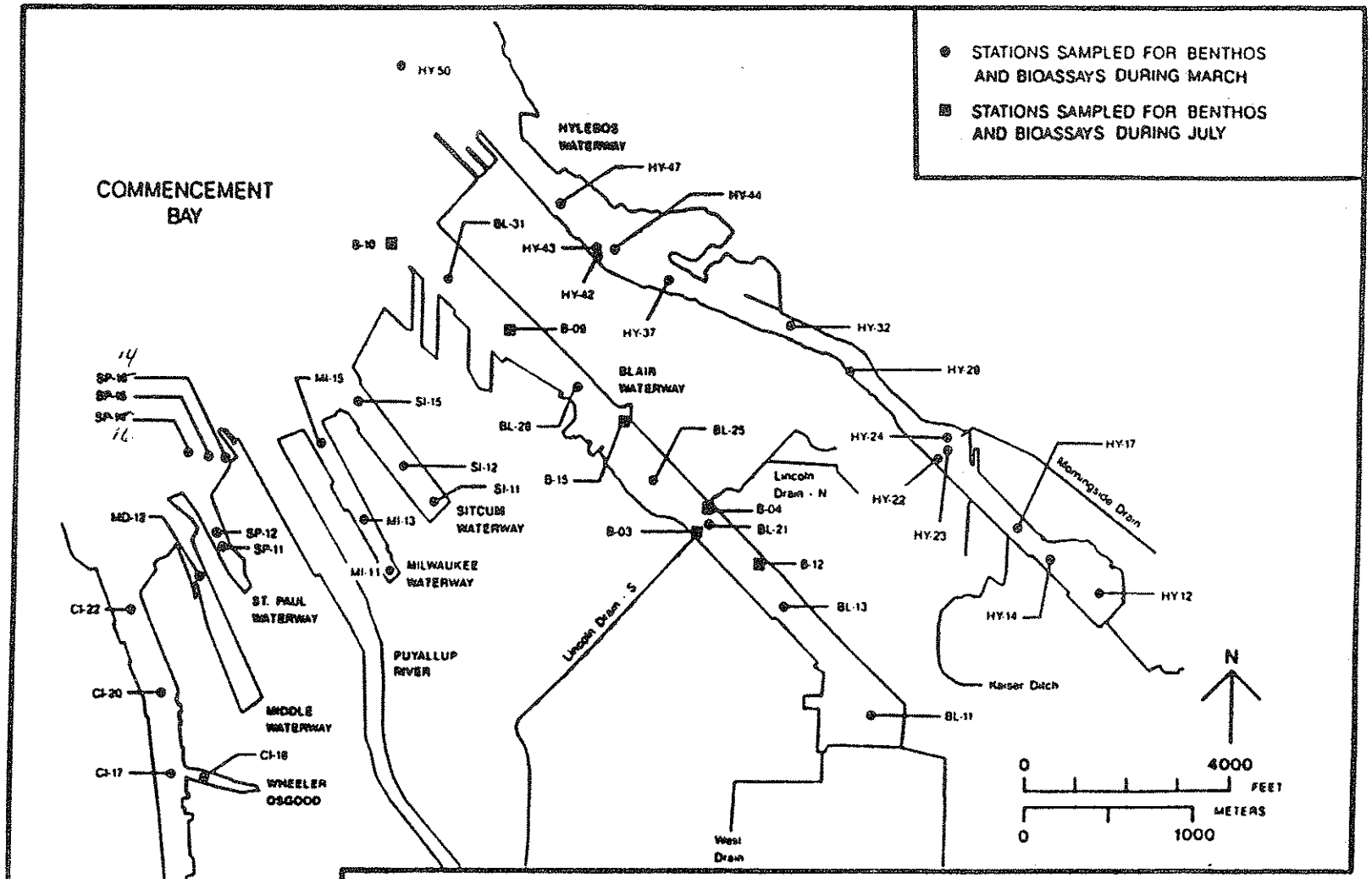

Figure B-1. Location of Commencement Bay stations sampled for benthic macroinvertebrates and sediment bioassays during March and July. (Reference: Barrick et al. 1985).

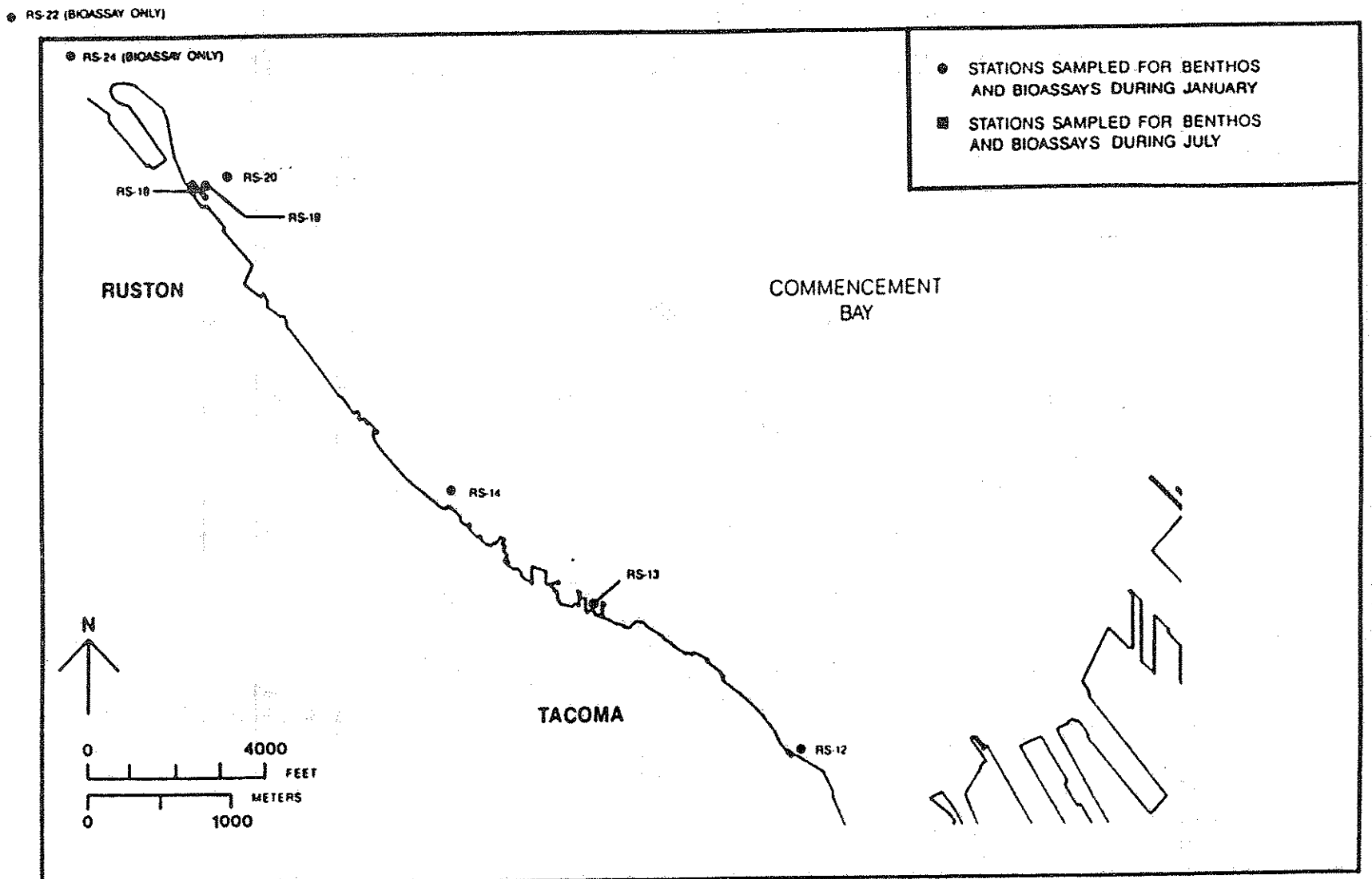

Figure B-1. (Continued).

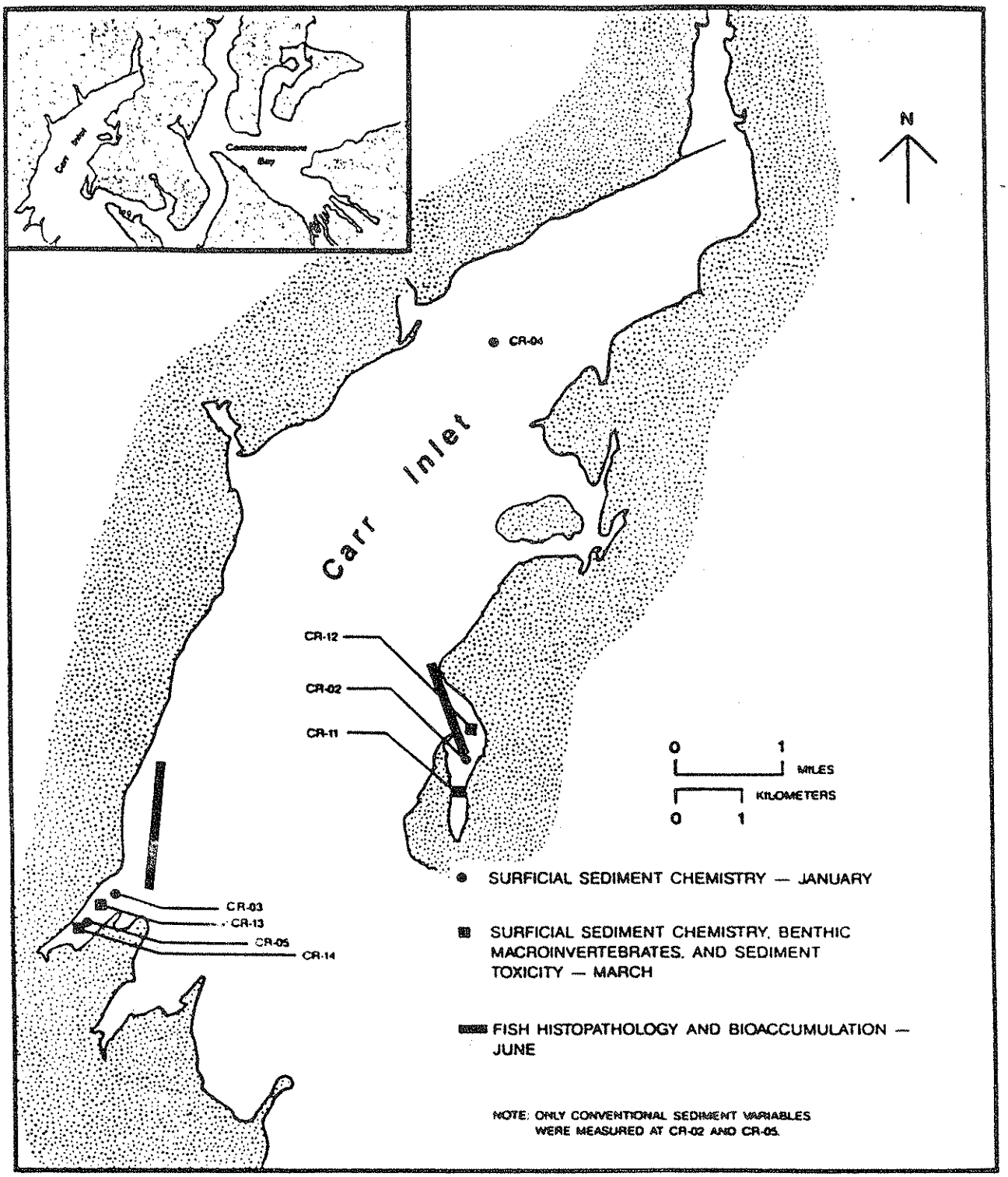

Figure B-2. Locations of reference stations sampled in Carr Inlet. (Reference: Barrick et al. 1985).

