Exponent®

NASSCO and Southwest Marine Detailed Sediment Investigation

Volume I

Prepared for

NASSCO and Southwest Marine
San Diego, California

Exponent®

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Volume I

Prepared for

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Acronyms and Abbreviations

95%LPL	95 percent lower prediction limits
95%UPL	95 percent upper prediction limits
AET	apparent effects threshold
AIT	Advance Installation Team
ANOVA	analysis of variance
ARAR	applicable or relevant and appropriate requirement
AVS	acid-volatile sulfide
BMP	best management practices
BPTCP	Bay Protection and Toxic Cleanup Program
BRI	benthic response index
CAD	confined aquatic disposal
CDF	nearshore confined disposal facility
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act of 1980
Corps	U.S. Army Corps of Engineers
CSF	carcinogenic slope factor
CTR	California Toxics Rule
DRO	diesel-range organics
EPA	U.S. Environmental Protection Agency
ER-L	effect range-low
ER-M	effect range-median
FSP	field sampling plan
FWS	U.S. Fish and Wildlife Service
GPS	global positioning system
GRO	gasoline-range organics
HPAH	high-molecular-weight polycyclic aromatic hydrocarbon
IRIS	Integrated Risk Information System
LAET	lowest apparent effects threshold
LOAEL	lowest-observed-adverse-effect level
LPAH	low-molecular-weight polycyclic aromatic hydrocarbon
MDS	multidimensional scaling
MLLW	mean lower low water
NASSCO	National Steel and Shipbuilding Company
Navy	U.S. Navy
NIW	National Iron Works
NOAEL	no-observed-adverse-effect level
NPDES	National Pollutant Discharge Elimination System
OEHHA	Office of Environmental Health Hazard Assessment
OSHA	Occupational Safety and Health Administration
PAH	polycyclic aromatic hydrocarbon
PCA	principal components analysis
PCB	polychlorinated biphenyl
PCT	polychlorinated terphenyl

RfD	reference dose
RPD	redox potential discontinuity
RRO	residual-range organics
RWQCB	California Regional Water Quality Control Board, San Diego Region
SDI	Swartz' dominance index
SEM	simultaneously extracted metals
SPI	sediment profile imaging
SWRCB	State Water Resources Control Board
TBT	tributyltin
TOC	total organic carbon
TRG	tissue residue guideline
TRV	toxicity reference value
UCL	upper confidence limit

Executive Summary

National Steel and Shipbuilding Company (NASSCO) and Southwest Marine Inc. have conducted a detailed sediment investigation to determine the existence and extent of potential beneficial use impairments in San Diego Bay attributable to chemicals associated with historical operations at the shipyards. This investigation was conducted in response to Resolutions No. 2001-02 and 2001-03 of the California Regional Water Quality Control Board, San Diego Region and subsequent Water Code Section 13267 letters issued to the shipyards (Robertus 2001a, pers. comm.).

Beneficial uses addressed by this investigation are aquatic life, aquatic-dependent wildlife, and human health (RWQCB 2001). The objectives of this investigation include determination of potential adverse impacts to beneficial uses, and development and evaluation of remedial alternatives that address adverse impacts attributable to shipyard-associated chemicals. Classes of chemicals that are considered to be potentially associated with shipyard activities include metals, butyltins, polychlorinated biphenyls (PCBs), polychlorinated terphenyls (PCTs), polycyclic aromatic hydrocarbons (PAHs), and petroleum hydrocarbons (RWQCB 2001).

NASSCO and Southwest Marine have conducted shipbuilding and repair activities at their current locations since 1960 and 1979, respectively. Comprehensive best management practices and pollution prevention programs have been instituted at both shipyards over the past 20 years, and neither shipyard currently discharges any materials, either process waste or storm water runoff associated with industrial activity, to San Diego Bay. Currently active potential sources of contaminants to the shipyard leaseholds are two municipal storm drains (one at each shipyard), effluent from Chollas Creek, surface water runoff from Sicard Street, and discharges from vessels. Both shipyard properties have been used for shipbuilding and other industrial operations since the first decades of the 20th century, and these historical activities are also sources of contaminants through past discharges to the bay or into the fill material on which the shipyards are located.

Data Collection

Two phases of fieldwork at the shipyards and reference locations were conducted in 2001 and 2002. The analyses carried out included chemical analyses of sediment, pore water, and tissues of indigenous organisms; mineralogical microprobe analyses; sediment toxicity tests measuring amphipod survival, echinoderm fertilization, and bivalve larvae development; sediment profile images; benthic macroinvertebrate community analyses; chemical bioaccumulation tests; histopathological examinations of fish; and analyses of fish bile for PAH breakdown products. Samples were collected from within and outside the shipyard leaseholds (to the north of Southwest Marine and between the leasehold boundaries and the shipping channel). Both surface and subsurface sediment samples were analyzed. The extensive data from these investigations were used to evaluate the horizontal and vertical distribution of shipyard-associated chemicals, to evaluate adverse impacts on benthic macroinvertebrates and fish, and to conduct risk assessments for aquatic-dependent wildlife and human health.

Reference stations selected by Regional Board staff (RWQCB 2001; Bay 2001a,b) were sampled during the two field phases of the investigation. Following conclusion of the fieldwork, Regional Board staff directed that data interpretation be conducted using a different set of data to represent reference conditions (the “final reference pool”) (Barker 2003). Physical conditions at the stations included in both sets of reference data differ from physical conditions at the shipyard sites in having coarser grain size and lower total organic carbon (TOC). The reference data sets also do not conform to established standards for reference stations (U.S. EPA 1994, 1997a, 2002a; U.S. DOI 1996). Because higher chemical concentrations are ordinarily associated with finer grain sizes and higher TOC content, and because benthic macroinvertebrate community composition also depends on these variables, the physical differences between reference and shipyard stations are expected to result in different chemical and biological conditions regardless of any influences of the shipyards.

Chemical Conditions

Concentrations of shipyard-associated chemicals in the sediment at both shipyards are generally higher than in the final reference pool samples. The highest concentrations of most chemicals are found at the northern boundary of the Southwest Marine site. The highest concentrations of PAH are found off of the municipal storm drain outfall in the Southwest Marine leasehold. Elevated concentrations of metals are also found near the same storm drain outfall and in the center of the NASSCO leasehold. Elevated concentrations of PCBs are found near the northern boundary of Southwest Marine, at the storm drain outfall on Southwest Marine's leasehold, and at the foot of Sicard Street on the boundary of the two shipyards. Petroleum hydrocarbons are distributed similarly to metals and PCBs, with an additional area of elevation near the southern boundary of NASSCO's leasehold. Concentrations of all chemicals generally decrease with distance from shore.

Two types of analyses were conducted to assess the bioavailability of metals in shipyard sediments. Acid-volatile sulfide (AVS) was measured in surface sediment during Phase 1 to evaluate whether chemical binding of divalet metals with AVS could limit the bioavailability of those metals. Concentrations of AVS were not high enough to bind all divalent metals present in the sediments. However, the presence of AVS is likely to partially limit the bioavailability of metals. Electron microprobe analyses of sediment particles were conducted to evaluate the mineral forms present in the sediment and the physical locations of metals with respect to those minerals. Copper and chromium were analyzed as representative metals. Copper was found to be primarily associated with the mineral chalcopyrite, and chromium was found to be exclusively associated with the mineral chromite. Both minerals are ores of the respective metals, and the minerals themselves were associated with particles of smelter slag in the sediment. In the mineral form, these metals make up an integral part of the mineral's crystalline structure, are consequently not subject to sorption-desorption reactions, and so would not be bioavailable to organisms living in or near the sediments. Because concentrations of all other metals are strongly correlated with concentrations of copper, the other metals are also likely to be present in the smelter slag and to have correspondingly low bioavailability.

The relative abundances of different PAH compounds in sediment were evaluated to determine whether the composition of PAH as a whole was characteristic of a petroleum source (petrogenic) or a combustion source (pyrogenic). This analysis indicates that PAH at the shipyards and at the reference stations is predominantly pyrogenic in origin. This finding is consistent with previous investigations of PAH composition in San Diego Bay. PAH of petrogenic origin appears to be present at one station in the Southwest Marine leasehold, and possibly also present at low levels at three other stations.

Sediment pore water was analyzed for all shipyard-specific chemicals to allow calculation of equilibrium partitioning-based candidate cleanup levels. Equilibrium partitioning theory is based on the assumption that a proportional relationship exists between chemical concentrations in pore water and sediment. A non-proportional relationship between chemical concentrations in pore water and in sediment was observed for most chemicals. Regression of pore water concentrations on sediment concentrations resulted in a positive intercept, indicating that positive pore water chemical concentrations would be expected even in the complete absence of the chemical in sediment. The positive intercept is likely due to the presence of fine suspended or colloidal material in the pore water samples that could not be removed by centrifugation (pore water was extracted using centrifugation, as recommended by the U.S. Environmental Protection Agency [EPA]). As a consequence, measured concentrations of chemicals in pore water are likely to be biased high. In the case of copper, the intercept value was higher than the California Toxics Rule values for marine life at all shipyard and reference stations. Because of the apparent bias and the variability in the relationships between sediment and pore water, the data could not be used to develop potential sediment cleanup levels based on the equilibrium partitioning approach.

Effects on Aquatic Life

Three types of toxicity tests were conducted on surface sediment samples, in conjunction with benthic macroinvertebrate community analyses and sediment chemistry measurements. Synoptic measurement of toxicity, benthic communities, and sediment chemistry is known as the sediment quality triad approach, and is a standard method for assessing the relationship between chemical concentrations and effects on aquatic life. Triad analyses were carried out at 30 stations. The toxicity tests measured amphipod survival, echinoderm egg fertilization, and bivalve larval development. Presence of potentially adverse effects was assessed based on statistically significant ($p<0.05$) differences between shipyard samples and samples in the final reference pool. Mean amphipod survival at the shipyards ranged from approximately 70 to 95 percent, and six of 30 locations evaluated were statistically different from the final reference pool samples. Mean echinoderm fertilization ranged from approximately 60 to 98 percent, and none of the shipyard locations were statistically different from the final reference pool samples. Mean bivalve development at the shipyards ranged from approximately 2 to 85 percent. Large variations, as much as 70 percent, were seen among the replicates of a single sample for the bivalve test, and large variations were also seen within and among the reference stations. Because this test used an experimental variation of the standard method, and because of inconsistencies in the results, the results of the bivalve test are not considered to be as reliable as the other toxicity tests. Twelve shipyard stations had bivalve test responses that were statistically different from the final reference pool samples; only one of these stations also had a significant amphipod response.

Toxicity test responses (percent survival, percent fertilization, and percent normal development) were evaluated with respect to the concentrations of shipyard-associated chemicals in the same samples. There were no statistically significant correlations between any of the shipyard-associated chemicals and any of the toxicity test responses. Four sediment samples collected during Phase 2 were analyzed for organochlorine pesticides; chlordane and DDT isomers were detected in all of these samples. Statistically significant correlations were found between pesticide concentrations and both echinoderm and bivalve toxicity responses.

Laboratory bioaccumulation tests were conducted using sediment collected during Phase 1. Uptake of sediment chemicals by a clam (*Macoma nasuta*) was measured. Tissue concentrations of most metals and organic compounds were higher in clams exposed to sediment with higher concentrations of these chemicals. However, no relationship between chemical concentrations in sediment and tissue was observed for cadmium, chromium, nickel, selenium, silver, or PCTs. Bioaccumulation relationships for arsenic and zinc, although statistically significant, were each controlled by only a single data point.

Benthic macroinvertebrate communities were evaluated using two methods: sediment profile images and collection and enumeration of benthic macroinvertebrates. Sediment profile photographs (vertical cross-sections through the sediment) were taken at 100 locations throughout both shipyards. The photographs were evaluated to determine whether the benthic macroinvertebrate species represented a pioneering community, a mature community, or some intermediate form. The photographs were also evaluated to determine the depth of the apparent redox potential discontinuity (RPD depth), as an indicator of sediment bioturbation. Species characteristic of mature benthic communities were found at almost all locations in both shipyards, often in combination with pioneering species. The combination of community types may indicate that some disturbance of the sediment surface is occurring, but that it is not strong enough to affect the deeper-living species that are characteristic of mature communities. Only pioneering species were found in some locations, particularly along the shore between NASSCO Berths V and VI, and near the northern boundary of the Southwest Marine leasehold. The presence of pioneering species indicates that some form of disturbance is present at these locations. NASSCO Berths V and VI are used for engine tests, and the benthic community conditions determined from the sediment profile images indicate the effects of physical disturbance of the sediment resulting from the engine tests. Similarity of apparent RPD depths indicated that bioturbation is occurring at all shipyard locations, and at an intensity comparable to the reference stations. Overall, the benthic communities at the shipyards consisted of deep-burrowing forms characteristic of healthy assemblages that are not subject to significant stress.

Benthic macroinvertebrates were collected and enumerated at the 30 triad stations during Phase 1. Multivariate techniques (cluster analysis, multidimensional scaling) and univariate statistical tests were used to evaluate the benthic community by comparisons with reference stations. The benthic community at one of the reference stations sampled was dominated by an invasive crustacean that changed the community structure, and data from that station was not used for comparative purposes. The other four reference stations, which were located within a range from the mouth of the bay to an area south of the shipyards, showed a gradient in community composition throughout the length of the bay. This gradient most likely reflects natural differences in physical and biological conditions throughout the bay in flushing, temperature, sediment type, and proximity to the Pacific Ocean. Notwithstanding the limitations of this reference data set, shipyard and reference stations were very similar in some respects, such as the relative abundances and relative numbers of species in major taxonomic groups. For example, the relative abundances of polychaetes, crustacea, molluscs, and other taxa were 69, 12, 9, and 9 percent at the reference stations, 68, 12, 15, and 5 percent at NASSCO, and 65, 15, 15, and 4 percent at Southwest Marine. Total numbers of benthic organisms per square meter ranged from 4,400 to 9,870 at the reference areas (excluding the station dominated by an invasive crustacean), 2,800 to 8,600 at NASSCO (excluding a station at the mouth of Chollas Creek), and 3,160 to 31,800 at Southwest Marine. Statistical tests on eight individual benthic indices identified some differences between shipyard stations and reference stations. Differences were categorized as absent, minor, moderate, or major. Of the 30 stations evaluated, 4 had minor differences, 2 had moderate differences, and 7 had major differences from reference station conditions. Stations categorized as having major differences from reference conditions generally had benthic macroinvertebrate abundances that were approximately 50 percent of average reference conditions. Three of the stations with major differences were off of the municipal storm drain outfall in the Southwest Marine leasehold, one was located between NASSCO Berths V and VI where engine testing occurs, and one was located near the mouth of Chollas Creek.

The newly developed southern California bays benthic response index (BRI) was also evaluated for shipyard stations and final reference pool samples. Because of wide variations in pollution tolerance scores used in the BRI, as well as other issues related to the development method, the BRI is considered to be a less reliable indicator of benthic community conditions than established statistical techniques.

In addition to the absence of any significant correlation between toxicity test responses and shipyard chemicals, there is also no correlation between benthic macroinvertebrate abundance or richness, and any shipyard chemicals except for selenium and petroleum hydrocarbons. Selenium was undetected in two-thirds of the sediment samples, and the detected values are at the quantitation limit; the statistically significant correlation is therefore considered to be an artifact of the distribution of selenium values. The strongest correlation—that of benthic macroinvertebrate richness with residual range organics—explains at most only 8 percent of the observed variation in species richness. The absence of statistically significant correlations strongly implies the absence of a cause-and-effect relationship between shipyard chemicals and biological effects. Biological effects are more consistently, and generally more strongly, correlated with the fraction of fine particles in the sediment than with any shipyard chemicals. Observed biological effects may be caused by non-shipyard chemicals that are associated with fine sediment. Pesticide concentrations at the four stations measured in Phase 2 were significantly correlated with benthic macroinvertebrate richness and benthic bivalve abundance (as well as with toxicity test results). Although definitive conclusions cannot be drawn from the limited amount of pesticide data collected in this study, pesticides are a plausible cause for biological effects at the shipyards because Chollas Creek is a known source of pesticides, and the plume of toxic discharges from Chollas Creek covers the shipyards.

Physical disturbance of shipyard sediment is another likely cause of biological effects. Sediment cores taken in Phase 2 at some locations show a homogeneous grain size distribution in near-surface sediment, a condition indicative of sediment mixing. All locations with apparent physical disturbance are in active areas of the shipyards, and disturbance is evident at five of the six locations where altered benthic communities are the only biological effect.

Spotted sand bass collected from four locations near the shipyards and a reference area were evaluated for lesions and other histopathological conditions. Grossly visible external conditions were evaluated as well as microscopic cellular changes to gill, gonad, liver, and kidney tissue; a total of 70 types of lesions and other conditions were evaluated. None of the serious liver lesions (e.g., neoplasms and pre-neoplastic lesions) typically found in fish from contaminated sediment sites was found at the shipyards. Statistical comparisons of site and reference locations ($p<0.05$) indicated that three types of lesions (of the 70 evaluated) were elevated at some shipyard location relative to reference, and six types of lesions were elevated at the reference station relative to the shipyards. Most of the conditions identified have an uncertain effect on fish health. Indices of fish growth and condition (length at age and weight at length) were calculated and used to compare shipyard and reference conditions. Both indices of fish health indicated that fish at shipyard locations and the reference location were similar. Neither growth nor health of fish at the shipyards are adversely affected relative to reference conditions.

Bile was collected from spotted sand bass to evaluate fish exposure to PAH. Fish metabolize PAH in the liver, and the breakdown products can be detected in the fish bile. Ten composite samples from each of four locations at the shipyards and from a reference area were evaluated. No statistically significant differences in PAH breakdown products in fish bile were found at the shipyards relative to the reference location.

Risks to Aquatic-Dependent Wildlife

Chemical concentrations measured in indigenous biota (mussels, eelgrass, and large and small fish) were used to assess risks to aquatic-dependent wildlife that may feed in the vicinity of the shipyards. Ecological risks were evaluated for six representative species: East Pacific green sea turtle, California least tern, California brown pelican, western grebe, surf scoter, and California sea lion. A standard food-web model, consistent with EPA guidance for ecological risk assessment, was used to evaluate exposure. Literature data were used for ingestion rates and dose-responsiveness information, and site-specific data were used for chemical concentrations in food and habitat utilization. Fish collected from four locations at the shipyards were used to model the diet of most of the bird and mammal species, benthic mussels collected from each shipyard were used to model the diet of the surf scoter, and eelgrass collected from each shipyard was used to model the diet of the sea turtle. Because fish were collected from four locations in and adjacent to the shipyards, separate risk estimates were calculated for each location. The results indicate that no shipyard chemical, in any location, exceeds exposure levels of concern (no-effect levels) for any representative aquatic-dependent wildlife species. No hazard quotient—the ratio of actual exposures to no-effect exposure levels—was greater than 0.016, indicating that any risks to aquatic-dependent wildlife feeding near the shipyards are negligible. Consequently, current conditions at the shipyards are protective of aquatic-dependent wildlife beneficial uses.