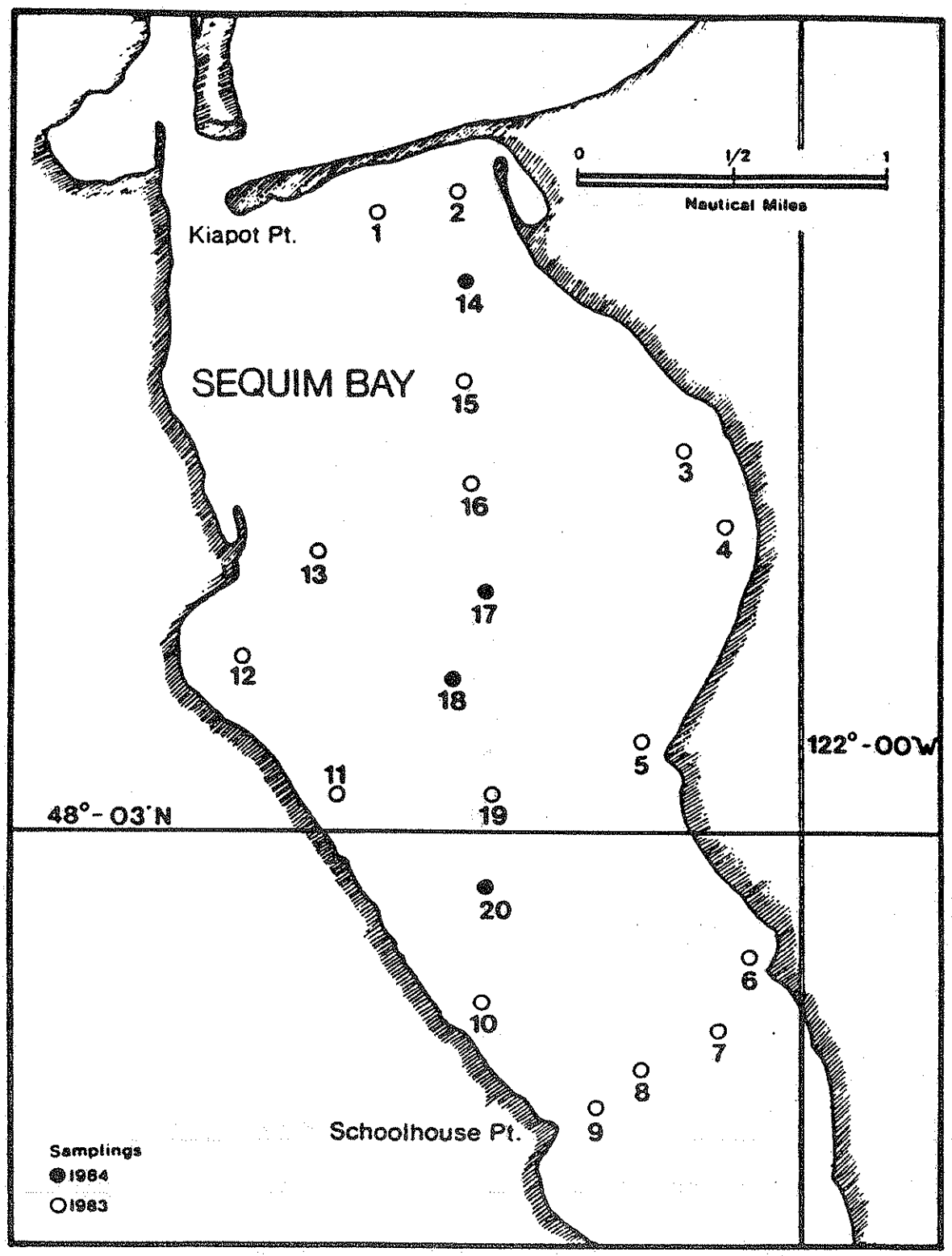

Figure B-3. Sequim Bay sampling stations. (Reference: Battelle 1985).

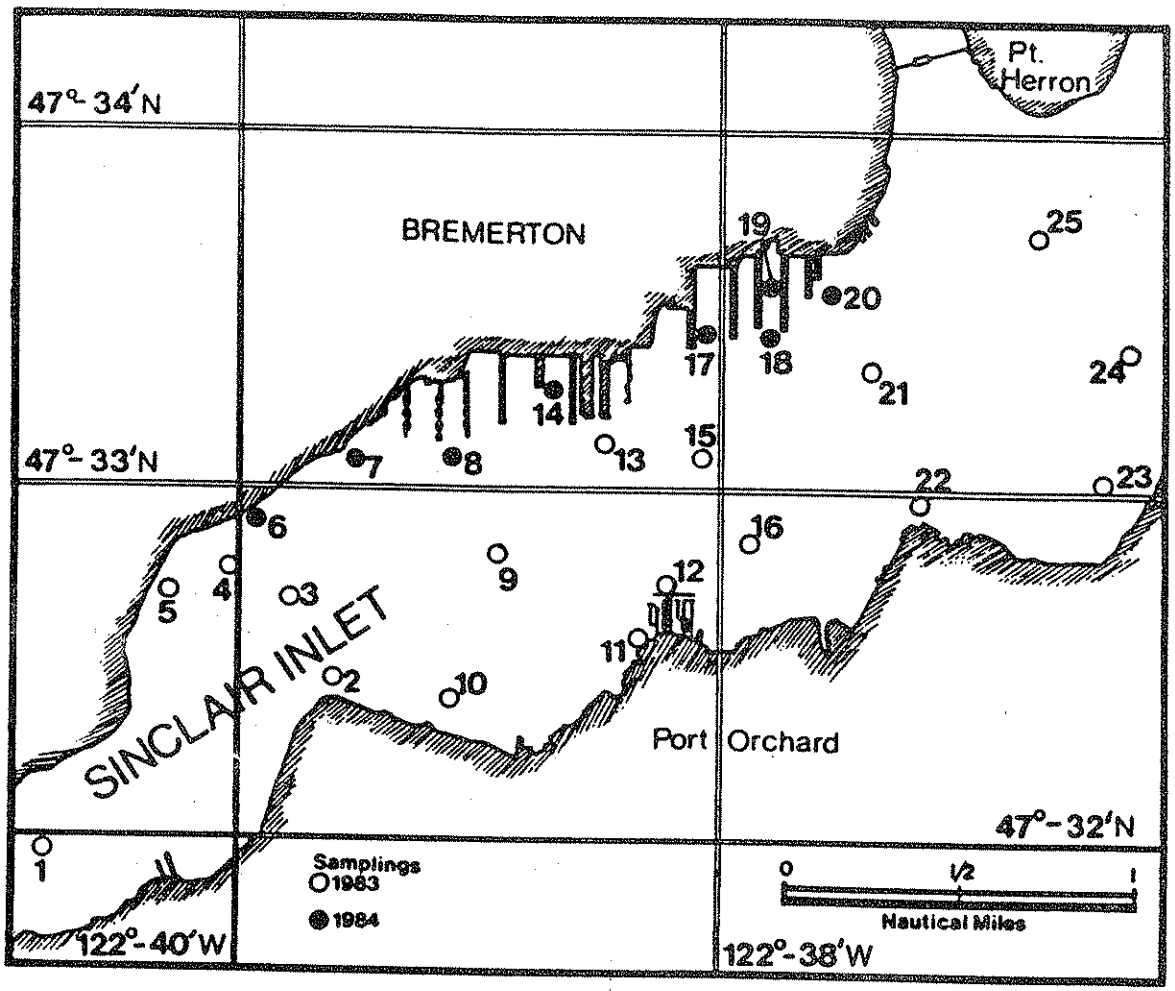

Figure B-4. Sinclair Inlet sampling stations. (Reference: Battelle 1985).