Risks to Human Health

Chemical concentrations measured in indigenous lobsters and fish were used to assess risks to human health. Although public access to the shipyard leaseholds and immediately adjacent waters is strictly prohibited and enforced by U.S. Navy (Navy) personnel, a human health risk assessment was conducted assuming that the shipyard leaseholds were accessible for fish and shellfish harvesting. Fish collected from four locations at the shipyards, and lobsters collected from each shipyard, were used to model human health risks through seafood consumption. The risk assessment model used for this analysis is consistent with state and EPA guidance. Ingestion of both fish and shellfish was evaluated, using chemical concentrations measured in spotted sand bass fillets and lobster tissue. Consumption rates for high-end consumers were used, as recommended by the Office of Environmental Health Hazard Assessment. Both cancer and noncancer risks were evaluated. Cancer risks calculated using maximum tissue chemical concentrations at the shipyards were well below the target human health risk level of 1×10^{-5}. Noncancer risks calculated using maximum tissue chemical concentrations at the shipyards were well below the target hazard index of 1.0. Consequently, current conditions at the shipyards are protective of human health beneficial uses.

Cleanup Levels

Because the aquatic life beneficial use is the only one of the beneficial uses that is currently affected at the shipyards, sediment cleanup levels were developed to address this beneficial use. Cleanup levels were developed for indicator chemicals: those chemicals that had any statistically significant relationship to toxicity test results, benthic macroinvertebrate community measures, or tissue chemistry concentrations in the bioaccumulation tests. This suite of chemicals consisted of arsenic, copper, lead, mercury, zinc, tributyltin, PCBs, PAH, diesel-range organics, and residual-range organics. Most of these chemicals had a statistically significant relationship only to tissue measured in the laboratory bioaccumulation tests. No-effect levels were calculated for each chemical. The no-effect level is the highest concentration of a chemical at which no adverse biological effect is observed. The no-effect concentration is referred to as the apparent effects threshold (AET). No-effect levels were calculated for each of the three toxicity tests, for moderate to major benthic community alterations, and for any benthic community alteration. Because there is no relationship between shipyard chemicals and biological effects, the AET for three out of four of these measures of biological effects corresponded to the highest concentration in the data set for each chemical. Only AET values for any benthic alteration were lower than the maximum concentration for most chemicals. These benthic AET values are the lowest AET (LAET) values for all measures of biological effects, and make up the candidate cleanup levels for protection of aquatic life.

Results of the toxicity tests and benthic macroinvertebrate assessments were combined into an overall assessment of the likelihood of adverse aquatic life effects at each station. The set of stations with chemical concentrations exceeding the LAET was compared to the set of stations having a high or very high likelihood of some aquatic life effect. This contrast allowed the predictive performance of the LAET criteria to be evaluated. Although use of the LAET as a cleanup criterion would result in the removal of the highest concentrations of metals and organic compounds at the shipyards, the sensitivity of this criterion is relatively low: only 11 to 14 percent of the locations with likely aquatic life effects exceed the LAET. This is consistent with the absence of a correlation between shipyard chemicals and biological effects. The overall reliability of the LAET criteria (proportion of correct predictions of effects or no effects) is higher, between 67 and 73 percent, because aquatic life effects are not likely at most of the shipyard stations.

Of 66 stations at the shipyards, 20 have some exceedance of the LAET in surface or subsurface sediment. At 10 of these stations, the only chemicals exceeding LAET are petroleum hydrocarbons. Because toxic constituents of petroleum hydrocarbons are degraded in weeks or months, and because no toxicity is associated with petroleum hydrocarbons in shipyard samples, exceedances of the LAET for petroleum hydrocarbons are not accurate predictors of long-term biological effects.

Causation analysis shows that chemical and biological effects data from the shipyards contradict the hypothesis that high concentrations of shipyard chemicals are responsible for adverse biological effects. Two separate causative hypotheses were tested: 1) that shipyard chemicals are the only (exclusive) cause of biological effects, and 2) that shipyard chemicals are one of several (non-exclusive) causes of biological effects. Tests of the logical consequences of these hypotheses show that both are false. Non-shipyard chemicals, such as pesticides, may instead be partly or wholly responsible for observed adverse biological effects. In addition, physical factors such as sediment disturbance and fine grain size may play a partial role in causing apparent adverse biological effects. For example, the combination of elevated chemical concentrations (LAET exceedance) and fine-grained sediment could be a non-exclusive cause of biological effects. However, there is only one station for which the combination of elevated shipyard chemicals and fine-grained sediment is associated with adverse aquatic life effects, and eight stations for which other causes must be responsible. The (partial) association of effects with percent fines may be spurious, however, because almost all of the final reference pool stations, which were used to assess biological effects at the shipyards, had percent fines lower than 65 percent.

Because the no-effect levels for all toxicity tests and for moderate to major alterations of the benthic community are all equivalent to the highest concentrations of shipyard chemicals in the data set, no remediation based on shipyard chemical concentrations is necessary to protect these aquatic life beneficial uses. Use of the LAET for shipyard chemicals as a cleanup level would only potentially eliminate some minor alterations in benthic macroinvertebrate communities. Because other contaminant sources such as Chollas Creek and municipal storm drains are uncontrolled, even such minimal improvement may not be achieved with any remediation to these levels.

Effectiveness and Technical and Economic Feasibility of Remedial Alternatives

Four possible remedial alternatives were identified and evaluated as part of a feasibility study. These alternatives were monitored natural recovery; dredging of areas with chemicals other than petroleum hydrocarbons above the LAET, with upland disposal; dredging of areas with chemicals other than petroleum hydrocarbons above the LAET, with disposal in a nearshore confined disposal facility; and dredging of all sediments above the 95 percent upper prediction limit for final reference pool chemistry values. These alternatives were evaluated with respect to short- and long-term effectiveness (i.e., changes in beneficial uses), technical feasibility, and economic feasibility. Criteria for evaluation of technical feasibility were compliance with applicable or relevant and appropriate requirements, implementability, and cost. Criteria for evaluation of economic feasibility were effects on shipyards and dependent economic activities, impacts on neighborhoods, and effects on utilization of aquatic life.

Because the biological effects detected at the shipyards are not caused by shipyard chemicals, and because ongoing sources of non-shipyard chemicals are uncontrolled, the long-term effectiveness of all alternatives is similar with respect to protection of aquatic life and aquatic-dependent wildlife. Considering them all, beneficial uses are currently at approximately 95 percent of ideal values. Remedial alternatives will result in long-term improvement of only 1 to 3 percent. Some long-term adverse effects may result from dredging of eelgrass beds and increases in water depth that prevent the reestablishment of eelgrass. The short-term effectiveness of the dredging alternatives is lower than for the natural recovery alternative, and is lowest for dredging to meet final reference pool chemistry values, because of the resulting destruction of existing healthy benthic macroinvertebrate communities and eelgrass beds. Both short-term and overall effectiveness of the dredging alternatives with respect to protection of human health are poor. Current conditions at the shipyards do not pose significant human health risks, whereas dredging and disposal activities will increase the risk of accidental injury and death to project personnel.

Achievement of technical and economic feasibility would require substantial tradeoffs for the dredging alternatives. Dredging cannot be performed continuously within the shipyards because of limitations on access to berth space and on movement of dredging equipment within the shipyards. Berths and dry docks are generally fully utilized, primarily by Navy vessels, and force protection measures required by the Navy prohibit the presence of non-mission-essential vessels. Interruption of current contractually obligated shipyard work would cause breaches of contracts and have highly adverse effects on the viability of the shipyards, and concomitant adverse effects on the local economy. Delay of construction or repair activities will generally affect approximately 1,000 local employees. Delays could also result in defaults on contract work, potentially resulting in millions of dollars of damages against the shipyards, as well as impacts to national goals related to naval readiness and replacement of the single-hull tanker fleet. To avoid these potentially significant adverse economic impacts, dredging equipment would have to be redeployed each time that an appropriate area of the shipyards became accessible. Because berth space is generally fully utilized, and is currently scheduled for several years into the future, completion of dredging in the interior portions of the shipyards would require years. The necessary prolongation of dredging activities would also have adverse economic impacts as a result of the large increase in costs represented by multiple redeployments and by long-term sequestration of land used for sediment dewatering. Because of the patchy distribution of the dredged areas that would result, and tidal currents and the physical disturbance that would occur within the leaseholds, sediment is very likely to be redistributed from undredged areas into dredged areas. Sediment redistribution compromises the technical feasibility of dredging. For the alternative of dredging to achieve final reference pool chemistry conditions, both technical and economic feasibility cannot be achieved simultaneously, and this alternative is therefore not implementable.

A comparison of all alternatives with respect to effects on beneficial uses, technical feasibility, and economic feasibility identifies monitored natural recovery as the preferred alternative. This conclusion is supported by the following findings of this study:

- Beneficial uses overall are currently at approximately 95 percent of ideal (unimpaired) levels. The incremental benefit of even the most extensive sediment removal would be minor, on the order of only 1 percent.

- Because of ongoing contaminant releases from Chollas Creek and municipal storm drains, and the indefinite continuation of physical disturbance, long-term conditions following active remediation are expected to be similar to current conditions.

- As a result of practical limitations on access in the working shipyards, and security restrictions around Navy vessels, active remediation alternatives would not be both technically and economically feasible.

Active remediation would not produce any clear long-term improvement in beneficial uses relative to current conditions. The active remediation alternatives also have adverse effects on human health risk. Monitored natural recovery is equivalent to or better than all other alternatives, and is therefore the preferred alternative.

1 Introduction

National Steel and Shipbuilding Company (NASSCO) and Southwest Marine Inc. shipyards have conducted a sediment investigation in response to Resolutions No. 2001-02 and 2001-03, adopted by the California Regional Water Quality Control Board, San Diego Region (RWQCB), on February 21, 2001. Regional Board staff issued guidelines for conducting the investigation on June 1, 2001 (RWQCB 2001). The investigation included two phases of fieldwork, which were conducted in 2001 and 2002. The overall work plan for the detailed sediment investigation (Exponent 2001) describes the major components of the investigation. The supplementary Phase 2 field sampling plan (FSP) (Exponent 2002) describes additional details of the second round of sampling. This document presents the results of field sampling and analyses of those data with respect to potential effects of sediment contamination on aquatic life, aquatic-dependent wildlife, and human health at the shipyards.

1.1 Objectives of the Current Investigation

The objectives of the current investigation are to:

1. Determine the nature and extent of sediment contamination resulting from historical waste discharges at the shipyard sites
2. Identify any limitations on beneficial uses of San Diego Bay associated with sediment chemicals discovered at the sites
3. Derive appropriate remedial alternatives to address shipyard-related sediment chemicals.

These objectives respond to Resolutions 2001-02 and 2001-03 and the specific information requirements of Regional Board staff as specified in Water Code Section 13267 (Robertus 2001a, pers. comm.), and in a manner consistent with State Water Resources Control Board (SWRCB) Resolution 92-49. These objectives are intended to protect beneficial uses of San Diego Bay at the shipyards, considering all the demands being made, and to be made, on those waters. The specific beneficial uses to be protected, those considered to be most sensitive to sediment contamination (RWQCB 2001), are:

- Aquatic wildlife—specifically, the benthic community
- Aquatic-dependent wildlife—specifically, birds, mammals, and reptiles consuming fish and other aquatic organisms
- Human health—specifically, consumption of fish and shellfish.

1.2 Summary of Regional Board Directives

RWQCB and its staff have provided directives and guidance to the shipyards throughout the planning and execution of this project. The major elements of this oversight, including meetings between Regional Board staff and the shipyard staff and their consultants, are as follows, in chronological order:

1. February 21, 2001—Resolutions No. 2001-02 and 2001-03
2. June 1, 2001—California Water Code Section 13267 letter directing the shipyards to carry out a sediment study, with attached guidance for conducting the sediment investigation (RWQCB 2001).
3. August 3, 2001—Meeting to discuss project work plan and Phase 1 activities.
4. October 12, 2001—Meeting to discuss other aspects of project work plan.
5. December 24, 2001—California Water Code Section 13267 letter with changed and additional requirements for the Phase 2 ecological risk assessment.
6. January 29–30, 2002—Meeting and public workshop to discuss results of Phase 1 sampling and analyses.

7. June 18, 2002—Meeting and public workshop under the direction of Regional Board staff to discuss project findings and status.
8. March 6, 2002—Letter containing rationale for previously identified background chemical concentrations, identifying and providing a rationale for a different set of background chemical concentrations, and requiring the shipyards to sample additional offsite locations throughout San Diego Bay (Robertus 2002a, pers. comm.).
9. June 20, 2002—Conference call in which Regional Board staff requested that additional sampling and chemical analyses be performed, and specified that alternate methods be used for Tier 1 ecological risk screening.
10. July 16, 2002—California Water Code Section 13267 letter directing the shipyards to conduct evaluations of potential effects on fish health (follow-up to December 24, 2001, letter) (Robertus 2001b, pers. comm.).
11. August 28, 2002—Electronic mail from Regional Board staff directing the shipyards to collect and analyze fish bile during Phase 2 (Alo 2002b, pers. comm.).
12. September 19, 2002—Telephone call from Regional Board staff specifying changes to Phase 2 fish sampling and target species for the ecological risk assessment (Alo et al. 2002).
13. October 18, 2002—California Water Code Section 13267 letter directing the shipyards to analyze pore water for polycyclic aromatic hydrocarbon (PAH) (Robertus 2002c, pers. comm.).
14. October 25, 2002—Electronic mail from Regional Board staff specifying a water quality objective to be used for the protection of aquatic life from PAH in pore water (Alo 2002c, pers. comm.).
15. October 27, 2002—Electronic mail from Regional Board staff requesting that the shipyards also analyze pore water for total organic carbon (TOC; Alo 2002d, pers. comm.).

16. December 12, 2002—Meeting and conference call between Regional Board staff, shipyard staff and consultant, U.S. Navy (Navy) staff, SCCWRP staff, and resource agency staff discussing possible revisions to the definition of background conditions.

17. January 22–23, 2003—Meeting between Regional Board staff, shipyard staff and consultant, Navy staff, SCCWRP staff, and resource agency staff discussing possible revisions to the definition of background conditions.

18. June 9, 2003—Specifications of the "final reference pool" to be used for interpretation of Phase 1 and Phase 2 data (Barker 2003).

19. June 27, 2003—Meeting between Regional Board and shipyard staff to discuss the shipyards' comments on the final reference pool. Agreement was reached to proceed with production of this report.

The sampling and data analyses described in this report follow these specifications.

1.3 Site Setting and History

The NASSCO and Southwest Marine leaseholds are located on the eastern shore of San Diego Bay, approximately halfway from the mouth of the bay to its inner end (Figure 1-1). The shipyards provide ship construction and repair services to both commercial customers and the Navy. The shipyards are a major part of the considerable physical and social infrastructure (regulatory compliant facilities, tradespeople, tooling, subcontractors, vendors) necessary to maintain San Diego's status as a vital West Coast homeport for the Navy.

The NASSCO and Southwest Marine shipyards are physically adjacent, have a similar range of water depths, and lie within the same hydrologic and biogeographic regime. No important differences are known to exist in surface water conditions, sediment type, sediment transport, aquatic wildlife, or aquatic-dependent wildlife at the two sites.

Piers and floating dry docks are present at both shipyards (Figure 1-2). All piers are built on pilings, with the exception of the largest pier at NASSCO (containing Berths 1 and 2 in Figure 1-2), which is solid and effectively separates the southern part of the NASSCO shipyard from the remainder of the leasehold.

The outer boundary of the shipyard leaseholds corresponds to the outer limit of the piers, but a security boom required by the Navy extends some distance farther into San Diego Bay to prevent public access to the shipyard sites.

1.3.1 NASSCO

The NASSCO site covers approximately 126 acres of tideland property, which includes approximately 80 acres of upland area and 46 acres of water area, leased from the San Diego Unified Port District. A sheet pile bulkhead and a seawall form the boundary between land and sea for most of the facility shoreline. NASSCO has the following operational features:

- Two inclined building ways, 950 ft long and 108 ft wide
- Eight ship berths ranging in length from 600 to 1,000 ft
- A 1,000-ft long by 170-ft wide graving dock
- An 820-ft long by 136-ft wide floating dry dock.

Ship repair and construction activity at NASSCO takes place primarily in the central part of the leasehold. Outfitting of newly constructed vessels takes place at Berths II, V, and VI, primarily V and VI. Ship repair takes place primarily at Berths II, III, and IV; some repair activities also are performed at Berths IX and X. The western part of the NASSCO leasehold, adjacent to its boundary with Southwest Marine, is currently not used for in-water activities.

NASSCO's customers for new construction work include both the Navy and commercial customers. NASSCO has a total employment of approximately 3,400 workers, of which approximately 2,500 are employed on its new ship construction activities. NASSCO is the only remaining shipyard on the West Coast devoted to the construction of large commercial and military vessels. NASSCO is currently performing multi-year construction contracts for both military and commercial customers. For the Navy, NASSCO is under a long-term contract to deliver T-AKE Class ships, which deliver ammunition, provisions, stores, spare parts, potable water, and petroleum products to armed forces conducting national defense operations throughout the world. The T-AKE will replace the aging T-AE ammunition ships and T-AFS combat store ships that are nearing the end of their service lives. NASSCO is also building four 1.3 million barrel capacity double-hulled commercial tankers for BP to transport crude oil from Valdez, Alaska, to oil refineries on the West Coast. These ships contain state-of-the-art environmental controls and will replace single-hulled tankers that must be phased out to meet the requirements of the Oil Pollution Act of 1990, enacted in response to the Exxon Valdez spill. The ships under construction at NASSCO play vital roles in meeting national defense and national environmental goals. Delays or interruptions in the delivery of these ships would have potentially broad consequences affecting these important national goals.

Ship repair and modernization work at NASSCO is carried out almost exclusively for the Navy. NASSCO performs work at its shipyard on various classes of Navy ships homeported in San Diego, including LHA/LHD amphibious assault ships, CG -47 guided missile cruisers, DD-963 destroyers, and FFG-7 guided missile frigates. Work on these ships is scheduled several years in advance to coincide with ship deployments. Availability of shipyard berths and dry docks is largely controlled by the need to meet these schedules, and to perform additional unscheduled work as required (e.g., as a result of deployments in response to international conflicts, such as those in Afghanistan and Iraq). Staffing on military vessels undergoing maintenance and repair includes not just shipyard employees, but large numbers of subcontractor personnel and Navy Advance Installation Teams (AITs) that perform specialized onboard ship modernization activities. Approximately 500–700 shipyard workers, subcontractors, and AITs are engaged in work on vessels under repair. As the only full service shipyard remaining on the West Coast, NASSCO is a strategic asset to the Navy.

Force protection measures are required for Navy vessels and prohibit non-mission-essential vessels from approaching Navy ships. A security boom prevents unauthorized vessels from approaching closer than 300 ft to the NASSCO shipyard.

The NASSCO site has been used for a variety of industrial operations since the first decades of the 20th century. A history of site usage includes the following (Brody 2003, pers. comm.):

- **Early 1920s or before—**Standard Oil Company of California (now Chevron) used the western portion of the site for fuel storage and marine terminal operations beginning sometime prior to 1921.
- **Mid-1930s—**The upland portion of the leasehold was created by filling tidelands between Standard Oil Company facilities located at the foot of Schley Street and the 28th Street Pier.
- **Approximately 1937 through 1940—**Warren Boat Co. of San Francisco used the site to build small power cruisers.
- **1940–1942—**Martinolich Shipbuilding Company took over and increased operations at the former Warren Boat Co. operation.
- **1941–1959—**Standard Oil Company leased a portion of the site adjacent to Martinolich for operation of retail fuel services station.
- **Approximately 1941–1944—**Robbins Marine Engine Co. conducted boat and machine repairs operations on part of the site west of the area used by Martinolich.
- **1942–1948—**Lynch Shipbuilding Co. acquired the Martinolich facility and conducted ship construction work at the site.
- **1944–1960—**National Iron Works (NIW) took over and enlarged the former Robbins facility. During this period, NIW changed its name to National Steel and Shipbuilding Corporation, which, although similarly named, is not the same entity as the current NASSCO. In 1948, NIW took over the Lynch

Shipbuilding facility; in 1955, it took over a portion of the 28th Street Pier leasehold; in 1957, it took over the Martinolich facility; and in 1959, it took over the former Standard Oil retail leasehold and another adjacent (Westgate-California) leasehold.

- **Approximately 1940s–1955—**The Navy used the 28th Street Pier for docking and ship repair.
- **1946–1957—**Martinolich leased an area west of its earlier operations for more substantial shipbuilding operations. During some or all of this period, Martinolich leased a floating dry dock from the Navy, which was moored adjacent to the upland facilities.
- **1946–1960—**Peoples Fish Packing Corp. and then Westgate-California Tuna Packing Corp. operated a tuna cannery between Lynch Shipbuilding and NIW.
- **1960–present—**NASSCO acquired the business and physical assets of NIW, and currently operates a shipbuilding and repair facility on the site.

Current shipyard operations at NASSCO incorporate thorough pollution prevention mechanisms to eliminate the possibility of direct releases of contaminants. Best management practices (BMP) were implemented at the shipyard in the mid-1980s. Pollution prevention measures currently in place consist of:

- **Collection systems for surface water and spills—**Curbs, sumps, pumps, and holding tanks collect all rainwater or other liquids released within the shipyard's paved area. All collected material is processed through the shipyard's onsite water treatment facility before it is discharged to the sewer system.
- **Industrial wastewater treatment—**All process water resulting from industrial operations is collected and treated onsite before it is discharged to the sewer system. Prior to the mid-1980s, all bilge and ballast water was

trucked offsite for disposal. Onsite treatment of bilge and ballast water was initiated in the mid-1980s, and the treated water is discharged to the sewer. In 1997, all non-contact cooling water and load test water (seawater pumped from the bay and circulated in pipes and tanks isolated from construction and repair activities) was redirected to the sewer system.

- **Storm water filtration—**As part of a technology development research program, storm water collected from the NASSCO shipyard is being filtered to remove suspended particles and dissolved metals. Capture of first-flush storm water from high-risk areas (dry dock, graving dock, paint and blasting areas) was initiated by NASSCO in the early 1990s. Capture of first-flush storm water was extended to additional areas of the yard in 1997, and since 2000, all storm water has been captured, treated, and discharged to the sewer system.

- **Training—**Ongoing pollution prevention training programs are used to establish and maintain high standards of environmental awareness by shipyard employees and subcontractor personnel.

NASSCO was the first commercial shipyard in the United States to be ISO-14001 certified for their environmental management system. The effectiveness of these measures was recognized by awards from the California Environmental Protection Agency in 2002 and from the U.S. Environmental Protection Agency (EPA) in 2003. The latter award was presented for NASSCO's role in development of a guidance document for implementation of environmental management systems in the shipbuilding industry; recommendations of the guidance document were modeled on NASSCO's pollution prevention system.

1.3.2 Southwest Marine

The Southwest Marine site covers approximately 27 acres of tideland property, which includes approximately 10 acres of upland area and 17 acres of water area, leased from the San Diego Unified Port District. A sheet pile bulkhead and a seawall form the boundary between land and sea for most of the facility shoreline; however, there is a small intertidal area where former Marine Railways 2 and 3 were located. Southwest Marine has the following operational features:

- Five piers ranging in length from 257 to 700 ft
- A floating dry dock of 22,000 tons lifting capacity
- A floating dry dock of 4,000 tons lifting capacity.

Ship repair activity at Southwest Marine takes place throughout most of the leasehold, with the exception of the eastern part of the leasehold adjacent to NASSCO. The easternmost pier at Southwest Marine is partly demolished and is not currently used for berthing ships or for any repair activities.

Southwest Marine provides ship repair, alteration, and overhaul services for various government and commercial customers. Southwest Marine's business base includes the repair of Navy small craft and commercial work/fishing boats to the long-term life-cycle maintenance of major classes of Navy surface combatant and amphibious warfare ships. Many of the contracts are multi-ship/multi-year and most require dry-docking services. Specific ship repair programs conducted at the Southwest Marine facility include the following:

- Multi-ship/year phase maintenance for LPD-4, LSD-41/49 Class amphibious warfare ships
- Post-delivery upgrade and repair of San Diego based AEGIS cruisers and destroyers
- Multi-ship/year continuous maintenance of CG-47 Class cruisers and DDG-51 Class destroyers
- Major overhaul and conversion of Military Sealift Command fleet support and special mission ships
- Dry dock repair of special interest commercial vessels.

This work is performed with planning and scheduling requirements that are the same as those discussed previously for NASSCO.

Force protection measures are required for Navy vessels and prohibit non-mission-essential vessels from approaching Navy ships. A security boom prevents unauthorized vessels from approaching closer than 300 ft in the Southwest Marine shipyard.

The subject property has been used for industrial operations since the first decades of the 20th century. The original shoreline of San Diego Bay was filled between 1906 and 1914 to create the land currently occupied by Southwest Marine. Occupants and activities on that land included (Woodward-Clyde 1995):

- **1914 to late 1970s—**The northwest and central portions of the site were used by San Diego Marine Construction Corporation. Blast and paint wastes were discharged directly to San Diego Bay from operations in the upland areas and the dry dock.

- **Approximately 1928 to 1972—**The southern portion of the site was used by Richfield Oil Company and Atlantic Richfield Company, including a fuel pier (Pier 4) and pipeline for fuel transfer operations.

- **1942 to 1984—**The northern portion of the site was used by the San Diego Gas and Electric Company (SDG&E). Four large (8-ft diameter) tunnels connected San Diego Bay to the Silver Gate power plant and were used for cooling water intake and discharge. Two wastewater settling ponds were used from approximately 1952 to 1974 for separation of oil and water pumped from basement trenches in the power plant (ENV America 2002).

- **Between 1950 and 1983—**The southeast portion of the site was used as a cannery by the San Diego Packing Company.

- **Between 1950 and 1983—**The southeast portion of the site was used for freight handling and transshipment by a variety of tenants including Southern

California Freight Lines, ONC Motor Freight Systems, Star & Crescent Ferry Co., Inc., Delta Truck Lines, Air Export International, and Bisher Truck Line.

- **1956 to 1972—**The southern portion of the site, including Pier 5, was used by Diesel Technical Services, Inc. Vessel maintenance and repair may have been conducted.

- **1968 to 1985—**The southern portion of the site was used by National Pump and Injector, evidently for sales of material including diesel engines and hydraulic systems.

- **Early 1950s to mid-1980s—**The eastern portion of the site contained oil and grease storage tanks and a warehouse. Tenants included Carter-Delg Oil Co., National Petroleum Co., and Kendall Motor Oils.

- **Mid-1950s to late 1980s—**The eastern portion of the site was used for storage and transshipment of freight. Tenants included Universal Carloading and Distribution Co., Western Parcel Service, International Forwarding Co., Santa Fe Trail Transportation Co., Ace High Transportation, Bill's Trucking, and Hawaiian Cargo.

- **1979 to present—**Southwest Marine leased the northernmost and westernmost waterfront parcels in 1979. Piers 4 and 5, and adjacent buildings, were occupied by Atlantic Richfield Co. and Diesel Technical Services until the mid-1980s, at which time Southwest Marine acquired the leases for these parcels also. Southwest Marine currently operates a ship repair facility on the site.

Current shipyard operations at Southwest Marine incorporate thorough pollution prevention measures to eliminate releases of contaminants from construction activities or through storm water runoff. Surface water runoff from the dry docks has been collected and discharged to the sewer system since the mid-1980s. In 1997, a storm water diversion system was installed that collects all surface water runoff from the land portion of the site and discharges it to the sewer system. In 1998, Southwest Marine received an "Orchid Award" from the American Institute of

Architects (AIA) for the installation of that system. In 2000, surface water collection systems were also installed on all the piers.

Southwest Marine has developed and maintains a BMP program that includes operational controls for production activities. BMP program implementation is regularly validated through inspections and audits. Weekly BMP training is conducted for all production personnel, and environmental staff provides training to subcontractors and customers.

In 2003, Southwest Marine was presented with an environmental stewardship award by EPA for its key role in the development of an Environmental Management Systems Guide for the shipbuilding and ship repair industry. Southwest Marine helped developed methods to implement pollution prevention programs, compliance procedures, and continuous improvement procedures, and the guidance document reflects Southwest Marine's own internal environmental policy.

1.3.3 Previous Investigations at the Shipyards

Assessments of the spatial distributions of sediment chemicals were carried out in 1997 at NASSCO and in 1998 at Southwest Marine. Both of these investigations were carried out at the direction of the RWQCB, and the purpose of both was to determine areas for possible sediment remediation.

In the 1997 investigation at NASSCO, sediment was sampled from 102 stations. Samples were analyzed for copper and zinc as indicators of the distribution of chemicals associated with the shipyard. The highest chemical concentrations were found near the shore in the central part of the shipyard, principally off the building ways.

The 1998 investigation at Southwest Marine sampled surface and subsurface sediment from 110 stations. Samples were analyzed for copper, lead, mercury, zinc, and polychlorinated biphenyls (PCBs). The highest chemical concentrations were found in several distinct locations along the shoreline at the shipyard.

Sediment sampling is also conducted at both shipyards to fulfill National Pollutant Discharge Elimination System (NPDES) permit obligations. NPDES monitoring data have been collected annually or semiannually at 34 stations near the shoreline for approximately 10 years. Three reference area stations elsewhere in San Diego Bay have also been sampled as part of the NPDES program. Analytes measured during NPDES monitoring included metals, tributyltin (TBT), PAHs, PCBs, polychlorinated terphenyls (PCTs), and petroleum hydrocarbons. The most recent NPDES sampling was in 2000.

Data from all of these prior surveys were used to locate sampling stations during the planning of Phase 1 of the present investigation.

1.4 Conceptual Site Model

A conceptual site model is a representation of the site that illustrates the potential chemical sources, affected media, exposure routes, and receptors. The purpose of the conceptual site model is to clearly illustrate and document the aspects of the site that are of concern. Consequently, it forms the basis for both conducting fieldwork and interpreting results. The conceptual site model for this investigation is shown in Figure 1-3. The categories of receptors shown in this diagram are equivalent to the three major types of beneficial uses to be protected—aquatic life, aquatic-dependent wildlife, and human health.

For potential effects on beneficial uses to exist, four elements must be present: 1) a source, 2) a mechanism of release, 3) a point of contact with the medium (i.e., exposure point), and 4) a route of exposure at the point of contact (e.g., ingestion). The route of exposure may involve several steps, such as transmission through a food web. If any of these elements is missing, the pathway is considered incomplete. Those exposure pathways judged to be potentially complete are of concern and are included in the conceptual site model.

1.4.1 Potential Sources of Chemicals to San Diego Bay

There are multiple potential sources of contaminants to San Diego Bay in the region of the shipyards, including:

- Past activities at the shipyards
- Storm water drains that discharge into the shipyard leaseholds
- Nonpoint surface water discharge through Chollas Creek
- Surface water runoff from the roadway between the properties
- Fill material added to the shoreline
- Accidental releases from ships.

1.4.1.1 Past Activities at the Shipyards

As described in Section 1.3, both shipyard sites have been used for a variety of industrial operations since the first decades of the 20th century. Activities that occurred prior to occupancy of the sites by NASSCO and Southwest Marine in 1960 and 1979, respectively, include several that are likely to have generated and released waste, most particularly previous shipbuilding and repair operations, and petroleum storage and ship fueling operations. In 1913, more than 1,500,000 gallons of oil was spilled and burned on the Standard Oil tank farm adjacent to the shipyards (SDFD 2003). Fire hoses were trained on the blaze for 3 days and nights, and a considerable quantity of petroleum hydrocarbons would have been washed into the bay. Historical shipyard operations are likely to have released sandblasting waste consisting of abrasive material, metal particles, and hull paint, possibly including biocides. Petroleum storage and fueling operations are likely to have released petroleum products through leaks and spillage. The power plant bilge wastes pumped into the settling ponds at the northern end of the Southwest Marine site are likely to have contained machine oils, hydraulic fluid, and possibly PCBs released from electrical equipment. Canneries at both shipyard sites may have discharged quantities of organic waste. Prior to the establishment of storm drain systems, surface water drainage from both shipyard properties may have carried a variety of particulate and dissolved contaminants into the bay.

1.4.1.2 Storm Water Drains

Storm water from upland areas on and beyond the shipyard property is currently discharged at the shoreline through a single pipe at NASSCO and a single pipe at Southwest Marine (Figures 1-4 and 1-5).

The single storm water outfall currently active at NASSCO (SW9) drains city and other industrial properties upland of the leasehold. All storm water from the NASSCO property is currently collected and treated onsite before being discharged to the city sewer system. Prior to 2000, when the storm water collection system was put in place, 10 storm water outfalls at the NASSCO shoreline drained different parts of the shipyard.

The single storm water outfall currently active at Southwest Marine (SW4) drains municipal property upland of the leasehold. Currently, storm water from the Southwest Marine property is collected and discharged to the sewer system. Other outfalls drained portions of the shipyard prior to 1997, when a storm water diversion system was put in place.

Contaminants released through storm water drains may make their way to site sediment through settling of particles containing those contaminants. Those particles may be present in the storm water discharge itself, they may be naturally occurring particles in the bay waters at the site to which the introduced contaminants become adsorbed, or they may be precipitates formed when freshwater is introduced into seawater (the "salting-out" effect).

1.4.1.3 Nonpoint Surface Water Discharge through Chollas Creek

Chollas Creek was identified as a priority hot spot due to the presence of copper, DDT, chlordane, and diazinon in the sediments, and the presence of impacts to aquatic life (RWQCB 1997). Both water in the Chollas Creek outflow and sediment at the mouth of the creek has been found to be toxic. Chollas Creek was placed on the state's 303(d) list of impaired water bodies in 1998.

Contaminants in discharges from Chollas Creek can be introduced to sediments at the shipyards through settling of particles—the same mechanisms as contaminants from storm water discharges.

More recent and detailed investigations have shown that the discharge plume from Chollas Creek (following rainfall of as little as 1.0 cm) blankets both the NASSCO and Southwest Marine shipyards. Surface water in this plume is toxic (echinoderm fertilization less than 80 percent) (Schiff et al. 2003). The plume carries suspended particles, and because of the high affinity of contaminants for particles, most of the toxic chemicals in the plume are likely to be attached to these suspended particles. Mixing of the freshwater plume with the salt water of the bay will reduce the solubility of dissolved material. Slowing of the water velocity in the plume as it leaves the creek channel and enters San Diego Bay will reduce the water's ability to carry a suspended load, and the solids and chemical precipitates will settle to the bay bottom underneath the plume. Chollas Creek is therefore a certain source of toxic chemicals to the shipyard leaseholds and adjoining areas.

1.4.1.4 Surface Water Runoff from the Roadway between the Properties

A public roadway (Sicard Street) forms the boundary between the two shipyards. Runoff from this roadway is uncontrolled and enters San Diego Bay. Contaminants conveyed by such runoff could include petroleum hydrocarbons and PAH from the surface of Sicard Street, Belt Street, Harbor Drive, and from the rail crossing on Sicard Street.

1.4.1.5 Fill Material Added to the Shoreline

The San Diego shoreline in the vicinity of the current shipyards was filled in the 1930s. The characteristics of this fill material at the time of placement are unknown. Before the site was paved, the fill material could have become contaminated by surface spills on the site and by contaminated surface water from adjacent upland areas. The wastewater settling ponds used by

SDG&E on the northern portion of the Southwest Marine site could also have introduced contaminants into the fill. Transport of contaminants from fill material to the bay may have occurred by surface runoff before the site was paved. Under current conditions, groundwater flow is the most likely mechanism by which contaminated fill could affect sediment at the shipyards. The potential for fill material to be a source of contaminants to San Diego Bay is listed in the Bay Protection and Toxic Cleanup Program (BPTCP) addendum report (SWRCB 1998).

1.4.1.6 Releases from Ships

The region of the bay in which the shipyards are located is heavily used by commercial and military shipping. Historically, discharges from ships were not fully controlled, and there is thus a potential for contaminated bilge water, or other deliberate or accidental discharges, to have conveyed petroleum hydrocarbons and other contaminants into the waters of the bay. Contaminants released from vessels may make their way to bay sediments through sorption to particles and subsequent particle settling.

1.4.2 Exposure of Aquatic Life

Bottom-dwelling, or benthic, organisms are the components of aquatic life that are most directly exposed to contaminated sediments. Pathways by which they may be exposed include dermal contact and ingestion (including ingestion of other benthic fauna). Epibenthic fish may also be exposed to chemicals through the ingestion of prey such as invertebrates or other fish, direct contact with sediment, and possibly through the incidental ingestion of sediment that may occur during foraging.

1.4.3 Exposure of Aquatic-Dependent Wildlife

Complete exposure pathways exist for wildlife that are resident in the shipyards or incorporate the shipyards within their foraging area. These potential wildlife receptors include marine reptiles, birds, and aquatic mammals. Marine reptiles may be exposed to chemicals through the ingestion of aquatic vegetation and possibly through the incidental ingestion of sediment. Birds and aquatic mammals also may be exposed to chemicals through the ingestion of prey items, such as shellfish and fish, and possibly through the incidental ingestion of sediment. The relative importance of different exposure pathways to ecological receptors depends upon their life history requirements and feeding habits as well as the physical and biochemical properties of the chemicals.

1.4.4 Exposure of Humans

The only potential human exposure to contaminants in site sediment is through consumption of fish and shellfish that may have bioaccumulated chemicals either directly from site sediments or through the food web. Direct contact with sediments is not a likely exposure pathway because the shoreline consists almost exclusively of riprap, sheet-pile bulkhead, and piers. There is almost no intertidal zone, and no access to sediments. Security measures in place at both shipyards prevent all public access to either the upland property or the in-water leasehold. The security boundary actually extends some distance outside the leaseholds. Under current site usage, there is actually no possibility of public fishing and shellfish harvesting at the shipyards, and no changes are anticipated in site usage. Nevertheless, potential impacts on human health are considered in this assessment without regard to access restrictions.