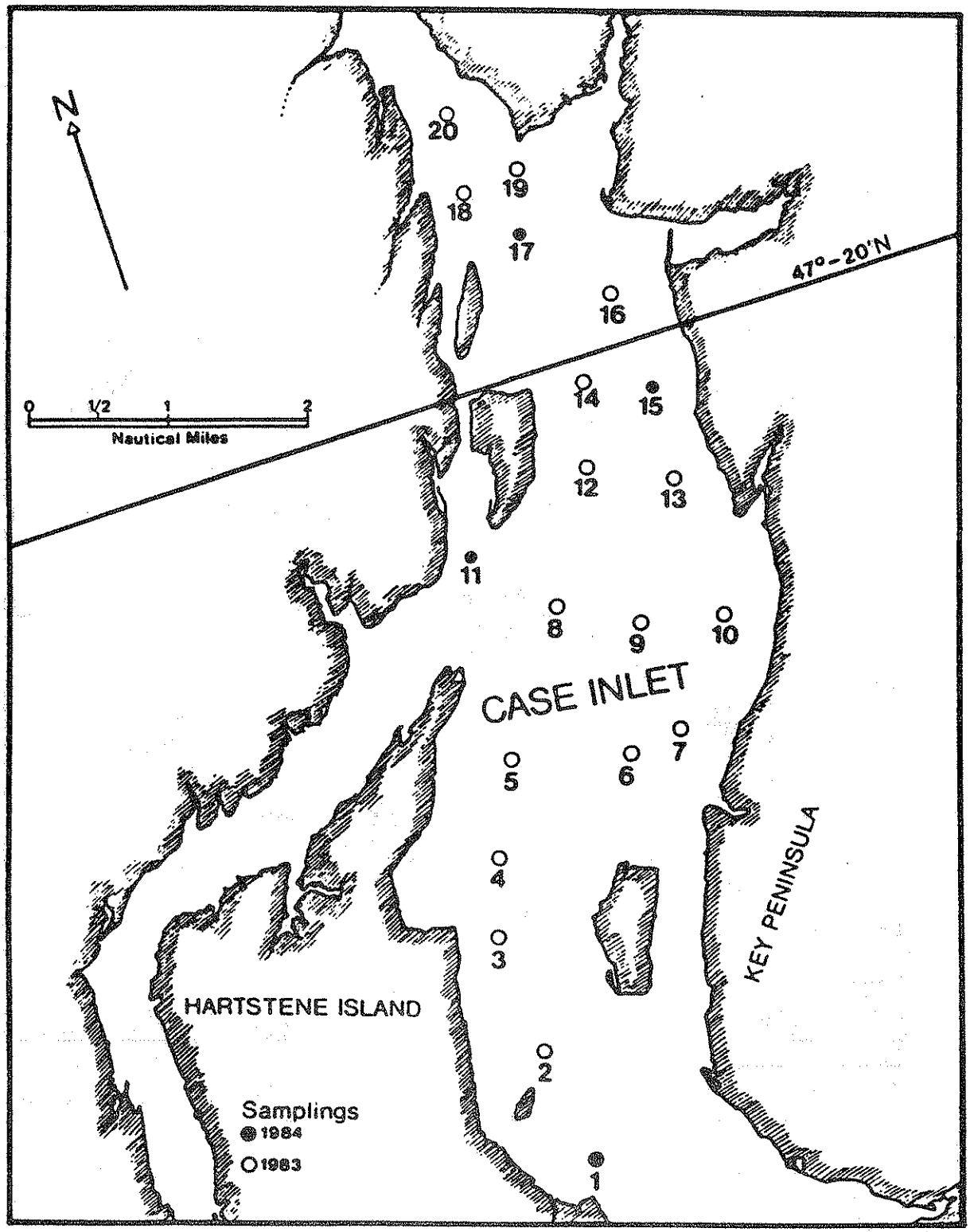

Figure B-5. Case Inlet sampling stations. (Reference: Battelle 1985).

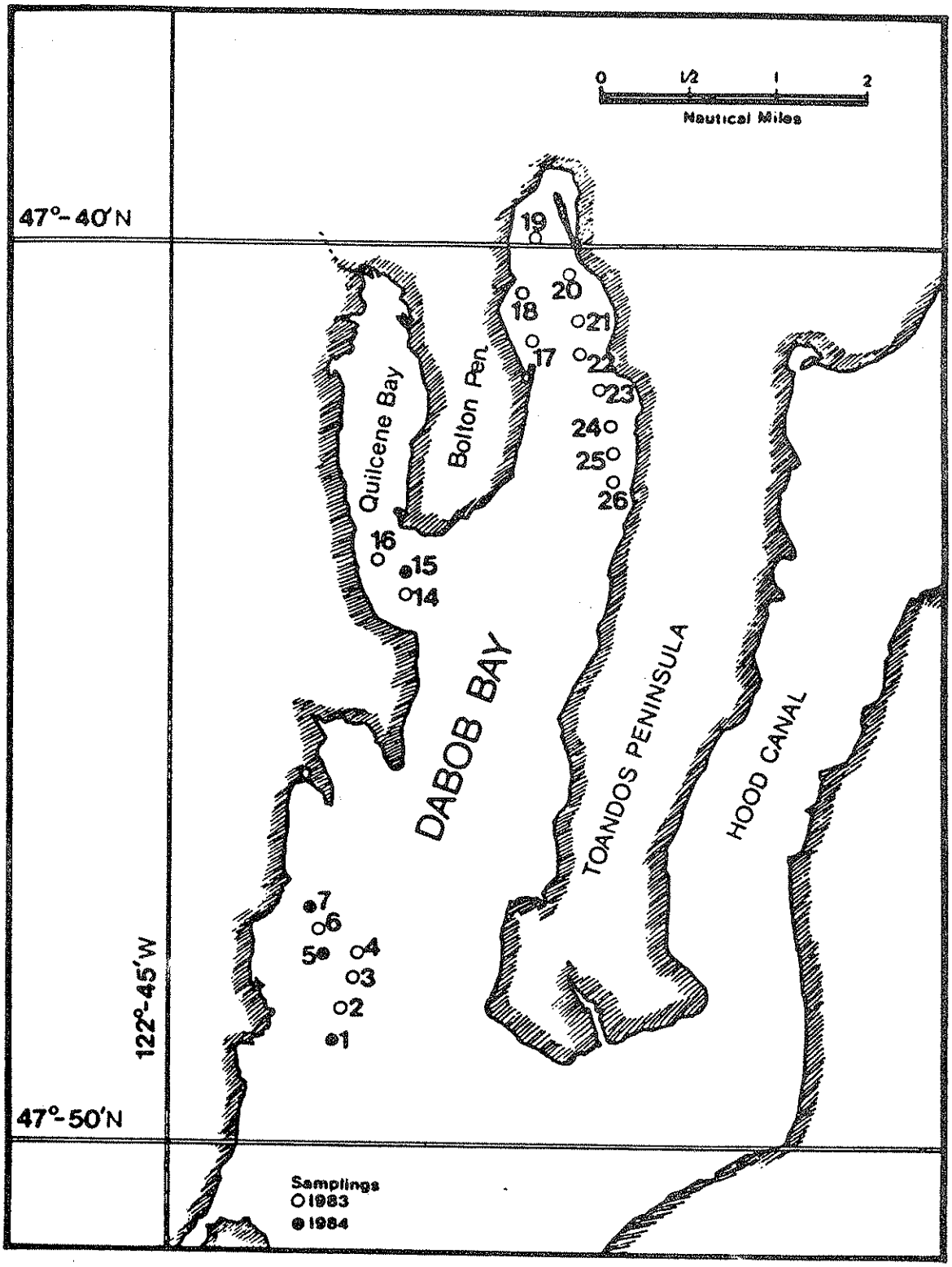

Figure B-6. Dabob Bay sampling stations. (Reference: Battelle 1985).

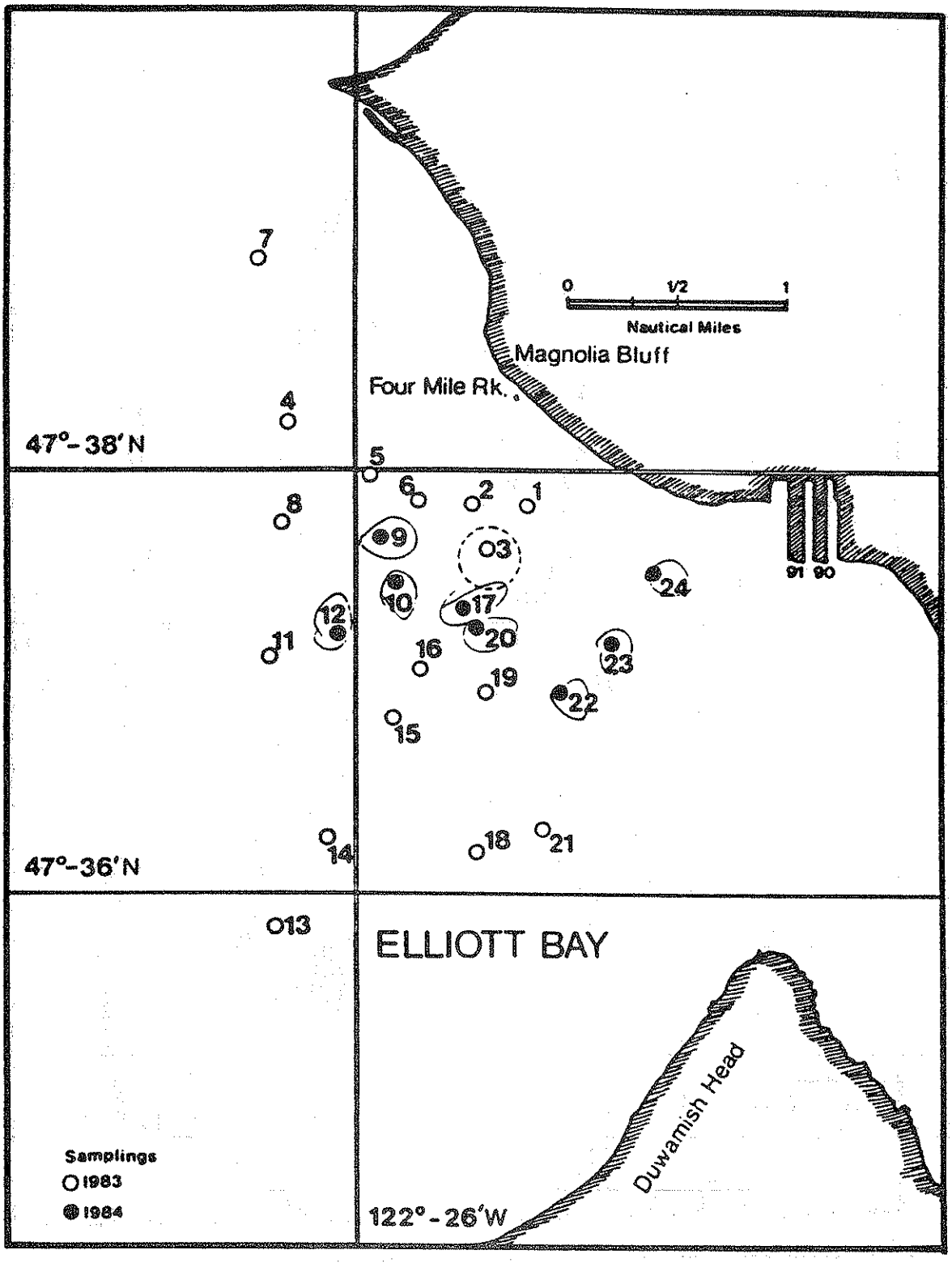

Figure B-7. Elliott Bay - Fourmile Rock sampling stations. (Reference: Battelle 1985).