1.5 Document Organization

This document consists of two major sections: Part 1 is a report on the findings of the field investigation, and Part 2 is a feasibility study that presents an analysis of remedial alternatives. Part 1 discusses the results of the analyses of sediment chemistry, sediment toxicity, benthic macroinvertebrate communities, and other site-specific measurements. Part 1 also contains assessments of potential impairments of aquatic life, aquatic-dependent wildlife, and human health beneficial uses at the shipyards. The results of these assessments are used to develop an effects-based candidate cleanup level that addresses the potential impairments of beneficial uses.

Part 2 of this document contains an evaluation of remedial alternatives for the effects-based cleanup level and other cleanup levels, specifically including chemical levels representative of the final reference pool identified by Regional Board staff. The feasibility study evaluates potential remedial alternatives, including consideration of remedial technologies, remedial scenarios, and disposal options. Remedial alternatives are evaluated with respect to improvements in beneficial uses, technical and economic feasibility, and cost.

Tables and figures referenced in the site investigation and feasibility study sections of this document are compiled at the end of the main text. Data tables and other supporting material are included in the appendices provided as separate volumes.

Part 1

Site Investigation

2 Study Design

Collection and analysis of data for this study was carried out specifically to allow an evaluation of potential limitations on the three beneficial uses of aquatic life, aquatic-dependent wildlife, and human health. Sediment, pore water, and tissue were analyzed for a variety of different chemicals, and additional tests and measurements were made to directly assess potential adverse biological effects. The analyses carried out during this investigation are summarized below. Additional details of the sampling and analysis program, including rationale for sampling locations and field and laboratory methods, can be found in the work plan (Exponent 2001) and the Phase 2 FSP addendum (Exponent 2002).

The chemicals analyzed in this study were those specified by Regional Board staff as potentially associated with shipyard activities (RWQCB 2001), and consisted of the following classes of chemicals:

- Metals
- Butyltins
- PCBs
- PCTs
- PAHs
- Petroleum hydrocarbons.

Chemical analyses were carried out on surface and subsurface sediment, sediment pore water, invertebrate tissue used for bioaccumulation testing, and fish and lobsters collected from the shipyards. A complete list of analytes, including physical characteristics as well as toxic chemicals, is shown in Table 2-1. When any chemical was not detected, the concentration was reported at the quantitation limit.

Five reference stations throughout San Diego Bay were selected by Regional Board staff to be used for this study (RWQCB 2001; Bay 2001a,b); the locations of these stations are shown in Figure 2-1. Sediment for chemical, toxicological, and bioaccumulation testing was collected at all of these stations. Reference samples of indigenous biota were collected at the locations shown in Figure 2-2. Additional sediment was also collected at an expanded list of stations subsequently specified by Regional Board staff; the locations of these stations are shown in Figure 2-3.

Sediment was sampled at a total of 66 stations at the shipyards. The locations of these stations are shown in Figure 2-4, and the types of analyses conducted at each are listed in Table 2-2. Biota were sampled at several different locations at each shipyard; these locations are shown in Figure 2-5. Sampling locations were monitored and recorded with a differential global positioning system (GPS) to ensure the highest possible accuracy (±1 meter) and conformity with planned sampling locations.

Fieldwork was carried out in two phases. The first phase was conducted to characterize the overall spatial distribution and quality of sediment conditions, and to allow sampling in the second phase to be targeted at locations or conditions of interest. The second phase was conducted primarily to evaluate the vertical distribution of chemicals in the sediment, chemical concentrations in pore water over a range of sediment chemistry conditions, and chemical concentrations in indigenous biota.

2.1 Phase 1

Phase 1 was carried out in August 2001, and consisted of measurements of the following:

- Surface sediment chemical concentrations throughout the shipyard leaseholds, in locations outside the leaseholds, and at reference areas
- Sediment toxicity at the shipyards in comparison to reference areas

- Benthic macroinvertebrate communities at the shipyards in comparison to reference areas
- Bioaccumulation potential, using sediment from selected locations.

Sediment chemistry, toxicity, and benthic macroinvertebrate analyses (the sediment quality triad) were all carried out at the same locations. Sediment chemistry was also measured at additional locations outside the shipyard leaseholds, both to the northwest of Southwest Marine and between the outer boundaries of the leaseholds and the shipping channel. Locations to the northwest of Southwest Marine were sampled to assess whether there is a spatial trend of chemical concentrations near the adjacent San Diego Gas and Electric discharge. Locations between the leaseholds and the shipping channel were sampled to assess spatial trends of chemical concentrations away from the shipyards. All sediment samples collected during Phase 1 were surface sediment (0–2 cm). Sediment was collected with a 0.1-m^2 stainless-steel van Veen grab sampler.

Prior to sediment sampling during Phase 1, sediment profile imaging (SPI) photographs were taken throughout the shipyards. SPI photographs are taken with a camera mounted above a prism that penetrates into the sediment. One face of this prism is a vertical pane of glass that allows the camera to photograph a vertical cross-section of the sediment. The resulting photographs provide information on physical and chemical conditions in the sediment (e.g., grain size and redox state) as well as a direct assessment of the condition of the benthic fauna. SPI photographs were taken prior to other Phase 1 work to allow station positions to be modified, if necessary, to target any unusual conditions identified in the photographs. SPI photographs were taken at 100 locations at the two shipyards (Figure 2-6) and at each of the five reference stations (Figure 2-7).

Triad analyses were conducted at 30 stations at the two shipyards and at the five reference stations. Sediment used for the three legs of the triad—chemical analyses, toxicity tests, and benthic macroinvertebrate community assessments—was collected synoptically at each station. The sediment used for chemical analyses and toxicity tests at each station was homogenized and then split in the field, to maximize the representativeness of the chemical data to the exposures of the toxicity test organisms.

The following toxicity tests were conducted at Phase 1 triad stations:

- The 10-day amphipod test using *Eohaustorius estuarius* exposed to whole sediment (ASTM 1999). The measured endpoint was percent survival.
- The 40-minute echinoderm fertilization test using the purple sea urchin *Strongylocentrotus purpuratus* exposed to sediment pore water (U.S. EPA 1995a; Carr and Chapman 1992, 1995). The measured endpoint was percent fertilization.
- The 48-hour bivalve development test using the mussel *Mytilus edulis* exposed to whole sediments at the sediment–water interface. The standard bivalve development test (U.S. EPA 1995a) was modified (at the direction of the Regional Board) by the use of additional apparatus to keep the larvae physically separated from the sediment (Anderson et al. 1996, 2001). The measured endpoint was percent normality.

Five laboratory replicates were carried out for each of these tests at each location. Sediment from one station at each shipyard was also used to conduct the 10-day amphipod test using serial dilutions of the sediment, to confirm that the organisms were responding to characteristics of the site sediment.

Benthic macroinvertebrate communities were evaluated by sieving collected sediment through a screen with a mesh size of 1.0 mm. The retained material was fixed with 10 percent formalin and then transferred to 95 percent ethanol at the taxonomic laboratory. Organisms were removed from the sample and identified to the lowest practical taxonomic level. Five replicate field samples were collected and analyzed at each station. All procedures for evaluation of benthic macroinvertebrates followed the recommendations of U.S. EPA (1987). Taxonomic identifications were carried out in accordance with the standards of the Southern California Association of Marine Invertebrate Taxonomists (SCAMIT 2001).

Laboratory bioaccumulation tests were carried out using sediment from four stations at the NASSCO leasehold, five stations in the Southwest Marine leasehold, and all five reference stations. These stations were located to include a range of sediment concentrations of potentially bioaccumulative substances at each shipyard. The bioaccumulation tests consisted of a 28-day exposure of the clam *Macoma nasuta* using standard methods (ASTM 2000). This test species is found naturally throughout San Diego Bay and because it is a surface deposit feeder, it is directly exposed to chemicals in surface sediment. Thirty-five clams were used for each exposure test, and five replicate exposure tests were prepared for each sediment sample. At the end of the 28-day exposure period, the soft tissue of the surviving clams was removed and analyzed.

2.2 Phase 2

Phase 2 was carried out in August, September, and November 2002, and consisted of measurements of the following:

- Chemical and physical properties in sediment cores
- Chemical concentrations in additional surface sediment samples within the shipyards and at Bight ’98 sampling locations outside the shipyards
- Chemical concentrations in pore water
- Chemical concentrations in tissue of eelgrass, lobsters, and fish from the shipyards and a reference location
- The distribution of eelgrass at the shipyards
- Histopathological examination of fish tissues
- PAH breakdown products in fish bile.

Sediment cores were collected during Phase 2 at 18 locations in or adjacent to the NASSCO leasehold and at 20 locations in or adjacent to the Southwest Marine leasehold. Core collection locations are shown in Figure 2-4 and listed in Table 2-2. Cores were collected using either a vibracorer or slide-hammer corer. Corers were driven into the sediment until refusal. Cores were sectioned at 2-ft intervals. Samples for chemical analyses were all taken from a single core; a second core at each location was ordinarily collected for measurements of engineering properties. After each core was brought on board the vessel, it was split lengthwise and sediment properties throughout the core were observed and recorded by a geologist. These core logs are included in Appendix C.

Additional surface sediment was collected during Phase 2 at the locations shown in Table 2-2 and analyzed for the constituents listed in Table 2-1. All surface sediment was collected with a 0.1-m^2 stainless-steel van Veen grab sampler.

Pore water samples were collected at five locations within the NASSCO leasehold, eight locations within the Southwest Marine leasehold, and at all five reference stations. The locations of pore water samples are shown in Table 2-2, and analyses listed in Table 2-1. Surface (0–2 cm) sediment for pore water analyses was collected using a 0.1-m^2 stainless-steel van Veen grab sampler. Because of the large quantity of sediment required, sediment from multiple grab samples was composited before pore water extraction. Pore water was extracted from the sediment by centrifugation on board the sampling vessel, as recommended by U.S. EPA (2001a).

The distribution of eelgrass (*Zostera marina*) at both shipyards was assessed by divers. Eelgrass beds were marked with buoys by the divers, and GPS coordinates of those buoys were recorded by a boat crew (Figures 2-8 and 2-9). Samples of eelgrass for chemical analyses were collected by divers from Bed 1 at NASSCO and Bed 8 at Southwest Marine; reference eelgrass was also collected in the vicinity of Station 2240.

Fish, lobsters, and mussels were collected from locations throughout the shipyards (Figure 2-5). Spiny lobsters (*Panulirus interruptus*) were collected in crab traps; successful collections were only made near piers. Chemical analyses of lobster were performed on both edible tissue (all soft tissue, including the hepatopancreas) and on the entire organism including the shell. Soft tissue was removed in the laboratory prior to analysis. Benthic mussels (*Musculista senhousei*) were collected from the shipyards and from Station 2240 using a van Veen grab sampler. Analyses were performed on the soft tissue of the mussels, which was removed in the laboratory prior to analysis. Fish were collected by trawling and using hook and line both within and outside the shipyard leaseholds and near Station 2240. The fish species collected were northern anchovy (*Engraulis mordax*), Pacific sardine (*Sardinops sagax*), and spotted sand bass (*Paralabrax maculatofasciatus*). Attempts were also made to collect gobies, without success at either site. Chemical analyses were performed on whole bodies of all fishes and also on skin-off fillets of the spotted sand bass (Table 2-1).

Histopathological examinations were made on 50 or more spotted sand bass from locations inside and outside each shipyard leasehold and also from the reference station. Visible external conditions were recorded on the vessel, and the liver, gonads, kidney, and gills of each fish were then removed and transferred to the laboratory for microscopic examination of tissue conditions. The right-side otolith (ear bone) of each fish was also removed for age determination.

During necropsy of the spotted sand bass, bile was removed by micropipette. Bile samples were composited in the laboratory to achieve sufficient material for analysis; a total of 10 composite samples were generated from each location (i.e., 50 samples total). Bile samples were analyzed for fluorescent aromatic compounds and total proteins.

2.3 Study Design Summary

The study design of this investigation has included an extremely comprehensive variety of environmental measurements, and the intensity of the sampling far exceeds that of other investigations in California bays (Table 2-3). The major elements of this investigation directly address the beneficial uses to be protected in San Diego Bay (Table 2-4), and all aspects of the study design were successfully completed. Consequently, the results reported here provide a thorough and accurate assessment of potential impairments of beneficial uses attributable to the NASSCO and Southwest Marine shipyards.

2.4 Contributors to the Study

Collection and analysis of the wide variety of samples used in this investigation required the specialized expertise of numerous organizations and individuals. The following team members (listed in alphabetical order) all made essential contributions to the collection and analysis of the data presented in this report:

- Dr. Larry Allen (California State University, Northridge)—Fish otolith analyses
- Alta Analytical Laboratory—PCB congener and homolog analyses of sediment, pore water, and tissue
- Anchor Environmental—Preliminary remedial design and remedial cost analysis
- AMEC Earth and Environmental—Echinoderm fertilization toxicity tests and bivalve bioaccumulation tests
- Blue Water Engineering—Station positioning
- Columbia Analytical Services—Analyses of sediment, pore water, and tissue for metals, organotins, PCB Aroclors®, PAH, petroleum hydrocarbons, TOC, grain size, and lipid.
- Dr. John Drexler (University of Colorado, Boulder)—Electron microprobe analyses
- Fish Pathology Services (Dr. Gary Marty, U.C. Davis)—Fish histopathology analyses
- Germano & Associates—Sediment profile imaging
- MEC Analytical—Field sampling in Phases 1 and 2, bivalve development toxicity tests, and benthic community analyses
- Northwestern Aquatic Sciences—Amphipod toxicity tests

- Soil Technology, Inc.—Engineering properties of sediment
- Texas A&M Geophysical and Environmental Research Group—PAH metabolite analysis of fish bile.

3 Reference Conditions and Reference Stations

During the course of this project, several different specifications for reference conditions or reference stations have been provided by Regional Board staff. As a consequence of these changes, and particularly the latest set of specifications (Barker 2003), reference area data have been collected that have not actually been used for interpretation of shipyard conditions. In addition, some data interpretations presented in this report have been made using reference area data that were not collected during this investigation. The remainder of this section summarizes issues related to the identification of appropriate reference conditions for the shipyard investigation.

3.1 Definition of Reference Conditions

The terms "background" and "reference" have both been used to describe the type of conditions in San Diego Bay to which shipyard data should be compared. Use of the term "background" refers to RWQCB Resolution 92-49, which, in its own words, "…authorizes Regional Water Boards to require complete cleanup of all waste discharged and restoration of affected water to background conditions (i.e., the water quality that existed before the discharge)." Resolution 92-49 does not specify that Regional Water Boards must require remediation to background conditions. In addition to the language quoted above, Section I.G of the resolution states that Regional Water Boards shall: "Ensure that dischargers are required to clean up and abate the effects of discharges in a manner that promotes attainment of either background water quality, or the best water quality which is reasonable if background levels of water quality cannot be restored, considering all demands being made and to be made on those waters and the total values involved, beneficial and detrimental, economic and social, tangible and intangible…." That Resolution 92-49 allows Regional Water Boards flexibility in setting cleanup levels has been affirmed in an opinion of the chief counsel of the SWRCB (Wilson 2002).

Resolution 92-49 does not provide an explicit definition of the term "background," but its implicit characterization of background conditions ("i.e., the water quality that existed before the discharge") is consistent with the use of the term "reference" by EPA and the U.S. Department of the Interior. These agencies' guidance documents for site-specific human and ecological risk assessments specify that a reference site should be similar to the contaminated site but for the source of contaminants at issue. Specifically, the relevant guidance documents state:

> "Background concentration" is defined . . . as the concentration of inorganics found in soils or sediments surrounding a waste site, but which are not influenced by site activities or releases. (U.S. EPA 1995c)
>
> A general guideline is to select reference locations that reflect the overall environmental conditions that can reasonably be expected in the site area given current uses other than those associated with the contamination under investigation. (U.S. EPA 1994)
>
> Baseline data should reflect conditions that would be expected at the assessment area had the discharge of oil or release of hazardous substances not occurred, taking into account both natural processes and those that are the result of human activities. (U.S. DOI 1996)
>
> [Reference is] A relatively uncontaminated site used for comparison to contaminated sites in environmental monitoring studies…. The reference area should be close to the site. It should have habitats, size, and terrain similar to the site under investigation…. The reference site need not be pristine. (U.S. EPA 1997a)
>
> The reference area should have the same physical, chemical, geological, and biological characteristics as the site being investigated, but has not been affected by activities on the site. (U.S. EPA 2002a).

The established standard for determining remediation of a particular site is therefore the general condition in areas close to, and similar to, the site in question. The language of Resolution 92-49 is consistent with this standard. This standard is also applied in the San Diego Regional Board's 2001 guidelines for assessment and remediation at the shipyards, which states that "The reference stations should be representative of current water quality conditions of San Diego Bay, including bay-wide urban anthropogenic sources of pollutants (at concentrations that are nontoxic) and excluding sources of pollutants associated with shipbuilding and repair activities" (RWQCB 2001).

Although neither the state nor the San Diego Regional Board has issued a definition of the term "background" that is as definitive as EPA's or the U.S. Department of the Interior's definitions of the word "reference," these two terms are clearly equivalent. The term "reference" is preferentially used in this document, because it is more consistent with national standards and generally accepted usage.

3.2 Identification of Appropriate Reference Conditions

Not all locations in San Diego Bay away from the shipyards necessarily represent appropriate reference conditions. Some such locations may be inappropriate because they are close to historical or ongoing contaminant sources that are different than those affecting the shipyards, and some such locations may be inappropriate because they have physical characteristics (depths, sediment type, water flow) that differ substantially from the nearshore, mid-bay conditions representative of the shipyard locations. Regional Board staff have issued several different specifications to the shipyards regarding reference data, or reference locations, to be used in this investigation. The following sections briefly describe each of these specifications, summarize the details of the latest specification, and comment upon identification of appropriate reference conditions.

3.2.1 Reference Condition Directives for the Shipyard Investigation

During the course of this investigation, Regional Board staff have issued the following five different specifications for characterizing reference conditions:

1. Background chemical concentrations listed in the Regional Board's June 1, 2001, guidelines for the sediment investigation (RWQCB 2001). These background chemical concentrations were based on data from NPDES Station REF-03, which is located at the Broadway pier, at or close to, the location of Bight '98 Station 2440. Derivation of the cited concentrations was described (Robertus 2002a, pers. comm.) as a weighted average of

13 years of monitoring data, with greater weight being given to more recent data.

2. Reference stations designated to be sampled during the sediment investigation (RWQCB 2001; Bay 2001a,b). These five reference stations were selected by screening Bight '98 stations to find stations throughout the bay that represent a range of sediment organic carbon and grain size and have low chemical concentrations and low amphipod toxicity. Twelve stations were identified by this process, of which five were specified as reference stations for this investigation.

3. Revised background chemical concentrations.(Robertus 2002a, pers. comm.). New background chemical concentrations were specified, to replace those in the June 2001 guidance document. These specifications were provided in March 2002, after completion of Phase 1 sampling. The new concentrations were based on a set of 12 Bight '98 stations (not the same as those identified by the previous screening step). The values were the 95 percent upper confidence limits (UCLs) on the mean concentrations.

4. Additional reference stations to be sampled in Phase 2 (Alo 2002a, pers. comm.). The shipyards were directed to sample the 12 Bight '98 stations that were identified in the previous specifications (Robertus 2002a, pers. comm.).

5. Final reference pool specifications (Barker 2003). Data from three different investigations—the Bight '98 study, the Navy's 2001 Chollas/Paleta Creeks investigation, and Phase 1 of the current investigation—were specified as a data set to define reference conditions.

3.2.2 Final Reference Pool Specifications

The "final reference pool" specification issued by Regional Board staff on June 9, 2003 (Barker 2003), lists 22 samples that are to be used as a basis for evaluating sediment chemistry, toxicity, and benthic macroinvertebrate conditions at the shipyards (Table 3-1). The June 9, 2003 specifications also include the Regional Board staff's recommendations regarding two aspects of the use of the specified reference area data:

- Use of 95 percent prediction limits on reference area conditions for evaluation of chemistry and toxicity data—upper prediction limits (95%UPL) for chemistry data and lower prediction limits (95%LPL) for toxicity data expressed as survival, fertilization, or normal development
- Use of the proposed California bays benthic response index (BRI) for evaluation of benthic macroinvertebrate data.

The final reference pool specification also acknowledges that alternative approaches can also be taken to data analysis.

3.2.3 Representativeness of the Final Reference Pool

The Regional Board staff's specification of the final reference pool was not accompanied by any technical analysis or detailed rationale explaining how the final reference pool was defined and why it is considered to be representative of appropriate site-specific reference conditions, as defined above. There are several aspects of the final reference pool that indicate that it is not truly representative of appropriate reference conditions for the shipyard investigation.

The Bight '98 stations included in the final reference pool are a subset of Bight '98 stations that were identified on the basis of gradients of chemical concentrations with distance from shore (Bay and Brown 2003). This analysis was carried out following a December 12, 2002, meeting in Regional Board offices at which an effort was made to define "Bay-wide ambient" conditions. This concept, and the distance-from-shore approach, does not encompass the most fundamental aspect of site-specific reference conditions, that is, an overall similarity to the site in location and physical characteristics but without site-related influences.

The final reference pool was clearly derived by picking and choosing individual data points, rather than identifying appropriate reference locations. The reference pool includes six stations that were sampled by multiple studies. Stations 2231, 2243, 2433, 2440, and 2441 were sampled by the Bight ’98 study, by the current study, and by the Navy’s 2001 study. Station 2238 was sampled by the Bight ’98 and Navy 2001 studies. For five of these stations, only selected data from one or two of the studies are included in the final reference pool. These cases are as follows:

- Station 2441—Only the shipyard Phase 1 data are included
- Station 2440—Only the Bight ’98 data are included
- Station 2231—Only the Bight ’98 data are included
- Station 2243—Only the Bight ’98 and shipyard Phase 1 data are included, but Phase 1 benthic macroinvertebrate data are excluded
- Station 2238—Benthic macroinvertebrate data from the Navy study are excluded.

In addition, all of these stations, and other Bight ’98 stations included in the reference pool, were also sampled by the shipyards during Phase 2 (at the direction of Regional Board staff). None of those data are included in the final reference pool.

Station 2441, which is located near the mouth of San Diego Bay, is very likely in a different hydrodynamic and biogeographic regime than the shipyards, because of its proximity to the open ocean. Station 2441 is therefore unlikely to be representative of conditions near the shipyards, which are located in the middle of San Diego Bay. Analyses of benthic macroinvertebrate communities show gradients throughout the length of the bay (discussed further in the body of this report), and Station 2441 is at one end of this spectrum and is notably distinct from stations in the mid-bay region. Data therefore show that Station 2441 is not only unlikely to be representative of conditions as they would be in the nearshore mid-bay area without the presence of the shipyards, but is in fact unrepresentative. (Results of the benthic macroinvertebrate analyses were available to Regional Board staff prior to the specification of the final reference pool.)

Bight ’98 Stations 2241, 2256, and 2257 are all included in the final reference pool, and all of these stations are located in the same area of San Diego Bay (south of the shipyards, on the other side of the channel). Bight ’98 Station 2258 is also located in this area of the bay, but is not included in the final reference pool.

The inconsistencies in the data selected for the final reference pool clearly indicate that those data were not selected by identifying appropriate reference locations on the basis of proximity to the shipyards, physical conditions, and absence of local sources. Because Regional Board staff have not provided any specific and detailed rationale for the selections, the method by which the final reference pool data were selected is unknown. However, by comparing the final reference pool samples with other data from the same locations, it is apparent that the final reference pool was selected by choosing data points with the lowest available chemistry concentrations, and the lowest available levels of biological responses. As a result, the final reference pool is biased toward the cleanest conditions available anywhere in San Diego Bay, and is not appropriate as a set of site-specific reference stations for the shipyard investigation.

3.2.4 Use of Reference Data for the Shipyard Investigation

Notwithstanding the inappropriateness of the final reference pool, these data have been used to evaluate shipyard conditions, following the direction of Regional Board staff. Because of the bias in the final reference pool, the results of evaluations using those data are biased toward overestimation of potential adverse effects at the shipyards.

The final reference pool is composed predominantly of Bight ’98 stations, and there are some technical issues related to use of those data. Several groups of chemicals that were included in the shipyard investigation were not included in the Bight ’98 study (and some were also not included in the Navy study). These chemicals include the butyltins, PCB Aroclors®, PCTs, and petroleum hydrocarbons. For these chemicals, reference conditions were characterized by only the Phase 1 data points that were included in the final reference pool. The Bight ’98 study had elevated detection limits for PCBs (only selected congeners were measured) and PAHs, and these chemicals were ordinarily undetected. The Bight ’98 study reported nondetected values at the method detection limit: the concentration that is statistically distinguishable from zero, but is not accurately quantifiable. The quantitation limit, the level at which concentrations are accurately quantifiable, is typically 5–10 times higher than the method detection limit. Consequently, Bight '98 even the detected values that are less than 5 to 10 times the detection limit have limited accuracy. For these reasons, reference conditions for PCB congeners and PAH were characterized by only the Phase 1 and Navy data points that were included in the final reference pool.

Regional Board staff requested that upper prediction limits for metals in the final reference pool be evaluated both by using the overall distribution of metals among final reference pool stations and by using a regression of metal concentrations on percent fines. Regression of each metal on percent fines in the final reference pool shows that no statistically significant regression relationship exists for five metals: cadmium, lead, mercury, selenium, and silver. Because percent fines in surface sediment is higher than the maximum concentration in the final reference pool at 28 percent of the shipyard stations, use of the regression approach would require extrapolation of the regression relationship beyond the range of data on which it is based. Because of its limited applicability, in terms of both individual metals and locations, the prediction limits for metals used here are based on the overall distribution of concentrations.

Of the three toxicity tests conducted as part of the shipyard investigation, only one, amphipod survival, was performed in all three studies. The echinoderm fertilization test was performed only in the shipyard and Navy investigations, and the bivalve development test was performed only in the shipyard investigation. For these toxicity tests, only data points from the appropriate investigations that were in the final reference pool were used to characterize reference conditions.

Because of the temporal variability of benthic macroinvertebrate communities, it is technically inappropriate to evaluate the shipyards with respect to benthic data that were collected as part of a different investigation 3 years earlier. Differences between methods and taxonomists, likely non-correspondence between taxa identified only to genus level, and even changes in species names, all make detailed comparisons between studies inappropriate. Use of benthic macroinvertebrate data from the entire final reference pool is therefore likely to result in the identification of differences that are the result of changes over time or variations in methods, rather than of actual effects of the shipyards themselves. Evaluation of synoptically collected data is critical to appropriate interpretation of benthic macroinvertebrate community conditions.

The final reference pool specification also dictates that benthic macroinvertebrate communities be evaluated using the BRI, a newly developed and unvalidated method that has several clear technical shortcomings (a critique of the BRI is included in Appendix M). For these reasons, although BRI values have been computed for the final reference pool and the shipyard stations, the definitive analysis of benthic macroinvertebrate communities presented here relies on the reference area data collected synoptically during Phase 1.

Analyses that are carried out using reference data from different investigations with different methods or data quality standards will lead to increased uncertainty about the results of comparisons to site data. Thus, use of the final reference pool not only leads to assessments that are biased in the direction of overprotectiveness, but will also reduce the level of confidence in those assessments relative to those that would be made using appropriate site-specific reference stations.

4 Sediment Chemistry Data

Surface and subsurface sediment samples were collected from a total of 66 locations in or adjacent to the shipyard leaseholds in Phase 1 and Phase 2. The five designated reference stations were sampled in both Phase 1 and Phase 2, and seven additional stations in San Diego, all previously sampled during the Bight '98 survey, were also sampled in Phase 2. Sediment samples were analyzed for metals (arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and zinc), organometallic compounds (butyltin, dibutyltin, TBT, and tetrabutyltin), PCB and PCT Aroclors®, PCB congeners, PAHs, and petroleum hydrocarbons. Sediments were also analyzed for grain-size distribution, organic carbon, and total solids (i.e., conventional analytes). During Phase 1, surface sediment samples were also analyzed for acid-volatile sulfide (AVS) and simultaneously extracted metals (SEM). During Phase 2, selected samples were analyzed for pesticides and for mineral constituents using an electron microprobe. Specific details of sampling locations and analytes are described in the section titled *Study Design*. All sediment chemistry data are included in Appendix B. Laboratory analytical data were reviewed to assess compliance with the data quality objectives and overall good laboratory practice. Quality assurance reports for all chemistry data are included in Appendix F. All sediment chemistry data were found to be acceptable.

4.1 Sediment Chemistry Characteristics at the Shipyards

Although the principal purposes of the sediment analyses were to support the interpretation of biological data and to allow comparison with background or site-specific reference stations, there are also a number of aspects of the chemistry measurement themselves that provide information about sediment characteristics at the shipyards. These topics are discussed in the following sections. The spatial distribution of chemicals and comparisons with final reference pool values are also discussed.

4.1.1 Grain Size Distribution

Clay and silt were the predominant particle sizes found throughout the shipyard leaseholds, occurring in roughly equal proportions and together making up 75–90 percent of the mass of surface sediment particles. Generally, the greatest fraction of fine particles (clay and silt) was found near shore. However, the lowest fraction of fines in the shipyard leaseholds, 31 percent, was found near shore at Station SW01. The largest fraction, 100 percent, was found at Station SW06. Between the boundary of the shipyard leasehold and the ship channel, the fraction of fines in surface sediment generally decreased to below 50 percent.

The fraction of fine sediment at the five original reference areas sampled during this investigation ranged from approximately 30 to 50 percent. At all of these stations except 2231 (in the center of the bay northwest of the shipyards), fine and very fine sand was the dominant particle size. At Station 2231, fine particles made up approximately 50 percent of the sediment, and fine and very fine sand made up approximately 25 percent.

Distinct vertical profiles of a decreasing fraction of percent fines were found in many cores. In other cores, the fraction of percent fines was constant or variable down to depths of several feet. Figures showing these vertical profiles are included in Appendix B5, and a summary of the presence or absence of a profile of percent fines in near-surface sediment is shown in Table 4-1. Distinct vertical profiles were found in all cores from outside the shipyard leaseholds except for NA21. Some cores from areas inside the leaseholds also showed distinct vertical profiles of percent fines in near-surface sediment, and others had variable percent fines.

The sort of graded bedding represented by vertical profiles of grain size, with a higher fraction of percent fines at the surface, can be produced by gradual deposition of a mixture of materials, typically in a declining flow condition, such as occurs with distance from a river or other point source, or following a large flow such as is produced by runoff following a storm. Ungraded or mixed bedding, as represented by the well-mixed profiles of sediment particles, can be produced by a more uniform source of materials, by repeated episodes of disturbance and rapid resettling, or by ongoing mixing of the sediment. Examples of these two types of profiles are shown in Figure 4-1. The ungraded bedding that is observed at the shipyards may therefore indicate the presence of recent or ongoing physical disturbance. The existence of ungraded bedding at Stations NA20 and NA21, the area at which engine tests are conducted, is consistent with frequent physical disturbance of the sediment in those areas. The existence of ungraded bedding at Stations SW20 and SW24 is consistent with sediment disturbance by high-volume storm water flow from Outfall SW4. In contrast, graded bedding is found at Station SW02, where the presence of eelgrass indicates that the sediment has not been recently disturbed. Therefore, the presence of ungraded or graded bedding is likely to provide an indication of the presence or absence of physical disturbance at locations within the shipyards. The spatial distribution of stations with apparent physical disturbance is shown in Figure 4-2.

4.1.2 Chemicals with Low Detection Frequencies

Most individual low-molecular-weight PAH (LPAH) compounds were undetected at most locations, and few high-molecular-weight PAH (HPAH) compounds were undetected at any location. Quantitation limits for LPAH compounds were generally 15 µg/kg or less. When sums were calculated and half the quantitation limit was used for those chemicals that were not detected, LPAH made up only 20 percent of total PAH. Because of the high fraction of nondetected values among the LPAH compounds, values of total HPAH have less uncertainty—and are likely to be more accurate—than values of total LPAH. HPAH is therefore a better basis for describing PAH distributions at the sites, for comparing the shipyards to reference conditions, and for evaluating biological effects with respect to PAH.

Of the petroleum hydrocarbons measured, gasoline-range organics (GRO) (6 through 10 carbons) were detected in only 2 of 163 shipyard samples, or approximately 1 percent. Residual-range organics (RRO) (25 through 36 carbons) also had a relatively low detection frequency: 111 of 167 samples, or approximately 66 percent. Diesel-range organics (DRO) (10 through 25 carbons) were detected most frequently, with a frequency of 73 percent. Because of the high frequency of nondetected values, GRO has been excluded from further analyses, and DRO and RRO have been treated separately rather than being summed to produce an estimate of total petroleum hydrocarbons.

Among the metals analyzed, selenium had a low detection frequency: 31 of 185 samples, or approximately 17 percent. Quantitation limits for selenium were generally 1.1 mg/kg or less. Detected values of selenium were at or only slightly above the quantitation limit. Results for selenium should be interpreted with caution because concentrations near the quantitation limit are expected to be more variable than elevated concentrations would be.

Tetrabutyltin had the lowest detection frequency of the butyltins: 38 of 185 samples, or approximately 21 percent. Quantitation limits for tetrabutyltin were generally 3 µg/kg or less. Detection frequencies of the other butyltins were approximately 80 percent.

4.1.3 Measures of Total PCBs

The analyses conducted provide three measurements of total PCBs in sediments: the sum of (a subset of) congeners, the sum of homologs, and the sum of Aroclors®. Because not all congeners were measured, the sum of congeners is expected to be lower than either of the other measures, which should better represent total PCBs. As expected, the sum of congeners was found to be lower than the other measures. The sum of homologs and the sum of Aroclors® were strongly correlated with one another. The sum of Aroclors® was generally greater than the sum of homologs, especially at higher absolute concentrations of PCBs. Because identification and quantification of Aroclors® is based on analyses of pure Aroclor® standards, and the PCBs present in the sediment are likely to have undergone some chemical weathering and thus differ somewhat from their original chemical forms, quantitation of Aroclor® mixtures is likely to be less accurate than quantitation of homologs. Laboratory analysts responsible for quantification of Aroclors® reported that the Aroclor® values in sediment are likely to be biased high. The relationships between these two measures of total PCBs is discussed more fully in the quality assurance report for chemistry data (Appendix F). The sum of PCB homologs is therefore considered to be a more accurate representation of total PCBs in the sediment. However, the sum of Aroclors® was used in the human health and ecological risk assessments because existing studies and standard measures of the toxicity of PCBs are based on measurements of Aroclors®.

4.2 Distributions of Sediment Chemicals

Distinct spatial gradients of sediment chemical concentrations were found across the shipyard sites. Contour maps showing the distribution of chemicals in surface sediment are included in Appendix B. The spatial distributions of all metals and butyltins show a similar strong pattern (Figures 4-3 through 4-16), with the highest concentrations at the northern boundary of the Southwest Marine leasehold. Some metal concentrations have lower concentrations in the center of the NASSCO leasehold, near the foot of the floating dry dock or the ends of the building ways. Concentrations generally decrease with distance from shore. Mercury shows an anomalous elevation at Station SW19, which is located near the ship channel outside the Southwest Marine leasehold, and well away from any comparable concentrations of mercury within the leasehold.

The spatial distribution of PAH is distinctly different from that of metals (Figure 4-17). Although higher than average PAH concentrations are also found at Southwest Marine's northern leasehold boundary, the maximum concentrations of PAH are found between Southwest Marine Piers 3 and 4, near the storm drain outfall SW4.

PCBs and PCTs are distributed in a pattern similar to that of metals (Figures 4-18 and 4-19). The highest concentrations are at Southwest Marine's northern leasehold boundary. Relatively elevated concentrations of PCBs are also found between Southwest Marine Piers 3 and 4, and at the foot of Sicard Street, near the common boundary of the two shipyard leaseholds. Local elevations in PCT concentration are also found near the foot of the NASSCO dry dock and the NASSCO graving dock.

DRO has a distribution similar to that of metals and PCBs (Figure 4-20). The distribution of RRO (Figure 4-21) also has a local maximum at Southwest Marine's northern boundary, but also has relatively elevated values underneath the NASSCO dry dock and at the stations off the mouth of Chollas Creek. DRO and RRO are also elevated at the southern end of the NASSCO leasehold, at Stations NA22 (the mouth of Chollas Creek) and NA21 (further out, beyond the pier line). A tugboat collision that occurred near Station NA21 in 1993 released approximately 300 gallons of diesel fuel, and another 400-gallon diesel spill from a Sealift ship occurred in this area on August 18, 2001 (four days after Phase 1 sampling) (Chee 2003, pers. comm.). Phase 2 sampling found higher concentrations of DRO and RRO in the top two core sections at NA21 than were found at the surface in 2001. The higher concentrations found in 2002 may represent the residue of this 2001 spill.

Elevated concentrations of chemicals are often found in finer sediment, because the greater surface area per mass of small particles provides relatively more area for sorption of dissolved chemicals. However, the distribution of fine sediment (clay and silt) at the shipyards does not show a peak at the northern boundary of Southwest Marine, as do most chemicals (see maps in Appendix B). The highest concentrations of fine sediment are found in the center of the Southwest Marine leasehold, under the dry dock, and in a band near shore in the NASSCO leasehold. The fraction of fine particles in the sediment decreases with distance from shore, as is typical of nearshore environments. The absence of a clear association between the distributions of fine sediment and of most measured chemicals indicates that sorption to sediment particles is not the primary determinant of chemical distributions.

Cores collected during Phase 2 showed distinct vertical profiles of chemical concentrations at most shipyard locations. Concentrations were generally highest in the surface sediment and decreased with depth. In general, concentration gradients with depth were steeper at Southwest Marine than at NASSCO. Several stations were notable for having chemical concentrations that increased with depth for at least the first 4 to 6 ft, specifically Stations NA01, NA04, NA09, and NA16. In conjunction with other characteristics of the sediment (discussed in following sections), these inverted profiles may represent the result of sediment disturbance and resettling, as discussed in the following summary section.