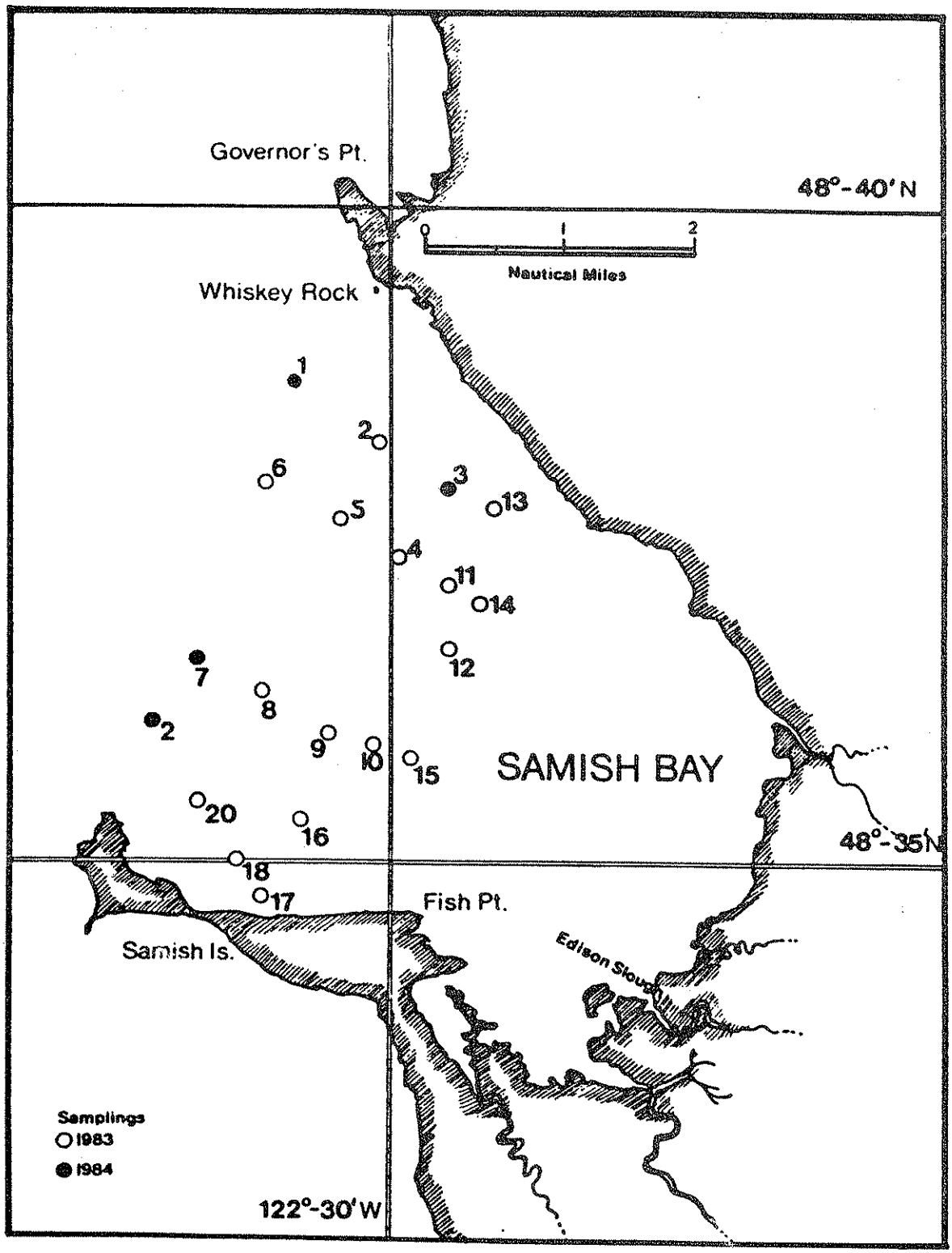

Figure B-8. Samish Bay sampling stations. (Reference: Battelle 1985).

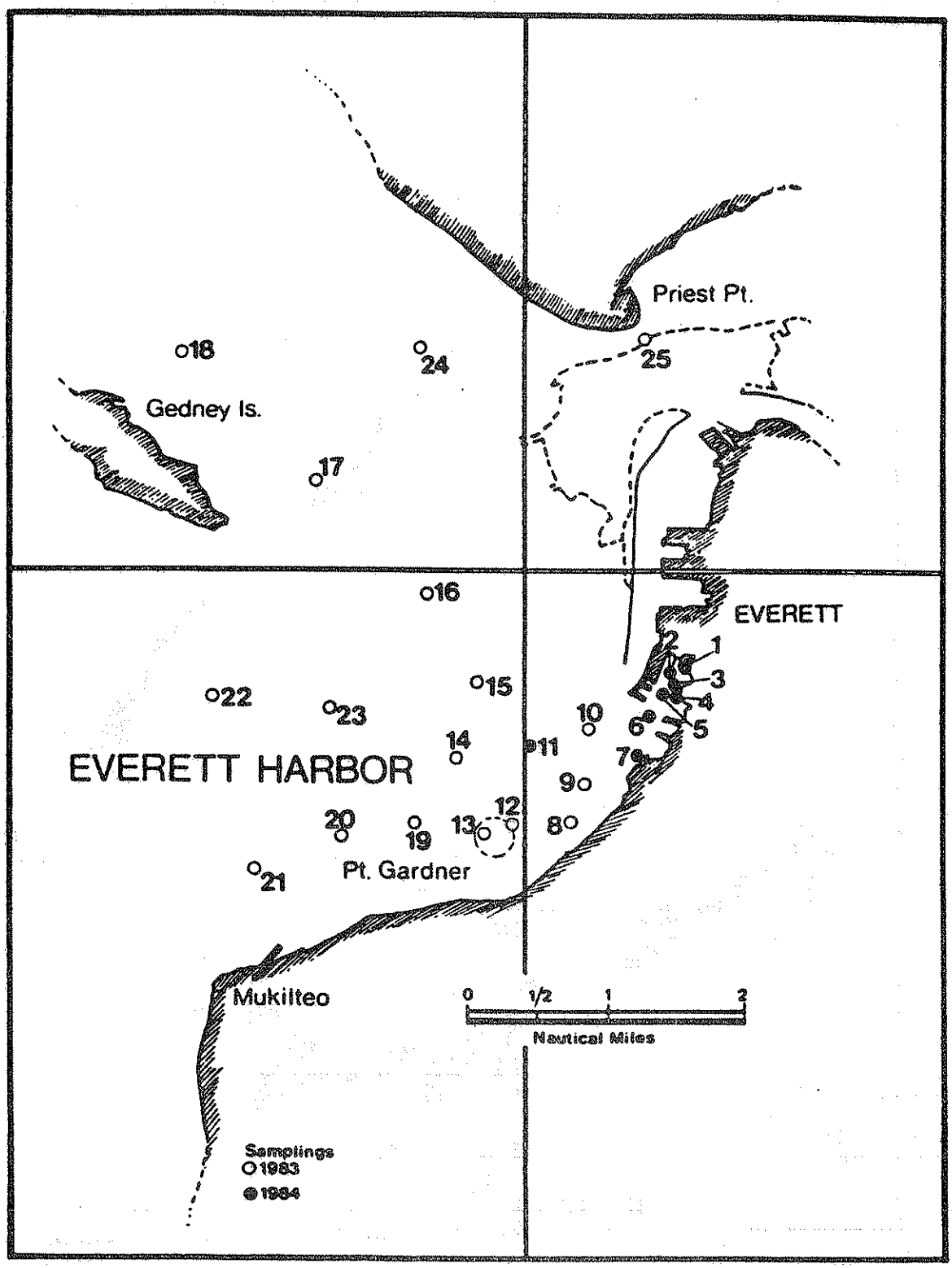

Figure B-9. Everett Harbor - Port Gardner sampling stations. (Reference: Battelle 1985).

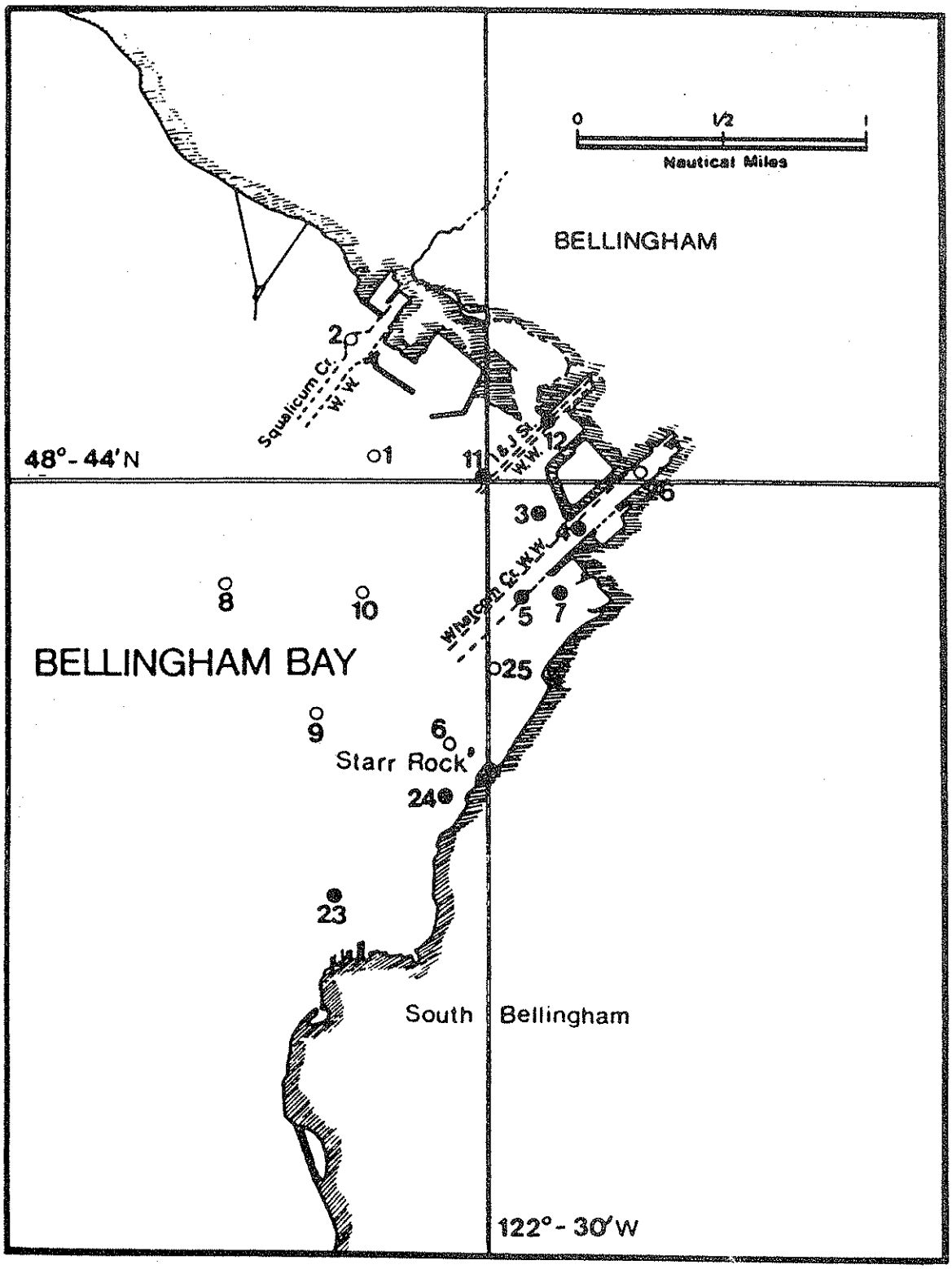

Figure B-10. Bellingham Bay sampling stations (Inner Harbor). (Reference: Battelle 1985).

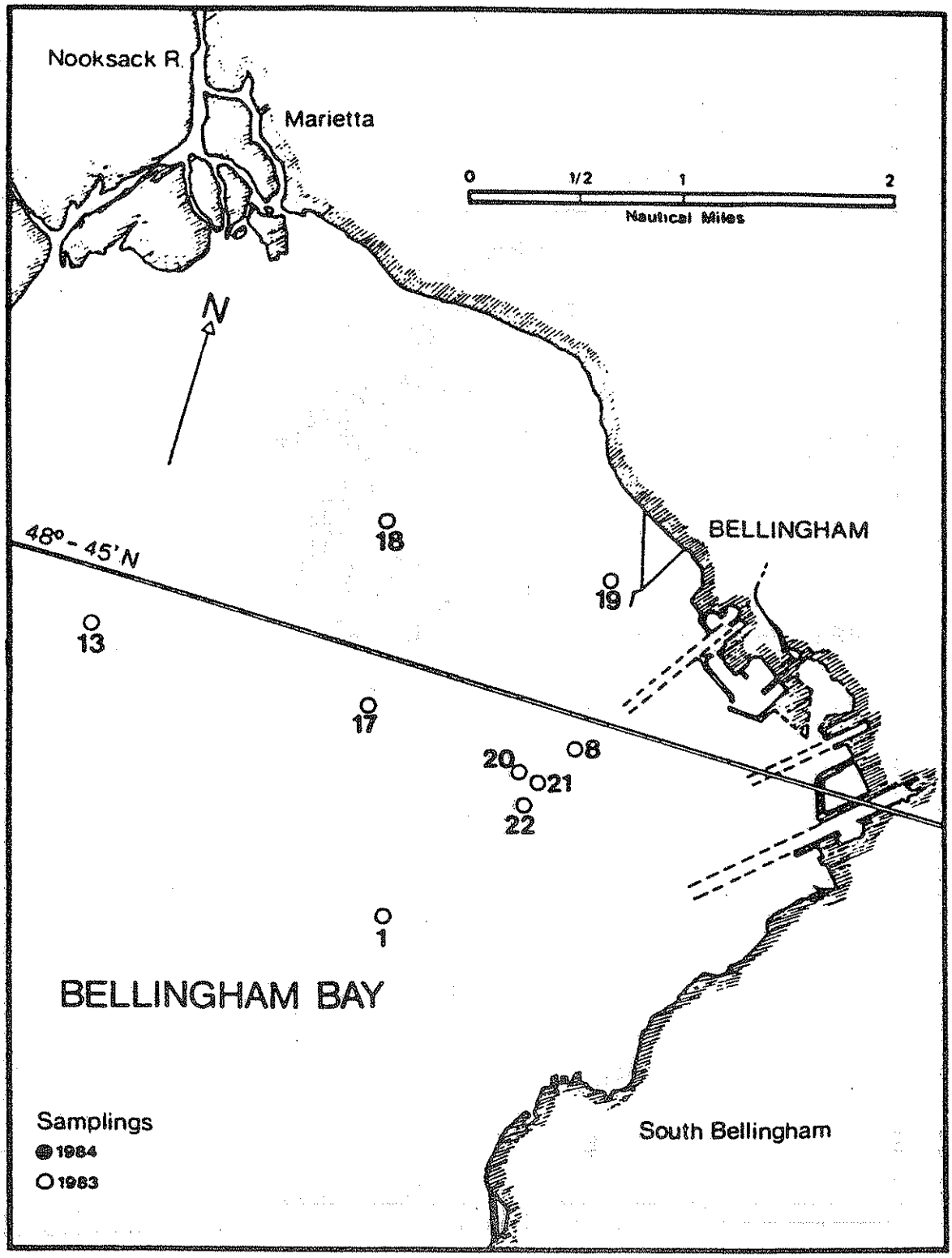

Figure B-11. Bellingham Bay sampling stations (Outer Harbor). (Reference: Battelle 1985).

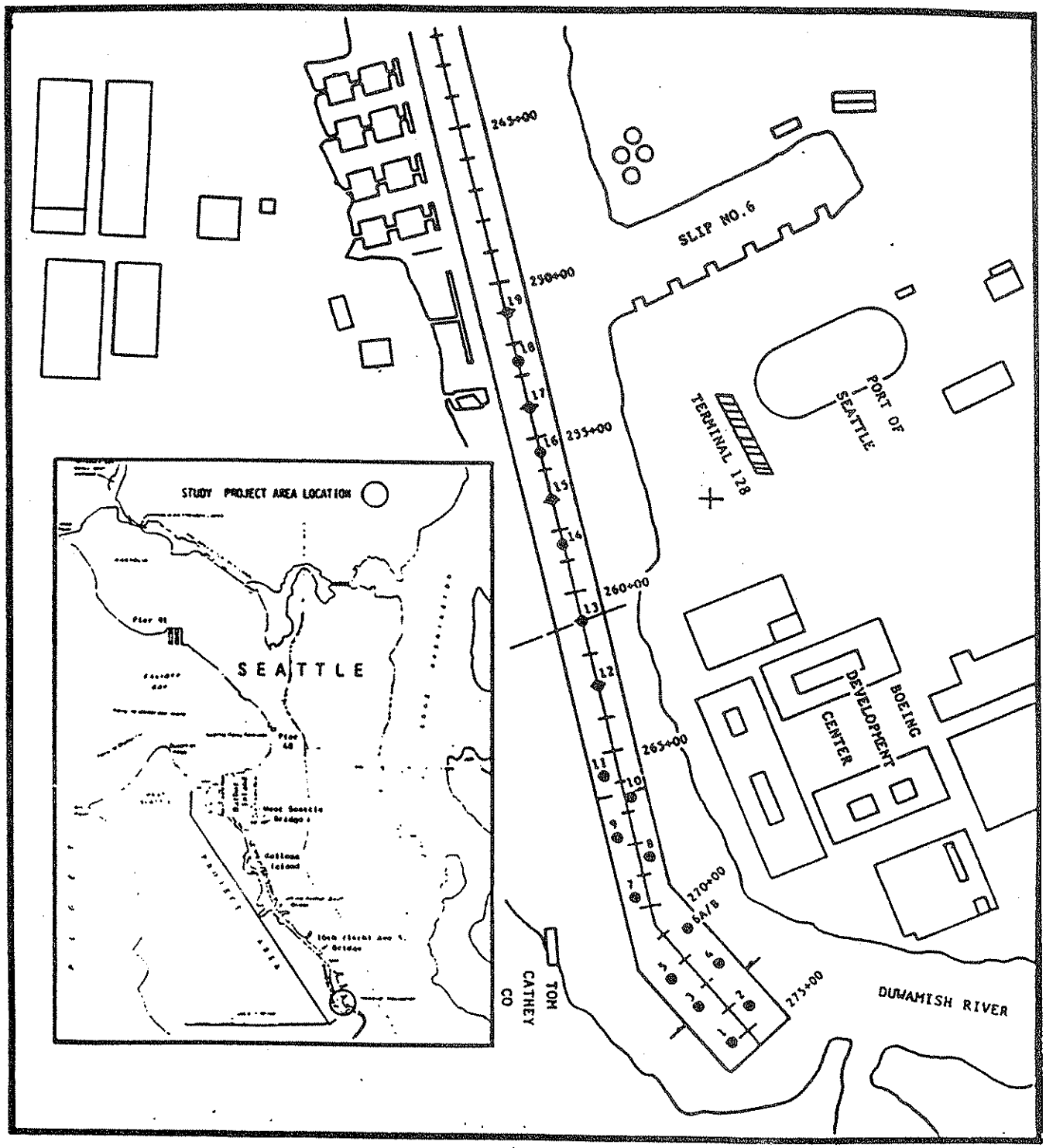

Figure B-12. Sediment sampling station locations for dredged material characterization in Duwamish River. (Reference: Chan et al. 1985).

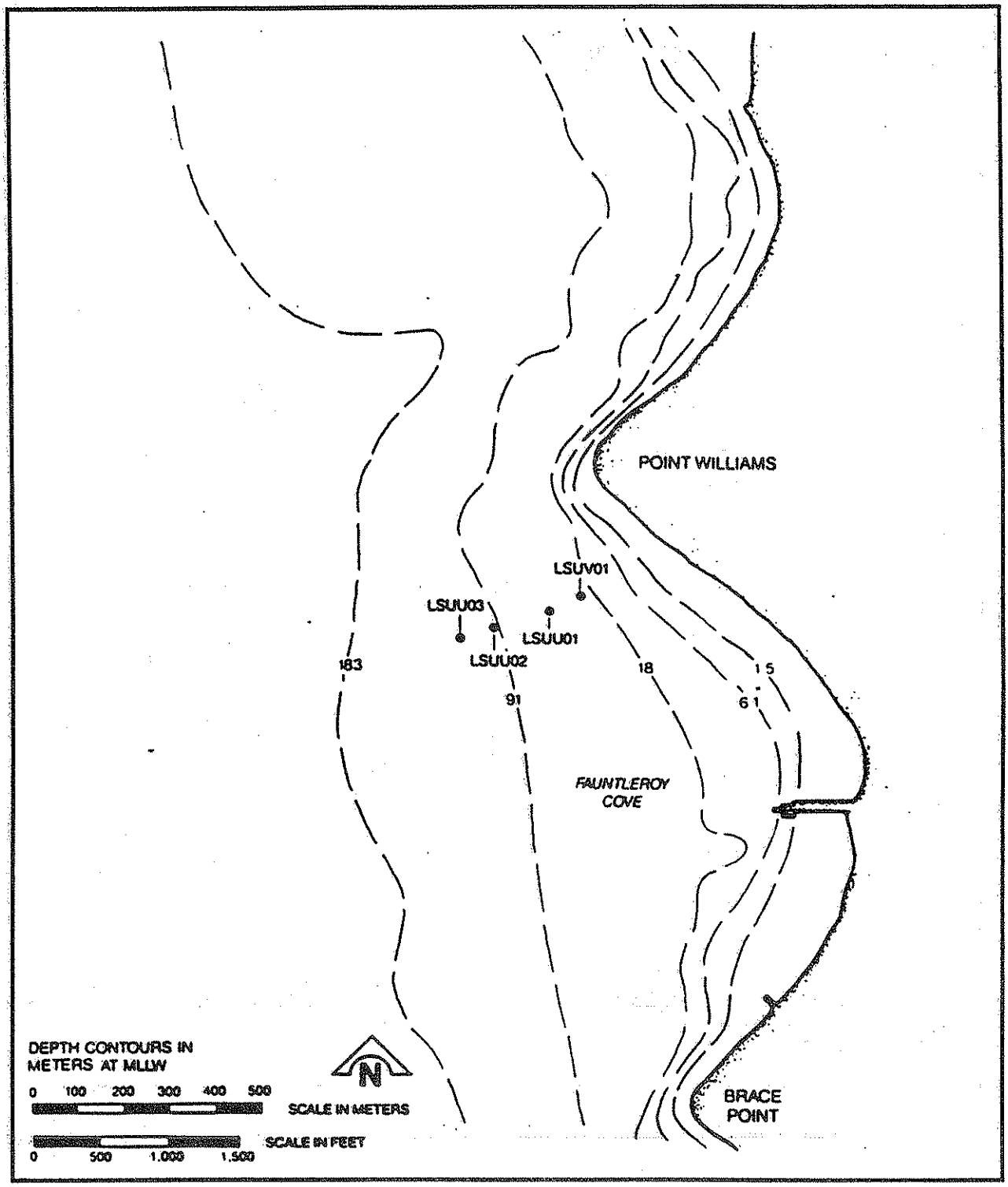

Figure B-13. Point Williams benthos reference sampling station locations. (Reference: Osborn et al. 1985).