4.3 Comparison to Final Reference Pool Conditions

Concentrations of most chemicals measured in surface sediment in and adjacent to the shipyard leaseholds were generally higher than the 95%UPL for the final reference pool (Table 4-2). The 95%UPL values were calculated for a single future prediction, without any adjustment for multiple comparisons. At every station sampled (66 total), at least one chemical exceeded its final reference pool level. GRO was not detected at any station in the final reference pool or at the shipyards. DRO and RRO were not detected at any of the final reference pool stations, but were detected at most shipyard stations. Because detection limits do not accurately represent the range of variation of concentrations, 95%UPL values have not been calculated for these petroleum hydrocarbons.

Sediment grain size in final reference pool samples was coarser than sediment at the shipyards. Whereas the fraction of fines at the shipyards ranged from approximately 75 to 90 percent, the range in the final reference pool samples was 13 to 77 percent, and 16 of the 22 samples had a fine fraction less than 60 percent. These differences were statistically significant ($p<0.05$, two-sample t-test with unequal variances). TOC in final reference pool samples was less than in shipyard samples. TOC in final reference pool sediment ranged from 0.25 to 1.6 percent, whereas TOC in shipyard sediment ranged from 0.51 to 7.4 percent. These differences were also statistically significant ($p<0.05$, two-sample t-test with unequal variances). Thus, the sediment characteristics of the final reference pool are fundamentally different from the sediments at the shipyards. Such differences may affect several aspects of data interpretation, because chemical concentrations are generally lower in coarser sediment, and biological communities also vary with sediment grain size and TOC content.

4.4 Sulfide Limitations on Bioavailability of Metals

Marine sediment is a complex chemical environment that contains many inorganic and organic constituents. Variations in reduction-oxidation potential in the sediment can cause some chemicals—specifically including free metals—to be present in many different oxidation states. Different oxidation states of a chemical have different characteristics that affect their ability to chemically bind to other sediment constituents. Binding of chemicals such as metals can reduce their bioavailability, and thus their toxicity.

Sulfide is one of the sediment constituents that can potentially form chemical complexes with divalent metals, and thereby reduce their bioavailability. Concentrations of sulfide and SEM were measured at the shipyards to assess whether this mechanism could be limiting metal bioavailability. Sulfide is typically present in sediment in a variety of forms, including dissolved sulfide (which is present in three forms: hydrogen sulfide, bisulfide, and sulfide) and particulate metal sulfide (which is primarily present as iron sulfide, and to a lesser degree, as sulfides of a variety of other metals). Iron sulfide is generally separated into two operational categories, AVS and pyrite (Cornwell and Morse 1987). AVS includes several chemical forms of sulfide: "amorphous" iron sulfide, mackinawite, greigite, and pyrrhotite. Pyrite resists dissolution by acids and is typically analyzed using methods that include an oxidative step.

Surface sediment for the shipyards study was analyzed for AVS and SEM. The molar ratio of SEM to AVS is considered to be an index of bioavailability; if it is less than 1, metals are believed to be unavailable and nontoxic to benthic organisms (e.g., Allen et al. 1993; Di Toro et al. 1992). Comparison of the residual SEM (i.e., the molar concentration of metals that is present in excess of the molar concentration of AVS) to sediment toxicity additionally allows evaluation of possible toxic effects related to SEM.

Results for AVS and SEM on a molar basis, residual SEM, and toxic effects are provided in Table 4-3. The highest concentration of residual SEM was 113 mmol/kg at Station SW04. Apart from this station, the concentrations of residual SEM ranged from −21 to 29 mmol/kg. The median concentration was 4.5 mmol/kg. AVS concentrations ranged from undetected (detection limit of 0.02 mmol/kg, or 0.7 mg/kg) to 49 mmol/kg (1,570 mg/kg). Molar SEM concentrations exceeded AVS concentrations at all except nine stations, which also tended to have some of the higher AVS concentrations found at the site. No pattern of residual SEM and biological effects was evident for any of the toxicity tests (described in Section 6). For example, no toxic effects were found at Station SW04, which had by far the highest concentration of residual SEM. Amphipod and bivalve effects were present at Station SW27, which had a relatively low concentration of SEM (2.1 mmol/kg). Bivalve effects were noted at three of the five stations with SEM concentrations below the AVS concentrations.

AVS is important in controlling the bioavailability of metals in sediment. However, additional factors such as organic carbon and iron and manganese oxides can also affect the solubility and availability of divalent metals (Ankley et al. 1996). The results of the residual SEM and toxicity tests indicated that biological effects were not related to the residual SEM concentrations, and that therefore, SEM is not a primary factor in controlling metal bioavailability at the shipyards.

4.5 Associations of Metals with Sediment Minerals

Analyses of the Phase 1 triad data—discussed in more detail in following sections of this document—indicate that there are no strong associations between biological effects and any of the measured chemicals. As described in the previous section, AVS concentrations do not fully account for limited metal bioavailability. Bioavailability can also be limited if potentially toxic chemicals—particularly metals—are chemically bound into other parts of the sediment matrix. Consequently, Phase 2 analyses included electron microprobe examination of the sediment particles from several stations. Microprobe analysis allows assessment of the distributions of metals among different components of the sediment. Microprobe analysis uses an electron microscope to identify the physical location and chemical form of metals—that is, whether the metals are within or on the surface of particles, and whether they are in a metallic form or are chemically bound as part of mineral constituents. The results can therefore be used to evaluate the impact of the matrix on metal bioavailability.

Because the goal of the microprobe analyses was to determine whether it is possible that metals at the shipyards may have limited bioavailability due to matrix effects, analyses were performed on a few samples with high metal concentrations rather than on all Phase 2 sediment samples. Four stations with high concentrations of copper, chromium, or both, and with and without toxicity, were selected for this analysis (Table 4-4). (Toxicity determinations are described in the section titled *Toxicity Test Results.*) Copper and chromium were selected for these analyses on the basis of their potential association with shipyard activities. Sediment for microprobe analyses was taken from the 0–2 ft horizon of the Phase 2 cores at the target stations.

The electron microprobe uses a narrowly focused electron beam that can be used to determine the chemical composition of individual mineral types within sediment particles. The amount of characteristic X-rays emitted by each element when excited by the electron beam allows a quantitative determination of the amount of each element present. Metal content in each mineral form is quantified relative to other mineral forms in the same sample, rather than in terms of overall sample mass. The results therefore can be used to identify which mineral form(s) a metal is predominantly associated with, but do not provide an estimate of overall metal concentration in the sample.

Microprobe analyses were conducted by Dr. John Drexler at the University of Colorado, Boulder. Dr. Drexler's report, and a table summarizing his findings, are included in Appendix B. Microprobe analyses produced similar analytical results for samples from Stations NA19, SW04, and SW27. Results for the sample from Station SW02 were different from the other stations, and were judged by the laboratory analyst to be less reliable. In the samples from Stations NA19, SW04, and SW27, copper was predominantly found associated with the mineral chalcopyrite, or copper iron sulfide. Chalcopyrite is a common mineral and the principal ore of copper. Chromium in these samples was predominantly associated with the mineral chromite, or iron chromium oxide. Chromite is the principal ore of chromium. Copper and chromium are each an intrinsic part of the crystal structure of the minerals in which they are found, and thus are not subject to sorption–desorption interactions with the surrounding water. A major fraction of the total mass of copper and chromium in these samples is therefore not bioavailable.

The microprobe analysis also identified slag as the most frequently occurring form of mineral in these samples. Slag is an amorphous residue produced by the heating and extraction of metals from ore in a smelter. Chalcopyrite was found in the slag and as separate mineral crystals. Chromite was found exclusively in the slag. Because copper and chromium are present primarily as mineral constituents, they are expected to have a low bioavailability. The relative abundance of slag, and the presence of minerals representative of two important metal ores, indicates that smelter waste is present in the sediment at the shipyards and plays a major role in determining the distribution and bioavailability of metals. Because copper and chromium were the only two metals targeted by the microprobe analyses, these analyses provide no information about the mineral forms of other sediment metals. However, because other metals have distributions similar to those of copper and chromium they may also be associated with the smelter waste, and consequently also have low bioavailability.

4.6 Pyrogenic vs. Petrogenic PAH

PAHs that originate from petroleum sources (petrogenic PAH) and combustion sources (pyrogenic PAH) have different chemical characteristics. These characteristics can be evaluated in environmental samples to determine whether the PAHs are petrogenic or pyrogenic in origin. Petrogenic PAHs are present in crude oil and coal and in petroleum products derived from these fossil fuels. Pyrogenic PAHs are formed during combustion of any carbon-containing fuel, including coal, petroleum products such as gasoline or diesel oil, and natural products such as wood.

A distinguishing characteristic between pyrogenic and petrogenic PAHs is the relative abundance of alkylated PAHs with respect to their parent compounds. (Alkyl groups consist of one or more linked carbon atoms and associated hydrogen atoms, with a general chemical formula of C_n+H_{2n+1}. Several different alkylated forms may be found for each unalkylated parent PAH compound.) Petrogenic materials have a higher relative abundance of alkylated PAHs than their parent compounds. When the relative abundance of petrogenic PAHs is plotted against the alkylation level (i.e., the number of alkyl carbons attached to the parent PAH), a bell-shaped curve is typically found, with PAHs at the two- and three-carbon alkylation levels present in greater abundance than the unalkylated parent compound (Sauer and Uhler 1994; Stout et al. 2002). For pyrogenic sources, the parent PAH tends to be present at the highest concentration and the alkylated PAHs are present at lower concentrations and show a decline in abundance with increasing level of alkylation (Sauer and Uhler 1994; Stout et al. 2002). This distribution may be modified by weathering, particularly of the LPAHs. Weathering processes include physical, chemical, and biological removal of PAHs by mechanisms such as sediment–pore water partitioning and biodegradation (McCarthy et al. 1998). Weathering tends to remove unalkylated parent PAHs more quickly than the alkylated homologs (McDonald et al. 1997), which in turn tends to shift the distribution of PAHs towards a petrogenic pattern (Stout et al. 2002). Weathering also removes LPAHs more quickly than HPAHs. The lower the molecular weight of the PAH, the more quickly it is weathered (Stout et al. 2002).

Alkylated PAH analyses were completed for samples from the 0–2 ft intervals of sediment cores collected at Stations NA19, SW08, and SW24, and for all of the surface sediment samples collected for the pore water study. These samples were analyzed for EPA's 16 priority pollutant PAHs and for the following additional PAHs:

- C1- through C4-alkylated naphthalenes, phenanthrenes + anthracenes, and chrysenes + benz[a]anthracenes
- C1- through C3-alkylated fluorenes
- C1-fluoranthenes + pyrenes
- Dibenzothiophene and its C1-, C2-, and C3-alkylated homologs.

A complete analyte list is provided in Table 4-5.

Analyses for parent and alkylated PAHs were performed by gas chromatography and mass spectrometry (EPA Method 8270C) with selected ion monitoring. The method was modified to include analysis of the alkylated PAHs. Each parent compound was identified individually, but for parent PAHs with the same molecular weight, the results for all PAH isomers at each alkylation level were summed. This procedure was used to ensure that all of the isomers present at each alkylation level were included in the final result. Parent PAHs with the same molecular weight include phenanthrene and anthracene, fluoranthene and pyrene, and chrysene and benz[a]anthracene. For each of these pairs of PAHs, the results for each alkylation level are combined. For example, all isomers of dimethylphenanthrene, dimethylanthracene, ethylphenanthrene, and ethylanthracene are included in the value for C2-phenanthrene + anthracene. All analytical results for PAHs are presented in Appendix B.

PAHs at the shipyards and reference stations appear to be predominantly pyrogenic in origin, with an additional petrogenic source evident at Station SW02 and possible low levels of petrogenic PAHs at other stations. This conclusion is supported by the relative distribution of parent and alkylated HPAHs in the samples and by the low proportion of LPAHs with respect to HPAHs. For the HPAHs (i.e., fluoranthene + pyrene and chrysene + benz[a]anthracene), the parent PAHs consistently show a higher concentration than the alkylated homologs, with decreasing concentrations with increasing levels of alkylation (Figure 4-22). This pattern is typical of pyrogenic PAHs.

At stations with detected levels of naphthalene and alkylated naphthalenes (i.e., Stations SW01, SW02, SW04, and SW08), the relative abundance of the naphthalene homologs increases with alkylation level (Figure 4-22). A similar pattern was found in 1994 in the general vicinity of reference Station 2440 (Zeng and Vista 1997). Zeng and Vista (1997) concluded that this pattern may indicate the presence of petrogenic PAHs. However, this pattern is consistent with weathered PAHs from various sources. Among PAHs, naphthalene is particularly water-soluble and biodegradable, and for this reason, the use of naphthalene and its alkylated homologs is not recommended for source identification (Stout et al. 2002).

The distribution of the LPAHs fluorene and dibenzothiophene and their alkylated homologs was not consistent with either a pyrogenic or petrogenic source. These distributions are likely to be the result of weathering and may reflect input of low levels of PAHs from petrogenic sources. However, the relative abundance of phenanthrene + anthracene was greater than their C1-homologs at all stations (Figure 4-22). This pattern is consistent with a pyrogenic source of PAHs (Zeng and Vista 1997). The HPAHs were much more abundant than the LPAHs, which accounted for no more than 20 percent of total PAHs at all stations except SW02 (Table 4-6). With the exception of Station SW02, these results are consistent with results of Zeng and Vista (1997) and McCain et al. (1992) for sediment samples collected in San Diego Bay in 1994 and 1984–1988, respectively, and indicate a pyrogenic or predominantly pyrogenic source of PAHs (Zeng and Vista 1997).

At Station SW02, LPAHs accounted for 44 percent of total PAHs. The alkylated HPAH profiles at Station SW02 clearly indicate a pyrogenic source of PAHs (Figure 4-22). However, an additional source apparently has contributed LPAHs to the sediment at this station. The alkylated PAH profiles are consistent with weathered PAHs in general and do not provide a conclusive indication of the source of these LPAHs.

The predominantly pyrogenic origin of PAH at the shipyards indicates that combustion sources, such as vehicle and power plant emissions, are the primary source of PAH in shipyard sediments. Consistency of these results with other investigations conducted elsewhere in San Diego Bay (Zeng and Vista 1997) indicates that PAH sources to shipyard sediment are similar to PAH sources to sediment throughout the bay.

4.7 Summary

Distinct and consistent spatial patterns were found for most sediment chemicals, with the highest concentrations found at the northern boundary of the Southwest Marine leasehold, and higher concentrations generally found near shore. A notably different pattern was found for PAH, suggesting that it may have originated from a different source.

Sediment chemical concentrations at the shipyards were commonly higher than concentrations in the final reference pool samples. Sediment at the shipyards was much finer than sediment in the final reference pool samples, with almost no overlap in the fraction of percent fines in the two groups. Because higher concentrations of chemicals are ordinarily found on finer particle sizes, the difference in sediment characteristics is likely to account for at least part of the differences in concentrations. Sediment at the shipyards also had higher TOC concentrations than final reference pool samples, which may affect both chemical distributions and habitat suitability for benthic fauna.

Although AVS is not present in sufficient quantity to bind sediment metals and limit their bioavailability, bioavailability is likely to be limited by sequestration of metals in smelter slag. Microprobe analyses of sediment particles show that copper and chromium are both present predominantly in the form of common ore minerals, and those minerals are associated with smelter slag. Although copper and chromium were the only metals examined with microprobe analyses, other metals are distributed similarly and may therefore also be associated with smelter slag and also have limited bioavailability.

The presence or absence of graded bedding in the upper layers of sediment, as indicated by the distribution of fine particles, is a potential indicator of the presence of physical disturbance at several locations throughout the shipyards.

Although petroleum hydrocarbons were detected in most locations, the ratios of alkylated and parent PAH compounds indicate that the PAHs are of pyrogenic, rather than petrogenic, origin.

This result, and the different spatial distributions of PAH and petroleum hydrocarbons, indicates that the PAH and the petroleum hydrocarbons originate from different sources. The presence of maximum PAH concentrations near a city storm drain outfall indicates that scavenging of vehicle and other combustion emissions from city streets is likely to be the primary source of PAHs to the shipyard leasehold.

Distinct vertical chemical concentration gradients were also found in most locations, with higher concentrations at the surface. The stations that have inverted concentration gradients (NA01, NA04, NA09, and NA16) also have ungraded bedding, as indicated by the distribution of fines. Because metals, at least, appear to be associated with slag particles, and because smelter slag has a greater density than clay and silt minerals, repeated disturbance and resuspension of the sediment could have resulted in downward migration of the heavier slag particles through more rapid settling.

5 Pore Water Chemistry Data

Pore water samples for metal, PCB, and butyltin analyses were collected in August and September 2002, as described in the Phase 2 FSP (Exponent 2002). Following completion of this sampling, on October 18 Regional Board staff issued the shipyards a 13267 letter specifying that additional pore water and associated sediment samples be collected for analysis of PAH (Robertus 2002c, pers. comm.). Additional sampling of sediment and pore water for PAH was conducted in November 2002. All pore water data are presented in Appendix D, and sediment data for the corresponding pore water samples are included with other sediment data in Appendix B.

5.1 Comparison of Site and Reference Data

Because the final reference pool was defined after pore water collection and analysis was completed, and because pore water was not sampled during the Bight ’98 or 2001 Navy investigations, pore water concentrations at the shipyard cannot be compared to the entire final reference pool. Only the three reference stations that were sampled for pore water in this investigation and that are included in the final reference pool (2243, 2433, and 2441) were therefore used for this comparison.

Statistical comparisons could not be carried out for chemicals with a high proportion of nondetected results. These chemicals were cadmium, hexavalent chromium, selenium, and all of the butyltins. Most of these chemicals were undetected at almost all stations. Dibutyltin and TBT were undetected at all three of the stations in the final reference pool, but were detected at several shipyard stations. For the other chemicals, statistical comparisons between site and reference were carried out using a one-sided nonparametric Wilcoxon test and a confidence level of 95 percent. Samples from the shipyard sites were pooled for this test, to provide a measure of replication. The results therefore characterize the shipyards as a whole, rather than individual stations. Statistical tests were carried out pooling all stations from both shipyards and pooling the data from each shipyard separately; the results of these tests, in terms of the significant differences identified, are the same in all cases.

Statistical test results show that concentrations of arsenic, nickel, and silver in pore water from the shipyard sites are not significantly different from concentrations at the stations in the final reference pool. Concentrations of copper, lead, mercury, zinc, PAH, and total PCBs (sum of homologs) are all higher at the shipyard sites.

5.2 Comparison to California Water Quality Criteria

Measured pore water concentrations for most chemicals at most shipyard locations are below water quality criteria established in the California Toxics Rule (CTR; U.S. EPA 2000a). Comparisons were performed to the criterion continuous concentration, which is the highest concentration of a pollutant to which aquatic life can be exposed for an extended period of time without deleterious effects (Table 5-1). CTR values are for dissolved, rather than total, fractions. Because pore water was not filtered (in accordance with EPA guidance [U.S. EPA 2001a]), comparison to criteria for dissolved constituents is protective. With the exception of Station SW02, which is an outlier as a result of suspended solids in the sample (discussed further below), measured concentrations of arsenic, cadmium, nickel, selenium, silver, and zinc are all below CTR values at all shipyard locations. Hexavalent chromium was undetected at a quantitation limit equal to the CTR criterion, so true concentrations were below the criterion. In contrast, measured copper concentrations are above CTR values at all shipyard stations and at all reference stations. Concentrations of total PCBs (the sum of homologs) are also above CTR values at all shipyard and reference stations except 2243 and 2441. Lead exceeds CTR values at 7 stations: NA06, NA16, SW02, SW04, SW08, SW24, and SW25.

Although the comparison to CTR criteria identifies several metals for which measured pore water concentrations are entirely below levels of concern, the measured pore water concentrations are biased high for most chemicals, as described in the section below titled *Relationship between Pore Water and Sediment Chemistry*. Exceedance of a CTR value by a measured pore water concentration is therefore not a definitive indicator of a potential effect on aquatic life.

5.3 Relationship between Pore Water and Sediment Chemistry

Use of pore water data for development of sediment cleanup levels is contingent on the existence of a good relationship, for each chemical, between the concentration in pore water and the concentration in sediment. Such relationships can be used to relate sediment concentrations to surface water quality criteria, making the assumption that surface water quality criteria are relevant to sediment-dwelling organisms. Consequently, linear regression analyses were conducted to determine whether there are any statistically significant relationships between chemical concentrations in pore water and sediment. Data were transformed as necessary to meet the requirements for a valid regression analysis.

Statistically significant (p<0.05) relationships between pore water and sediment were found for copper, lead, mercury, zinc, TBT, and PCBs (Table 5-2). No relationship between pore water and sediment was found for arsenic, chromium, nickel, and silver. Cadmium and selenium were not evaluated because they were detected in only one or two pore water samples. For copper, lead, mercury, zinc, and PCB, the regression had a positive intercept that was statistically significantly different from zero. A positive intercept means that concentrations of those chemicals would be found in pore water even in the absence of sediment. Such positive intercepts are not consistent with the equilibrium partitioning model, which requires that pore water chemical concentrations reach zero when sediment chemical concentrations reach zero. The R-squared values for these regressions (Table 5-2) quantify the amount of variation in pore water concentrations that is associated with variation in sediment concentrations; these values range from approximately 60 percent to 85 percent. Sources of variation other than sediment, although they cannot be identified, may be responsible for the positive intercepts. The consistency of the observation of a positive intercept indicates that influences other than sediment on pore water concentrations are systematic and impart a positive bias. One possible source of such a bias is very fine suspended or colloidal material in the samples that carries sorbed chemicals and that could not be removed by centrifugation. Even if the amount of this material is constant from sample to sample, the chemical concentrations associated with that material are likely to be higher in samples from sediment with higher chemical concentrations. Thus, the amount of bias in measured pore water concentrations may increase with the concentration of sediment chemicals. The true slope of the pore water : sediment relationship therefore may be less than the observed slope.

For mercury, zinc, TBT, and PCB, variance of the data increases with concentration—a frequently observed circumstance for environmental data. Consequently, concentrations of these chemicals in both pore water and sediment were transformed prior to regression analysis. The resulting regression equations are therefore nonlinear (Table 5-2). Non-linear relationships are also not consistent with the equilibrium partitioning model of sediment : pore water relationships.

Pore water concentrations of most chemicals at Station SW02 are markedly higher than at other shipyard stations. Scatter plots of pore water concentrations against sediment concentrations show that, for those chemicals for which there is a relationship, pore water concentrations at Station SW02 are not consistent with this relationship. Figure 4-23 is a plot of the sediment and pore water data for copper, showing the unusual value for SW02. The relationships for other chemicals, except for arsenic and TBT, show a similar nonconformity for Station SW02. Field personnel noted a cloudy appearance to the pore water sample collected at Station SW02, which indicated the presence of some suspended material remaining after centrifugation. Because SW02 is located in an eelgrass bed, the sample may contain dissolved or particulate organic materials that have a density too low to be removed by centrifugation, but that serve as sorption sites for chemicals. The presence of such materials in the sample would lead to relatively elevated measured "pore water" concentrations in that sample. For this reason, concentrations of most chemicals measured in the pore water sample from Station SW02 are not considered to be accurately representative of *in situ* pore water conditions at that location. Data from SW02 were consequently omitted from the regressions of pore water on sediment for all chemicals except arsenic and TBT, for which SW02 did not appear to be an outlier.

The observed relationships between chemical concentrations in pore water and sediment is further discussed in the section titled *Development of Candidate Cleanup Levels,* with respect to use of these relationships for developing candidate cleanup levels.

6 Toxicity Test Results

Sediment toxicity tests were conducted as part of the sediment triad analyses, including measurements at 30 stations at the shipyard sites and 5 reference stations. The following three toxicity tests were conducted on each sample:

- A 10-day amphipod survival test using *Eohaustorius estuarius* exposed to whole sediment. The endpoint for this test is the percentage of amphipods alive.
- A 48-hour bivalve larva development test using the mussel *Mytilus galloprovincialis* exposed to whole sediment at the sediment–water interface. The endpoint for this test is the percentage of live normally developed larvae.
- A 40-minute echinoderm egg fertilization test using the purple sea urchin *Strongylocentrotus purpuratus* exposed to sediment pore water. The endpoint for this test is the percentage of eggs fertilized.

Additional amphipod tests were conducted on samples from Stations NA07 and SW04. These locations were selected on the basis of the results of previous investigations. Both of these locations were previously observed to have elevated chemical concentrations and thus were expected to have elevated toxicity. These tests used serial dilutions of the original sediment to confirm the responsiveness of the test organisms to a gradient of chemical concentrations.

Amphipod tests were conducted following methods specified by ASTM (1999); bivalve tests were conducted following methods specified by U.S. EPA (1995a), U.S. EPA/Corps (1998), and ASTM (1998) with consideration of conditions described in Anderson et al. (1996, 2001); and the echinoderm tests were conducted following methods specified by U.S. EPA (1995a) and Carr and Chapman (1992, 1995).

Five replicate analyses were conducted for each test at each sampling station. For the amphipod and echinoderm tests, the replicates were prepared at the laboratory by subsampling the homogenized sediment collected at each station. For the bivalve tests, the replicates were prepared in the field by collecting six small cores from the grab sampler at each station (including one core collected for water quality measurements).

6.1 Sediment Toxicity Test Results

Results of the amphipod, echinoderm, and bivalve toxicity tests for all triad stations are shown in Tables 6-1, 6-2, and 6-3, respectively. Results for the control samples run with each batch of shipyard samples are shown in Tables 6-4 through 6-6. Results for the amphipod serial dilution test are shown in Table 6-7. The serial dilution test results are also shown in Figures 6-1 and 6-2; the endpoint in these figures has been expressed as percent mortality (100 minus percent survival) and scaled so that all mortality values are expressed as a fraction of the mortality in the undiluted sample. A linear response with sample dilution can be seen in both Figures 6-1 and 6-2, indicating that the mortality response is dependent on a physical or chemical characteristic of the samples. All sediment toxicity data are provided in Appendix G.

6.2 Determination of Toxic Effects

Toxicity determinations were made by comparing the results from the shipyard samples to the data from the final reference pool stations, as specified by Regional Board staff (Barker 2003, pers. comm.). The amphipod test was conducted at the final reference pool stations used by the Bight ’98 study, the Navy’s 2001 Chollas/Paleta Creeks study, and the Phase 1 shipyard study, and all of these data were used to evaluate amphipod toxicity data at the shipyards. The echinoderm test was conducted only by the Navy’s 2001 study as well as the Phase 1 study, and the bivalve test was conducted only by the Phase 1 shipyard study; only the appropriate subset of the final reference pool stations was used to evaluate the results of these toxicity tests.

Regional Board staff specified that the 95%LPL of final reference pool conditions be used as a threshold to identify statistically significant toxicity (Barker 2003, pers. comm.). This is not the most appropriate statistic to use, because it does not account for variability among replicates at the shipyard stations. Therefore, in addition to using the 95%LPL, a more appropriate statistical technique, Dunnett's test (Zar 1996), was also used to evaluate the shipyard data with respect to final reference pool conditions. For the Dunnett's test, a one-tailed experiment-wise 95 percent confidence level was used. For all three types of toxicity tests, both the 95%LPL and the Dunnett's test identified exactly the same set of shipyard stations as significantly different from the final reference pool. The equivalence of these results is attributable to the precision of the replicates at each shipyard station: the variation among replicates is small enough that it did not affect the results of the tests. Because identical results were obtained with both statistical methods, the statistical significance of the results is presented without further reference to the method used.

Statistically significant differences from the final reference pool were found for amphipod survival and bivalve development. Echinoderm fertilization was not significantly different at the shipyards than at the reference area. The range of bioassay responses at each station, the mean response at each station, and the stations that are significantly different from reference area conditions are shown in Figure 6-3 (amphipod test), Figure 6-4 (bivalve test), and Figure 6-5 (echinoderm test). The ranges of endpoint values for the amphipod tests are relatively narrow. Mean survival was 70 percent or greater at all stations. In comparison, the BPTCP used a cutoff value of 48 percent amphipod survival to identify results that were significantly different from reference conditions. In contrast to the amphipod results, the range of responses for the bivalve test is quite wide and includes several stations that exhibit severe responses. The high variability of bivalve results at shipyard stations was also found at reference area stations, where both within-station and between-station variability is high. The high variability may be a consequence of the generally low reliability of larval tests (U.S. EPA 1992b) and the modified exposure method used in this test. The variability of these results suggests that the results of the bivalve test be interpreted with caution.

The spatial distribution of statistically significant effects is shown in Figure 6-6. There is little concordance between the results of these tests: only one station, SW27, showed statistically significant effects in both the amphipod and bivalve tests. This lack of concordance could result from:

- The two tests responding to different types of sediment conditions or chemicals.

- Low amphipod toxicity—The statistically significant responses may represent one end of a single distribution of toxicity responses at the shipyards with a high mean survival. The stations found to have lower survival may result from random sampling effects rather than from actual spatial variations in toxicity.

- Unreliability of the bivalve test—Because of the high variability observed and the nature of the test itself, the stations found to have statistically significant effects may represent the result of statistical fluctuations rather than actual spatial variations in toxicity.

The relationship between toxicity test results and sediment chemicals is discussed in Section 9, *Assessment of Potential Effects on Aquatic Life.* However, the data collected as part of this investigation do not allow a conclusive determination of the cause for the lack of concordance between the amphipod and bivalve tests.

7 Bioaccumulation Test Results

Sediment bioaccumulation tests were conducted during Phase 1 to determine whether indigenous fauna would be sampled in Phase 2. Bioaccumulation tests were conducted using sediment from four stations in the NASSCO leasehold, five stations in the Southwest Marine leasehold, all five of the original reference area stations, and a control sample. Because Regional Board staff subsequently directed the shipyards to sample indigenous fauna in Phase 2 regardless of the results of the bioaccumulation tests (Robertus 2002b, pers. comm.), the bioaccumulation test data have not been used. The data are presented in Appendix I, and tissue concentrations from the bioaccumulation tests in relation to sediment data are briefly described here.

Analyses were conducted and completed as planned. At one station (NA20) the mass of clam tissue in two of the replicates was less than the minimum needed for all of the chemical analyses, and tissue from these two replicates was therefore combined for analysis.

Examination of the chemical concentrations in *Macoma* tissue relative to the chemical concentrations in sediment indicates that bioaccumulation of chemicals is occurring. For many chemicals, concentrations in tissue increase as chemical concentrations in sediment increase, as shown in Figures 7-1 through 7-10 (replicate data for each station have been averaged in these figures). Linear regression models were fit to these relationships to assess their statistical significance (at $p = 0.05$). (Regressions were performed using both untransformed and log-transformed data, with equivalent results.) No statistically significant relationships were found for cadmium, chromium, nickel, selenium, silver, or PCTs. The relationships for arsenic and zinc, although statistically significant, are each controlled by a single data point. The relationship for mercury, although statistically significant, has a very low slope. The results of these tests therefore provide an indication of the bioaccumulation potential of chemicals in the shipyard sediment.

8 Evaluation of Benthic Macroinvertebrates and Fishes

Benthic macroinvertebrates and fishes were sampled from the shipyards and reference areas to evaluate the condition of biological communities currently living at the shipyards.

Information on the condition of benthic macroinvertebrate communities was obtained using both SPI photographs and sediment grab samples from the shipyards and from reference areas. The SPI photographs provide a categorical assessment of the maturity of the benthic community, and the grab samples provide a quantitative assessment of the species assemblages actually inhabiting the shipyard sediments. These data allow a comprehensive assessment of the sediment-dwelling community. Because the benthic macroinvertebrate data were collected as part of the triad study, the community assessments derived from these data can be closely associated with sediment chemical concentrations.

Fish were collected inside and outside the shipyards and at a reference area using a variety of trawls and hooks. Histopathological examination of the fish and analyses of fish bile for PAH breakdown products were carried out to evaluate the exposure of fish to chemicals or other stressors at the shipyards.

8.1 Benthic Macroinvertebrate Communities

Benthic macroinvertebrate sampling—both SPI and grab sampling—was conducted in August 2001 as part of the Phase 1 investigation of the NASSCO and Southwest Marine shipyards. SPI photographs were taken at 101 shipyard stations and 5 reference stations, and benthic macroinvertebrate samples were collected at 30 shipyard stations and 5 reference area stations (the triad stations). This section discusses the results for both the SPI and grab samples. Data tables and SPI photographs are contained in Appendices A and K. The discussion in this section focuses on analyses of similarities among stations, including differences between reference and shipyard stations. Benthic community conditions at the shipyards have been evaluated primarily in relation to the data collected synoptically at the five reference stations sampled in Phase 1. In addition, as specified by Regional Board staff (Barker 2003), the newly proposed BRI for southern California bays has been used to evaluate the shipyard data with respect to benthic data from the final reference pool. Because of inconsistencies in the method used to develop the BRI, lack of independent validation, and lack of peer review, the BRI is not considered to be as reliable an indicator of benthic macroinvertebrate conditions as the more thorough analyses of community characteristics. Issues with the BRI approach are described more fully in Appendix M. An evaluation of the benthic macroinvertebrate data in relation to other triad data is presented in the section titled *Assessment of Potential Effects on Aquatic Life*.

8.1.1 Sediment Profile Photographs

SPI photographs were collected throughout the shipyards and at the reference stations, including 57 stations at NASSCO and 43 stations at Southwest Marine. Typically, three vertical profile photographs were taken at each station. The high density of SPI stations allows an assessment of benthic habitat quality and continuity throughout the shipyards. A variety of indicators of benthic habitat quality can be assessed from the SPI photographs, including the redox potential discontinuity (RPD) depth and the presence of methane bubbles. Both of these measures indicate the extent of anoxic conditions in the sediment, and may control the distributions of some benthic fauna. The SPI photographs also allow direct assessment of the type and density of benthic macroinvertebrates in the sediment. These data can be used to identify the benthic successional stage present, and thereby indicate the relative amount of physical disturbance present.

8.1.1.1 Redox Potential

Differences in the reflectance of sediments in the SPI photographs can be used to estimate the apparent RPD depth: surface sediment with an oxidized surface has a higher reflectance than the dark anoxic layer below. The RPD depth is affected by the diffusion rate of oxygen into the sediment from overlying water, and by sediment mixing resulting from bioturbation by benthic macroinvertebrates. In the absence of bioturbation, the rate of oxygen diffusion into the sediments, and the rate of its consumption by redox reactions, results in a typical RPD depth of

2 mm in mud (Rhoads 1974). Bioturbation increases the RPD depth (Rosenberg 2001). The reflectance boundary observed in SPI photographs is referred to as an apparent RPD depth; in the presence of bioturbation, the actual RPD depth may be slightly less than the reflectance boundary as a result of the downward mixing of oxidized sediment. Although *in situ* measurements of redox potential were not made or used to calibrate the apparent RPD depth observed at the shipyards, the apparent RPD depth nevertheless serves as a means of detecting the presence or absence of bioturbation—and hence of benthic macroinvertebrates.

RPD depths at the shipyards generally ranged from about 1 to 2.5 cm, with a low of 0.71 and a high of 6.6 cm. RPD depths at the reference stations were similar (not statistically different, $p = 0.31$ by two-tailed analysis of variance [ANOVA]), with a low of 1.3 and a high of 5.5 cm. The larger range at the shipyard stations is due to the larger number of stations, among which it is more likely to see higher or lower values from the overall distribution. These RPD depths indicate that bioturbation by benthic macroinvertebrates is occurring at all shipyard locations, and at an intensity comparable to reference areas.

8.1.1.2 Sediment Methane

In anaerobic sediment, organic material is broken down by bacteria that preferentially reduce sulfate to sulfide. Under conditions of high organic loading, where sulfate and other electron acceptors are scarce, biodegradation will be carried out by archaeal microbes that reduce carbon dioxide to methane. The methane generated by this process can appear in the sediment as small bubbles. These methane bubbles can be observed in SPI photographs, characteristically appearing as roughly circular voids with a glassy appearance produced by reflection of the camera strobe light. The appearance of methane bubbles in the sediment is indicative of organic-rich anaerobic conditions, and under these conditions, the benthic community is likely to be affected by the presence of high concentrations of organic matter and the resulting anaerobic conditions.

Methane bubbles were observed in only 3 of the 326 SPI photographs. Each of the occurrences of methane bubbles was at a different station, and in only one of three to five replicate photographs at each of those stations. This low incidence of methane bubbles indicates that there are no areas at which high levels of organic enrichment would be expected to affect the composition of the benthic macroinvertebrate community.

8.1.1.3 Macroinvertebrate Community Successional Stages

Following disturbance or defaunation of soft-bottom marine environments, recolonization by benthic macroinvertebrates typically takes place through a succession of stages, with each stage represented by organisms with different ways of interacting with the sediment (Pearson and Rosenberg 1978; Rhoads and Germano 1982; Rhoads and Boyer 1982). Soon after sediment has been disturbed, it is colonized by small tube-dwelling polychaetes that feed at the sediment surface (Stage I). These polychaetes can appear in very high densities, and the tubes they build modify the structure of the sediment surface. After the initial establishment of Stage I communities, other organisms then become established. Stage II of the succession is characterized by organisms that burrow shallowly into the sediment but nevertheless feed at or near the sediment surface. Burrowing activity loosens and aerates the sediment, a process that makes it more suitable for further colonization. Stage III is characterized by organisms that burrow well into the anaerobic sediment and feed at depth off of organic matter and microbial decomposers. These deep burrowing organisms typically irrigate their burrows with oxygenated surface water. Both Stage II and Stage III organisms are typically larger than the initial Stage I colonizers, and are present in lower population densities. The Stage III community is regarded as the mature stage of a fully developed benthic community.

The three characteristic benthic successional stages can be identified in SPI photographs through the structures that the organisms create (tubes, burrows) and through the modifications they induce in sediment properties.