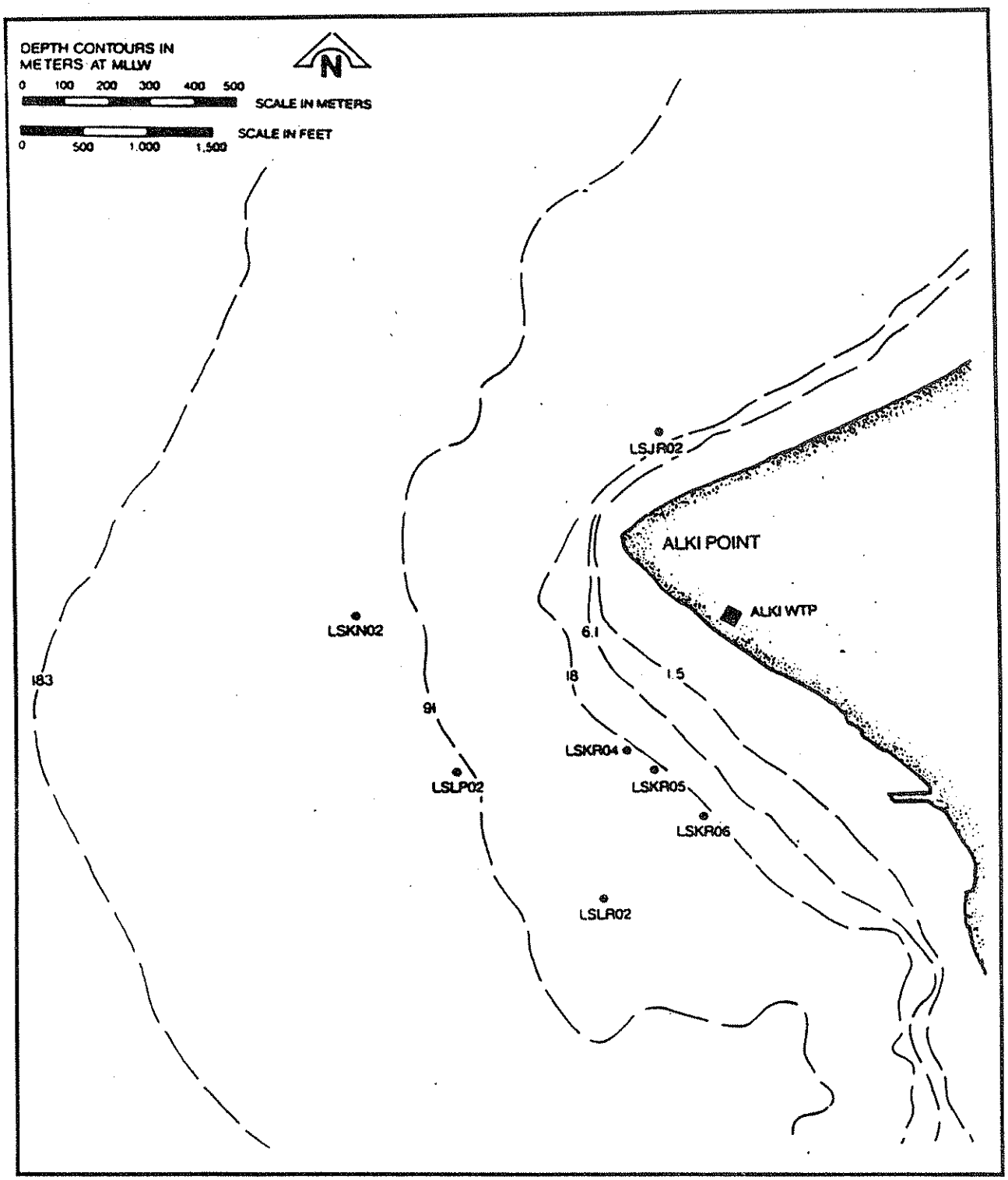

Figure B-14. Alki Point benthos sampling station locations. (Reference: Osborn et al. 1985).

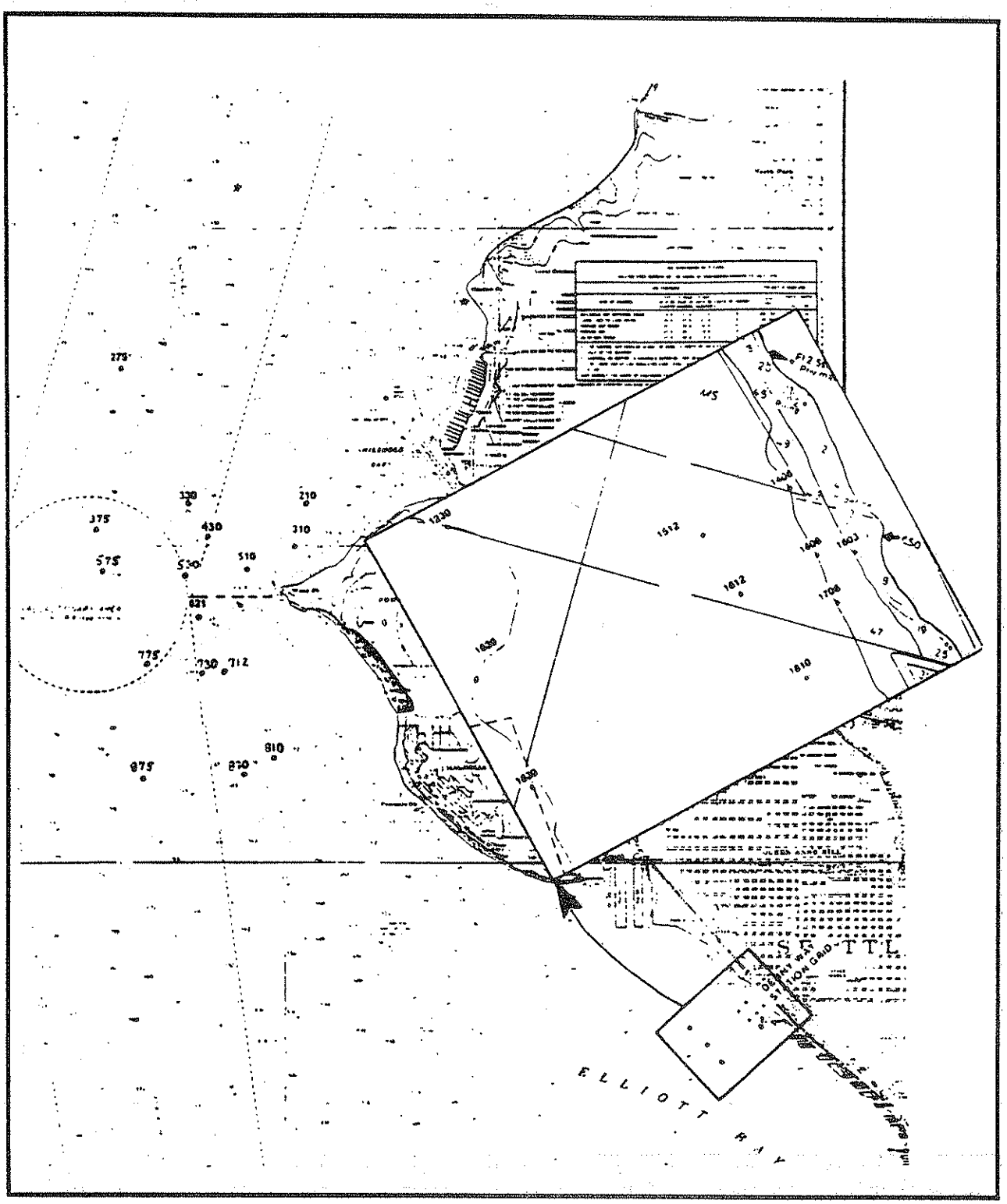

Figure B-15. Map showing the 26 stations in the central basin of Puget Sound and Elliott Bay sampled during Phase III of the TPPS program. (Reference: Romberg et al. 1984).

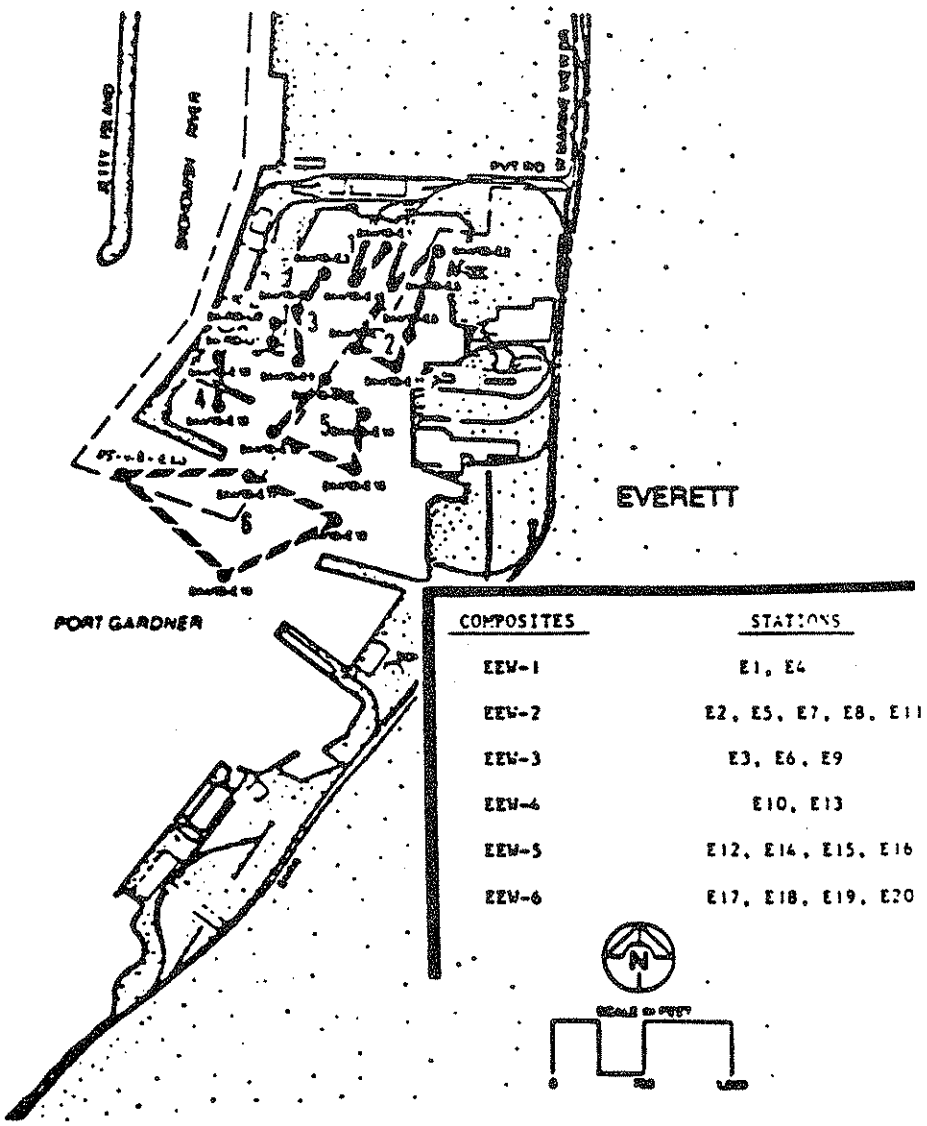

Figure B-16. Navy sediment sampling locations in the East Waterway of Everett. (Reference: U.S. Navy 1985).

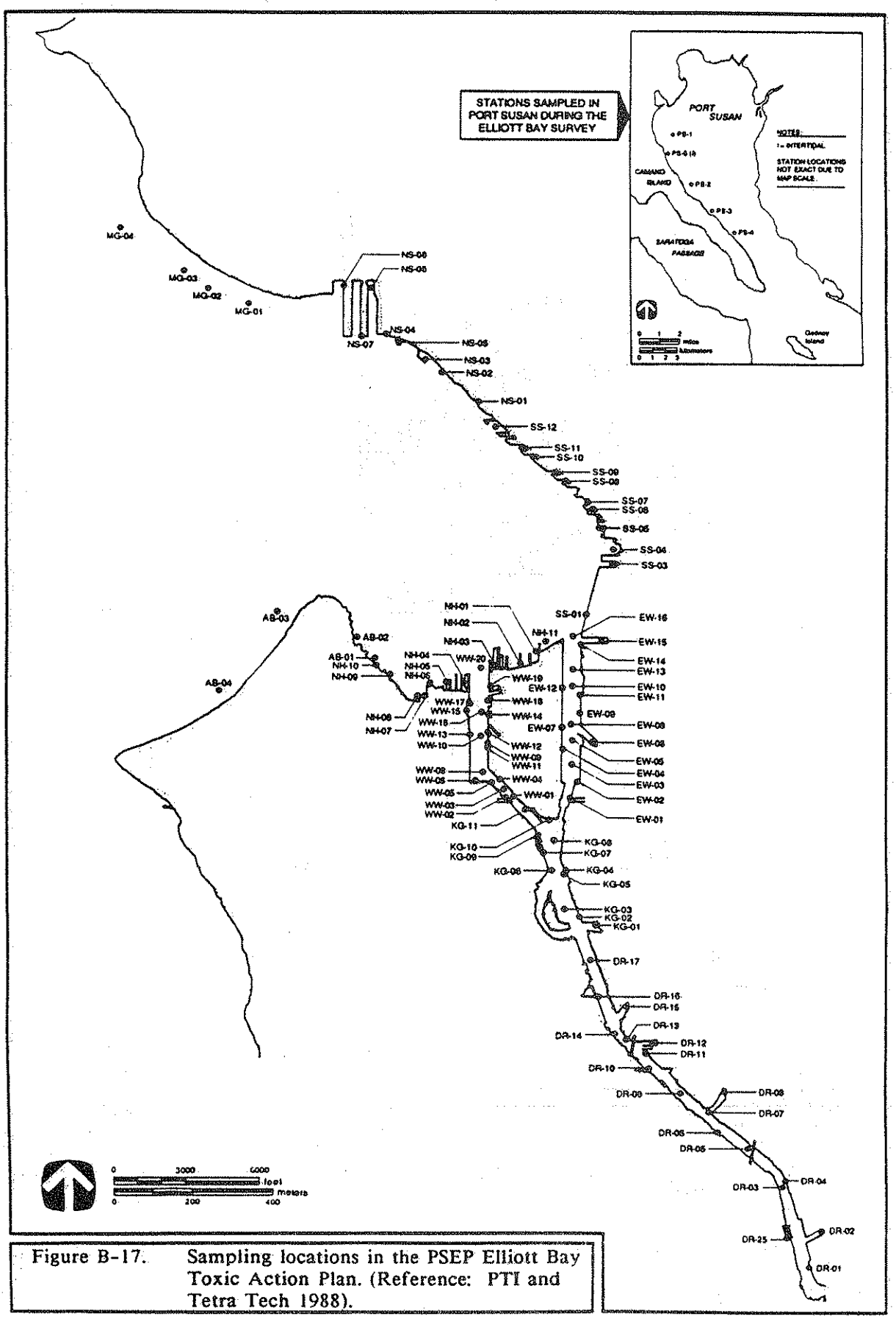

Figure B-17. Sampling locations in the PSEP Elliott Bay Toxic Action Plan. (Reference: PTI and Tetra Tech 1988).

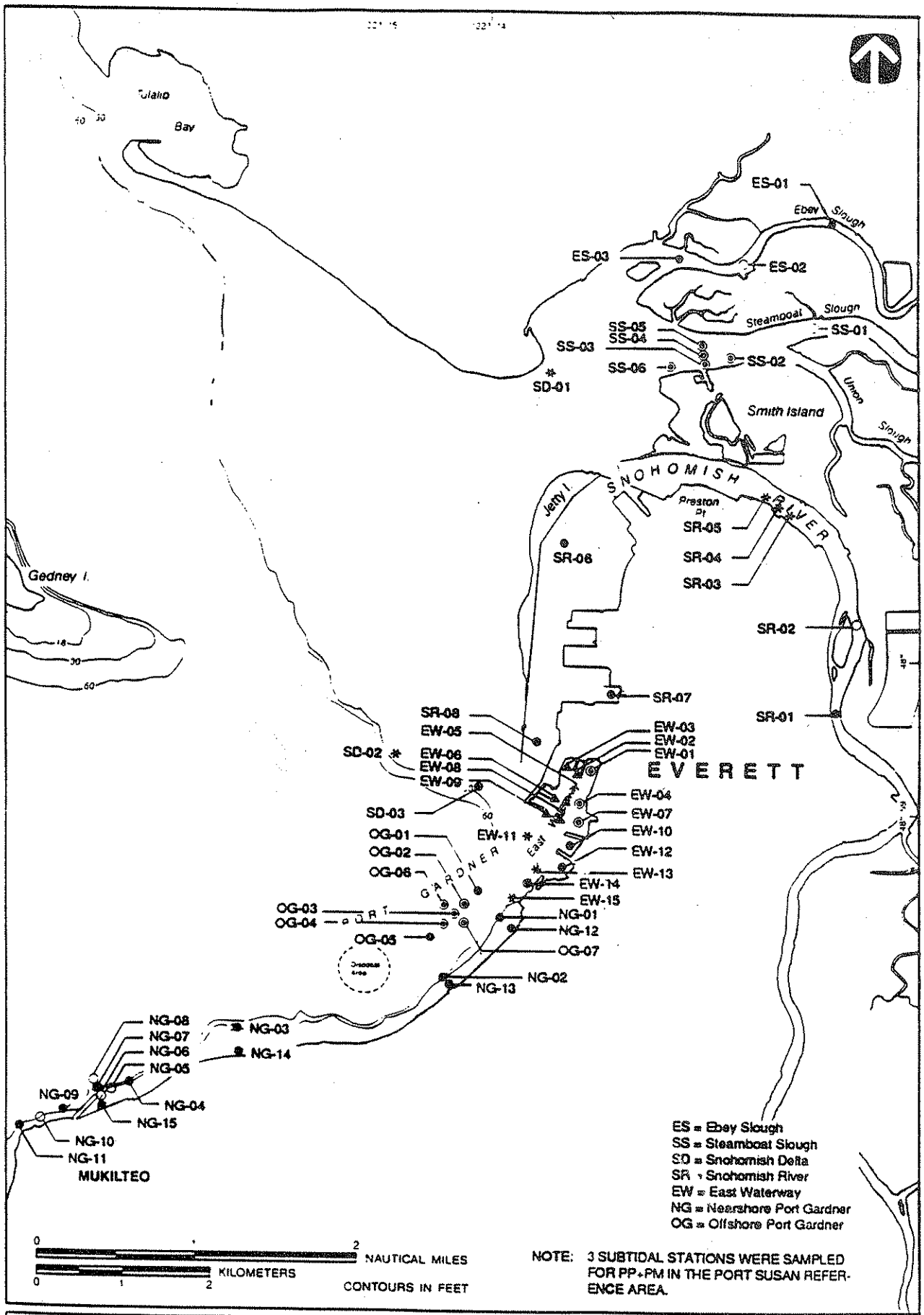

Figure B-18. Approximate sampling locations for PSEP chemical studies in Everett Harbor. (Reference: PTI and Tetra Tech 1988).

PP (INCLUDES VOLATILES)
PP - V
PP + PM
PP - V + PM
V + PM

PP = Priority Pollutants
V = Volatile Compounds
PM = Pulpmill Compounds

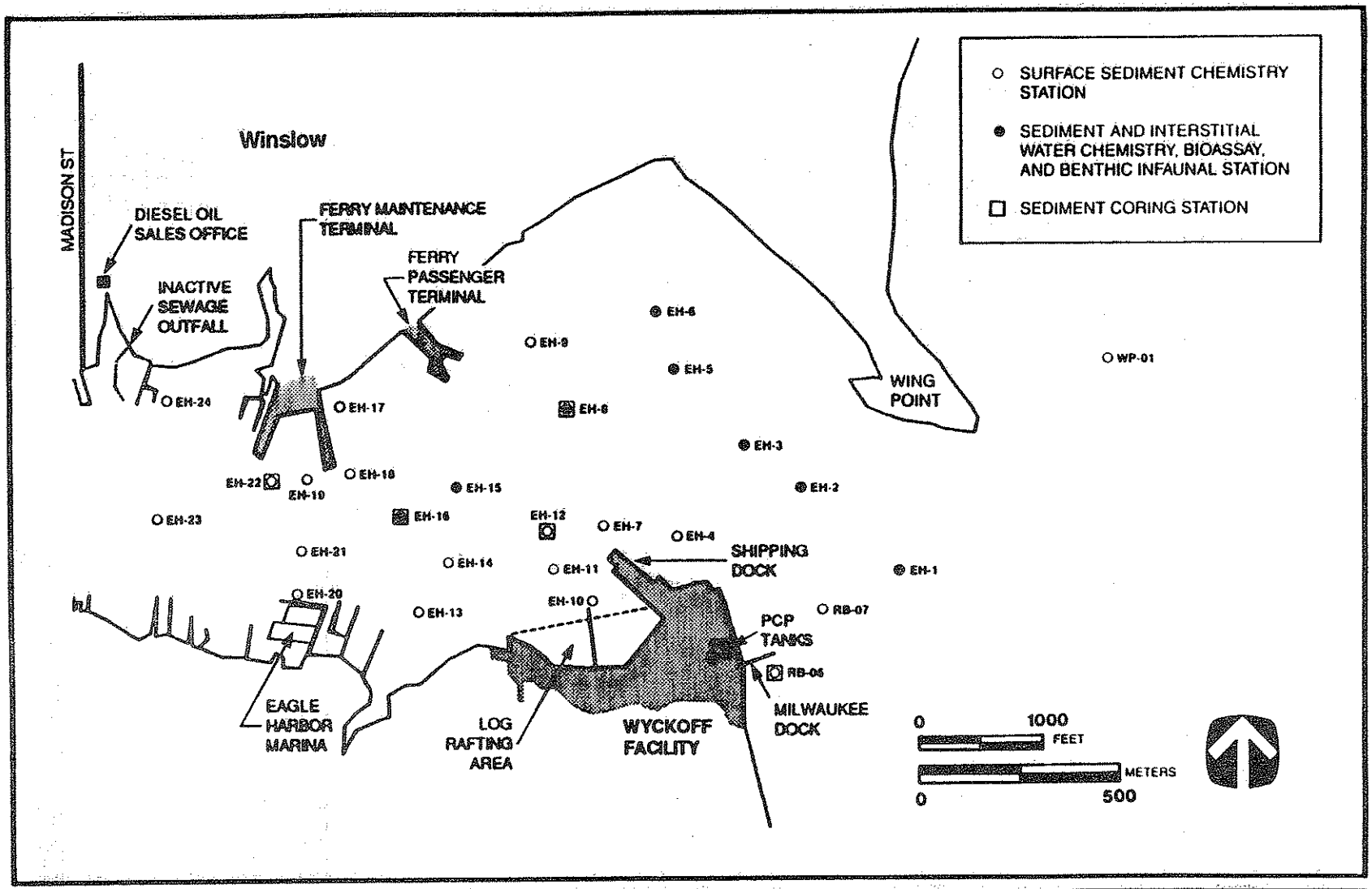

Figure B-19. Sample station locations in Eagle Harbor Preliminary Investigation. (Reference: Barrick et al. 1986).

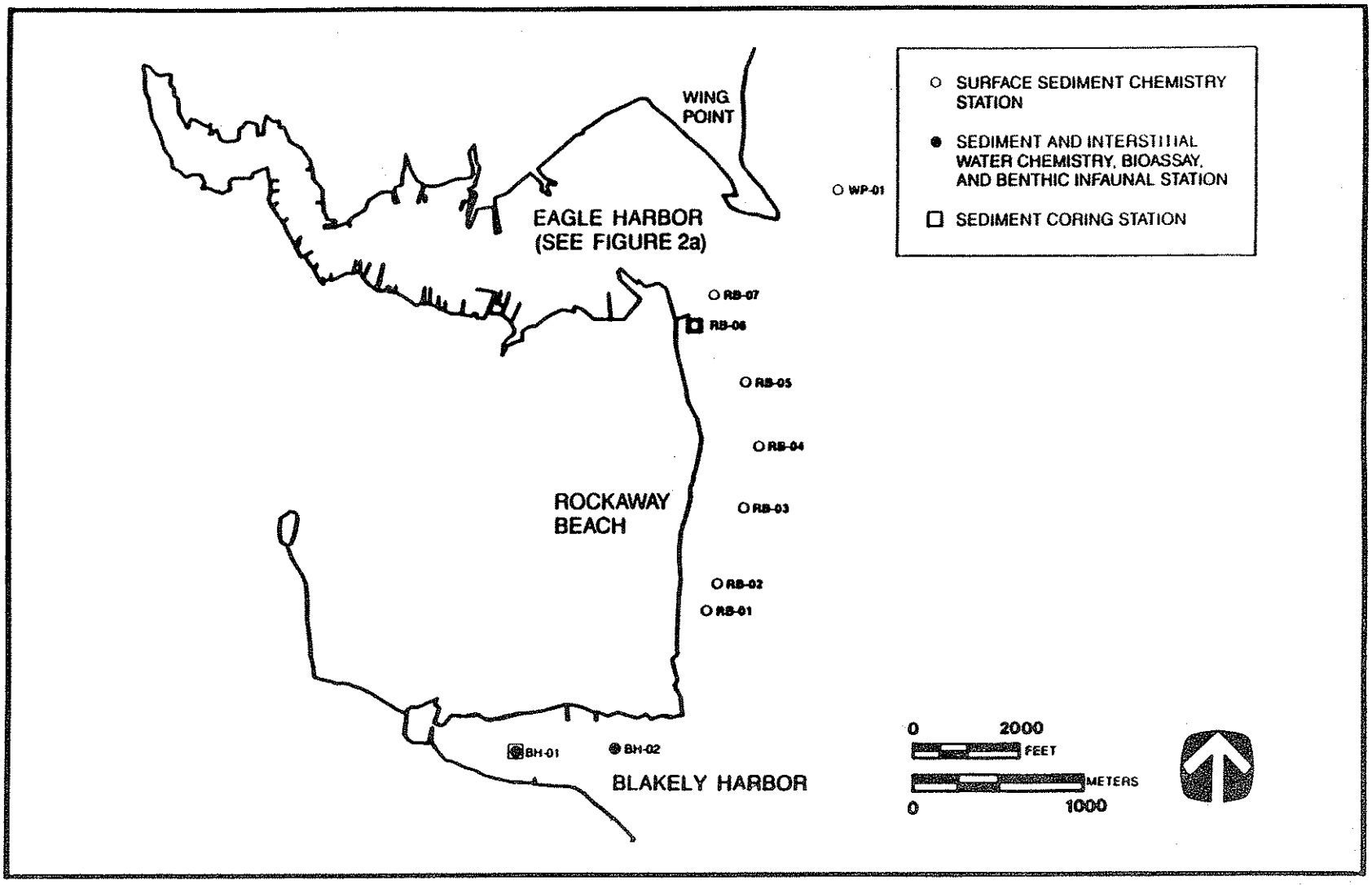

Figure B-20. Sample station locations in Blakely Harbor reference area. (Reference: Barrick et al. 1986).

TABLE B-7. RECOMMENDED PROCEDURES FOR TREATMENT OF DATA USED TO DEVELOP PUGET SOUND AET

Issue	Recommendation	Status
Biological Data Evaluation		
Reference	Accept reference data that meets guidelines for mean values and standard deviation (S.D.) (e.g., mean amphipod mortality <25 percent and standard deviation (S.D.) <20).	Implemented for amphipod bioassay.
	Use one of reference data options described in text when full study-specific data set does not meet guidelines above.	Used partial reference data set for amphipod bioassay.
Statistical comparisons	Test between-site differences using ANOVA with pairwise alpha of 0.05. Classify significant (P<0.05) differences from reference as "impacts", when the effects guidelines are exceeded (e.g., 25 percent mortality for amphipod bioassay, 50 percent depression for infauna).	Implemented for amphipod bioassay and infauna.
	If apparent effect does not exceed the guideline, then the site is classified as "nonimpacted."	Implemented for amphipod bioassay and infauna.
Power analysis	Classify stations that are not significantly different from reference (P>0.05) with S.D. less than guideline (e.g., S.D. <15 for amphipod bioassay) as "nonimpacted."	Implemented for amphipod bioassay.
	For nonsignificant stations exceeding S.D. guideline:	Implemented for amphipod bioassay.
	a) Classify as "nonimpacted" if variance does not exceed guideline corresponding to power = 0.6 (i.e., power is predicted as ≥0.6).	
	b) Classify as "inconclusive" if variance exceeds guideline corresponding to power = 0.6 (i.e., power is predicted as <0.6).	

TABLE B-7. **Continued**

Chemical Data Evaluation		
Chemical qualifiers	In calculating sediment quality values, exclude undetected values and data that are recovery-corrected by a factor >10.	Implemented, except for recovery-corrected data.
AET representativeness	Accept nonimpacted stations for which there is confirming evidence within a concentration factor of 3 by at least one additional nonimpacted station.	Implemented.
	Also require on a case-by-case basis, confirmation by at least one nonimpacted station at a different geographic location.	Not implemented.

TABLE B-8. STATIONS EXCLUDED FROM AET CALCULATIONS AND RELIABILITY TEST

Amphipod Stations Rejected Because of Inadequate Statistical Power (P>0.05)[a]	
EBCHEM	DR-05
EBCHEM	DR-08
EBCHEM	EW-04
EBCHEM	NS-04
EVCHEM	EW-14
EVCHEM	SD-02
EVCHEM	SR-07

Amphipod Stations Rejected Because of Chemical Anomaly Rule[a]	
CBMSQS	CI-16
CMBSQS	HY-12
EBCHEM	DR-10
EBCHEM	DR-12
EBCHEM	NS-06
EBCHEM	SS-04
EBCHEM	SS-10
EBCHEM	WW-14

Benthic Stations Rejected Because of Chemical Anomaly Rule[a]	
EBCHEM	AB-01
EBCHEM	SS-10
TPPS3AB	WP-11
TPPS3AB	WP-15

[a] See Table B-6 for summary of anomaly rules.

TABLE B-9. STATIONS ANOMALOUS WITH RESPECT TO SEDIMENT CHEMISTRY[a]

Survey	Station	Chemical	Ratio of Anomalous Station to Next Highest Station
		Chemical Anomalies for the Amphipod Mortality Bioassay	
CBMSQS	CI-16	N-nitrosodiphenylamine	4.5
	HY-12	Di-n-butyl phthalate	3.2
EBCHEM	DR-10	DDE	4.2
	DR-12	Arsenic	4.8
	NS-06	Di-n-octyl phthalate	14.3
	SS-04	Dibenzoanthracene	3.6
	SS-10	Chromium	4.0
	WW-14	Antimony	7.1
		Lead	14.3
		Chemical Anomalies for Benthic Infaunal Abundance	
EBCHEM	AB-01	Mercury	14.3
	SS-10	Chromium	4.2
TPPS3AB	WP-11 (3/15/82)	Benzo(a)pyrene	4.8
		Benzo(ghi)perylene	3.4
		HPAH	3.0
		Phenanthrene	3.4
		Total benzofluoranthenes	7.1
	WP-14 (7/15/82)	Di-n-octyl phthalate	20.0

[a] Nonimpacted stations that potentially set AET but exceed the next highest nonimpacted station for one or more chemicals by greater than a factor of 3 (e.g., benthic effects at EBCHEM Station SS-10 were statistically significant at $P \leq 0.05$ but the observed depressions were <50 percent, so the station was classified as nonimpacted).