SPI photographs show that mature Stage III communities are present throughout both shipyards (Figure 8-1 and Appendix A). In many locations, Stage I fauna are found in conjunction with the Stage III communities, suggesting that a moderate amount of disturbance is present. This disturbance is not so great, however, as to disturb the established mature community of Stage III fauna. In a few locations, only Stage I communities were observed, indicating that a higher level of disturbance has recently occurred in those areas. One notable location in this regard is between Piers 4 and 5, near the southeast end of the NASSCO shipyard. All of the stations nearest the shore in this location have only Stage I communities. This observation is consistent with the usage of these piers: engine tests on completed vessels are conducted in those locations, and the amount of water movement resulting from the propeller action of fixed vessels is very likely to create physical disturbances in the sediment. Visible erosion of the bank is also present in this area, corroborating the likelihood of sediment disturbance. Other locations at which Stage I communities are found, or a combination of Stage I and Stage III communities, may also be affected by physical disturbances due to ship movements within the shipyards.

Overall, the SPI photographs show that there are no defaunated areas at the shipyards, and that mature benthic communities are present throughout most of the area.

8.1.2 Summary of Benthic Macroinvertebrate Grab Sampling and Taxonomic Identification

Benthic macroinvertebrates were collected from 15 stations at NASSCO, 15 stations at Southwest Marine, and 5 reference stations in San Diego Bay. Station locations are shown in Figures 2-1 and 8-2. The macroinvertebrate samples were collected as part of sediment quality triad sampling conducted at all of these stations. Five replicate 0.1-m^2 samples were collected at each station. The samples were sieved on a 1-mm screen in the field, and preserved for later examination. Large volumes of material were collected at many of the stations, and samples were split in the laboratory to reduce the sorting time required. Organisms were then removed from these samples and identified to the lowest taxonomic level possible. A quarter of the samples were re-sorted to provide a quantitative estimate of uncertainty in the species abundance estimates. All organisms found during re-sorting are included in the data tables in Appendix K, and abundances are expressed in terms of number of individuals per 0.1 m^2.

8.1.3 Detailed Evaluation of Benthic Macroinvertebrate Communities

The benthic macroinvertebrate data were evaluated in terms of reference area conditions and comparability between reference stations and shipyard stations. This information was then used to assess differences in the benthic macroinvertebrate community at shipyard and reference stations. Similarity of benthic communities at all stations were also calculated, and the result used to construct a dendrogram (cluster analysis) and to carry out a non-metric multi-dimensional scaling (MDS) analysis. Cluster analysis (U.S. EPA 1977) and MDS (Minchin 1987) are multivariate techniques that incorporate all abundance data for all species at all stations into a single analysis. Both are established techniques for the analysis of ecological data (Legendre and Legendre 1998). The following sections discuss the results of these evaluations. All abundance data are shown in Appendix H, along with summaries of the abundances of the major taxa.

8.1.3.1 Reference Area Conditions

The five reference area stations are generally similar to one another in terms of overall abundance and abundances for major taxonomic groups (polychaetes, molluscs, crustaceans, and echinoderms). However, there are several distinct features that distinguish some reference stations from the others (Table 8-1). These distinguishing features include:

- The dominance of Station 2231 (in the center of San Diego Bay to the northwest of the shipyards) by the tanaidacean *Kalliapseudes crassus,* and the large number of species (richness) at this station

- Greater numbers of echinoderms at Station 2441 (near the mouth of the bay) relative to other reference stations

- A gradient in species composition from Station 2441 to Station 2243 (along the axis of the bay).

Of approximately 60 different taxa observed in each replicate at Station 2231 (118 different taxa overall), only one species, *Kalliapseudes crassus*, accounted for 85 to 90 percent of the total organisms in each replicate. Whereas only two other species were represented by more than 100 individuals in any replicate, each replicate contained thousands of *K. crassus.* At the other four reference stations, *K. crassus* was either absent (i.e., Stations 2441, 2433, and 2440) or extremely rare (i.e., two individuals for Station 2243). Despite the overwhelming dominance of *K. crassus* at Station 2231, that station had a markedly higher species richness than any other reference station. Replicates at the other reference stations had 30 to 50 taxa, in contrast to 60 or more taxa in each replicate at Station 2231. *K. crassus* is a tube-building tanaidacean, and the fibrous material observed during sampling at Station 2231 was likely tubes of this species. Differences in the physical structure of the sediment caused by these tubes are likely responsible for the increased richness observed at Station 2231, by providing both substrate and physical refugia for additional species. Because *K. crassus* is not considered native to San Diego Bay, the very high abundance of the species at Station 2231 renders this station anomalous with respect to the benthic community. Benthic macroinvertebrate data from Station 2231 therefore were not pooled with the other reference data for comparison to shipyard stations.

Echinoderms were present at low numbers at Stations 2433, 2440, and 2243 (in the northern and central parts of the bay; these stations had 0–10 individuals in each replicate), whereas the replicates at Station 2441 (near the mouth of the bay) had echinoderm abundances of 16 to 48 individuals. Two types of echinoderms, an ophiuroid (brittle star) and a holothuroid (sea cucumber), were present at Station 2441 at higher abundances than at other reference stations. Because of the low numbers of echinoderms at reference stations other than Station 2441, and the anomalously high numbers at Station 2441, echinoderms were not included in the set of major taxonomic groups used for statistical comparisons between shipyard stations and reference stations.

The composition of the benthic macroinvertebrate community changes somewhat along the gradient from Station 2441 to Station 2243 (i.e., proceeding from the station nearest the bay mouth to the station deepest in the bay), although the polychaete taxon *Lumbrineris* sp. was one of the three most abundant taxa at all four stations. The three most abundant taxa at Station 2441 (i.e., nearest the bay mouth) were the polychaetes *Lumbrineris* sp. and *Leitoscoloplos pugettensis*, as well as the cnidarian *Edwardsia californica*. At Station 2433, the three most abundance taxa included the same two polychaetes identified for Station 2441, as well as the polychaete *Diplocirrus* sp. SD1. At Station 2440, the three most abundance taxa also included the two polychaetes identified for Station 2441, as well as the polychaete *Pista percyi*. Finally, at Station 2243, the three most abundance taxa included the polychaetes *Lumbrineris* sp. and *Exogene lourei*, as well as nematodes.

8.1.3.2 Overview of Benthic Communities

In this section, an overview is provided of the major taxonomic compositions of the benthic communities found at the shipyard sites and in the reference area. Most marine benthic communities are dominated by species belonging to three major taxonomic groups: Polychaeta, Mollusca, and Crustacea. Figure 8-3 and Table 8-2 present the relative composition of these three major taxa for communities within the three major study sites: NASSCO shipyard, Southwest Marine shipyard, and the reference area. Data from all stations within each major study site were pooled for this analysis.

Benthic communities in all three study sites were dominated by polychaetes, both with respect to relative abundance and relative number of taxa. Both the relative abundance and relative taxa richness of polychaetes within the three study sites were nearly identical, ranging from 65 to 69 percent and 37 to 39 percent, respectively. The relative abundance and relative taxa richness of crustaceans within the three sites were also nearly identical, ranging from 12 to 15 percent and 23 to 25 percent, respectively. Molluscs showed some differences among the sites, with relative abundance at the shipyard sites (15 percent at each site) being slightly greater than the value of 9 percent found in the reference areas. By contrast, relative taxa richness at the shipyard sites (15–18 percent) was slightly lower than the value of 22 percent found in the reference areas. Overall, the comparisons based on major taxonomic composition of benthic communities showed that communities at the shipyard sites were very similar to communities in the reference areas.

The major taxonomic composition of benthic communities found at individual shipyard and reference area stations is presented in Figures 8-4 through 8-7 and Table 8-2. Those results show that on a station-specific basis, some differences were found between various shipyard stations and the reference area stations. Those differences are evaluated statistically in the following section (Section 8.1.3.3).

In addition to comparisons based on major benthic taxa, the 10 most abundant benthic taxa at the shipyard sites and reference areas were evaluated (Table 8-3). Those results showed that 6 of the 10 most abundant taxa in the reference areas (all polychaete taxa) were included within the 10 most abundant taxa at the shipyard sites. These six taxa are *Lumbrineris* sp., *Exogene lourei*, *Leitoscoloplos pugettensis*, *Mediomastus* sp., *Pista alata,* and *Scyphoproctus oculatus*. The remaining four abundant taxa within the shipyard sites included the polychaete *Pseudopolydora paucibranchiata*, the molluscs *Musculista senhousia* and *Theora lubrica*, and the crustacean *Synaptotanais notabilis*. Three of these taxa (i.e., *M. senhousia*, *T. lubrica*, and *P. paucibranchiata*) are not native to Southern California and have been introduced to the region. The comparisons of the most abundant benthic taxa between the shipyard sites and the reference area showed that there were numerous similarities between the dominant taxa communities in the three study sites.

8.1.3.3 Comparison of Benthic Metrics between Shipyard and Reference Stations

Six different quantitative measures (or metrics) of the benthic macroinvertebrate community were used to contrast shipyard and reference stations. These benthic metrics are:

- Total abundance
- Abundance of major taxonomic groups (polychaetes, molluscs, and crustaceans)
- Total taxa richness—The number of distinct taxa
- Swartz' dominance index (SDI)—The minimum number of taxa making up 75 percent of the total abundance
- Percent dominance—The fraction of all organisms represented by the three most abundant species

- Shannon-Wiener diversity index (H′)—A measure of both the number of species and the distribution of individuals among species; higher values indicate that more species are present or that individuals are more evenly distributed among species.

The mean and range of each benthic metric at each shipyard and reference station are presented in Figures 8-8 through 8-15.

Reduced abundance, reduced richness, reduced SDI, increased dominance, and reduced H′ are typically found at sites that have been affected by toxic chemicals or physical disturbance. Elevated abundances of selected (pollution-tolerant) taxa, such as some polychaete species, may also be found at disturbed sites. High polychaete abundance and low taxa richness are typically indicative of communities that are in the early stages of recovery from physical disturbances or organic enrichment. Toxicity is ordinarily manifested as decreased abundances, particularly of sensitive taxa such as crustaceans (Long et al. 2001).

Taxa richness, SDI, percent dominance, and H′ are all indicators of the structure of the benthic community. Taxa richness is simply the total number of different taxa, and is the simplest of all measures of diversity. Decreases in taxa richness often result from the absence of rare species. SDI provides a measure of the number of dominant taxa. Disturbed communities often become dominated by a few opportunistic species that may be tolerant to pollution. Therefore, decreases in SDI are frequently found in disturbed habitats. Percent dominance is another measure of dominance that considers the abundance of dominant taxa rather than the number of dominant taxa. H′ incorporates measures of both the number of taxa and the relative abundance of different taxa, assigning lesser weight to rare species. Because all of the benthic metrics provide related measures of the structure of benthic communities, significant differences between reference stations and shipyard stations for multiple indicators represent a greater weight of evidence for differences in the benthic community structure.

Statistical comparisons between shipyard and reference stations were carried out using parametric (Dunnett's) or non-parametric (Wilcoxon rank-sum) techniques, as appropriate to each variable. Each variable was individually tested for homogeneity of variance, using the original variates, logarithmic transformation, and square-root transformation. If homogeneity of variance was found using either the original or transformed variates, a parametric test was used; otherwise, a non-parametric test was used. The transformation and type of test used for each variable is shown in Table 8-4. All statistical tests were one-tailed tests, carried out with an experiment-wise significance (alpha) level of 0.05. All variables except percent dominance were tested for decreases relative to reference conditions; percent dominance was tested for an increase relative to reference.

The results of these statistical tests are shown in Table 8-5.

8.1.3.4 Classification Analysis of Benthic Communities

In addition to the univariate statistical comparisons of benthic metrics at shipyard and reference stations described above, a multivariate analysis of the benthic communities at all stations was conducted using the Bray-Curtis measure of similarity (Boesch 1977; Hruby 1987). Classification analysis provides an integrative evaluation of all benthic taxa and has the power to detect relatively subtle patterns. Norris and George (1993) concluded that multivariate techniques show greater promise than univariate comparisons for detecting and understanding spatial and temporal trends of benthic macroinvertebrate communities.

Prior to conducting the classification analysis, abundance data for individual benthic taxa were log-transformed to reduce the influence of the most abundant taxa on the results of the analysis. In this manner, less numerous and rare taxa were an increased influence on the results of the analysis. Stations were clustered based on similarity in a dendrogram (Figure 8-16). In this diagram, stations that are most similar with respect to species composition of the benthic communities are clustered more closely together at the right side of the diagram. Clusters joined at successively higher levels of the dendrogram (to the left of the diagram) are relatively more different with respect to community composition. The dendrogram provides a means of visually interpreting the quantitative results of the similarity calculations. The dendrogram is annotated with the results of the evaluation of differences in macroinvertebrate communities, which is discussed in the following section.

The classification analysis identifies seven major station groups:

- **Station Group 1:** NASSCO Station NA22, located near the mouth of Chollas Creek
- **Station Group 2:** The two innermost reference stations (2440 and 2243)
- **Station Group 3:** Twelve stations (7 NASSCO, 5 Southwest Marine), with most exhibiting statistically significant differences from reference stations for one or more benthic metrics
- **Station Group 4:** Thirteen stations (7 NASSCO, 6 Southwest Marine), with all exhibiting either no alterations or minor differences from reference stations based on benthic metrics
- **Station Group 5:** Two adjacent Southwest Marine stations (SW13 and SW15) located in a dry dock area
- **Station Group 6:** Two adjacent Southwest Marine stations (SW04 and SW08) located in a shallow protected area
- **Station Group 7:** The two outermost reference stations (2441 and 2433).

Most stations located in close geographic proximity cluster closely based on the characteristics of their benthic communities. Most stations with one or no differences based on the benthic metrics cluster closely with other stations having one or no differences.

8.1.3.5 MDS Analysis of Benthic Communities

The abundance of each individual taxon can be considered a separate dimension along which the stations can be arranged. Considering all taxa simultaneously, each station can be represented by a point in a multidimensional space. Because of the large number of dimensions, these relationships cannot be visualized. However, when different species covary, many of these different dimensions convey the same information, and a simpler representation of the data is possible. MDS is a multivariate statistical technique that reduces the number of dimensions needed to represent the data, while preserving the ordering relationship of the similarities between stations. MDS calculates coordinates in a reduced number of dimensions such that the distance between points (stations) is related as closely as possible to their similarity. MDS is related to other factor analysis methods but does not assume a linear relationship between distance and dissimilarity. The resulting dimensions are independent measures of similarities between stations, and do not necessarily correspond to the abundances of any particular species. MDS is an established technique for analysis of benthic species abundance data (Zenetos and Papathanassiou 1989; Help et al. 1988).

Before the MDS analysis was conducted, abundance data were log-transformed and the Bray-Curtis similarity between each pair of stations was calculated, as for the cluster analysis. MDS was used to generate both two-dimensional and three-dimensional representations of the data. The two-dimensional representation accounted for 95.8 percent of the variance in inter-station similarity, and the three-dimensional representation accounted for 96.5 percent of the variance. Thus, two dimensions are sufficient to represent most of the variability between stations, and increasing the number of dimensions does not markedly improve the explanatory power of the MDS results. All Phase 1 stations are shown in relation to the two MDS dimensions (axes) in Figure 8-17.

Nonmetric MDS largely preserved the seven station groups identified by the classification analysis. Examination of the MDS plot reinforces several features of the data set that have been previously noted, including:

- Station 2231 is different from all other shipyard and reference stations.
- Other reference stations are arranged approximately along a gradient from the lower left to the upper right of the plot, suggesting that this gradient represents the transition in benthic populations from the mouth of the bay to

the inner bay. Most shipyard stations fall between Stations 2440 and 2243 along this gradient. Stations 2441 and 2433 are substantial outliers.

- Station NA22 is distinctly different from all other shipyard stations.
- Stations SW04 and SW08 are similar to each other, but different from other shipyard stations.

In addition, the MDS results indicate that Station SW15 is relatively different from other shipyard stations, although no statistically significant differences in abundance or other metrics were found.

The two major station groups (Station Groups 3 and 4) exhibited a gradient of alterations based on benthic metrics from none/minor to moderate to major alterations arranged along an axis perpendicular to the axis described by 2440 and 2243. Station Groups 5 and 6 were moderate outliers from the two major groups. The spatial distribution of shipyard stations from the various station groups is presented in Figure 8-18. The following spatial patterns are apparent:

- **Station Group 1:** The single station in this group (NA22) was located at the southeast boundary of the site and is the station closest to Chollas Creek
- **Station Group 3:** Six stations (NA04, NA05, NA11, NA12, NA15, NA16) were clustered in the central part of a large open area in the southeast part of the site; three stations (SW21, SW22, and SW23) were clustered in a confined nearshore area in the northwest part of the site; and three stations (SW03, SW17, and NA20) were isolated in various parts of the site
- **Station Group 4:** Eight stations (SW02, SW09, SW11, SW18, SW25, SW27, NA01, and NA03) were located in a relatively continuous band along the offshore area of the northwest part of the site; five stations (NA06, NA07, NA09, NA17, and NA19) were located in a relatively continuous band along the nearshore area of the southeast part of the site

- **Station Group 5:** Both stations from this group (SW13 and SW15) were located adjacent to each other in a dry dock area in the northwest part of the site

- **Station Group 6:** Both stations from this group (SW04 and SW08) were located adjacent to each other in a shallow protected area in the northwest part of the site.

8.1.3.6 Species Clusters

A second classification analysis was conducted in which benthic macroinvertebrate species were grouped using the Bray-Curtis measure of similarity and log-transformed abundances. The classification analysis resulted in seven major benthic groups. However, three of the groups included most of the major benthic taxa found throughout the study site:

- **Benthic Group 1:** Three very abundant and widespread species (the mussel *Musculista senhousei*, the syllid polychaete *Exogene lourei*, and the spionid polychaete *Pseudopolydora paucibranchiata*)

- **Benthic Group 2:** Three very abundant and widespread species (the bivalve *Theora lubrica*, the terebellid polychaete *Pista percyii*, and the orbiniid polychaete *Leitoscoloplos pugettensis*), as well as one less abundant species (the spionid polychaete *Scolelepis* sp.)

- **Benthic Group 3:** Thirteen species arranged as four subgroups and two outliers:

  - **A:** Nematodes, oligochaetes, and the nereid polychaete *Neanthes acuminata*

  - **B:** Two polychaetes (the capitellid *Mediomastus* sp. and the spionid *Prionospio heterobranchia*)

- **C:** Two polychaetes (the dorvilleid *Dorvillea longicornis* and the polynoid *Harmothoe imbricata*), as well as one crustacean (the amphipod *Grandidierella japonica*)

- **D:** Three crustaceans (the amphipods *Podocerus fulanus*, the isopod *Paracerceis cordata*, and the tanaid *Synaptotanais notabilis*)

- **Outliers:** The capitellid polychaete *Scyphoproctus oculatus* and the cnidarian *Scolanthus* spB.

8.1.3.7 Station Clusters Based on Major Benthic Groups

An additional classification analysis of sampling stations was conducted using the Bray-Curtis measure of similarity and log-transformed abundances of only those species found in Benthic Groups 1, 2, and 3. The classification analysis resulted in seven major station groups, which were similar to the station groups discriminated previously on the basis of all benthic taxa. Most stations clustered into two major groups, with Station Group 2 containing stations having no or minor alterations and Station Group 3 primarily containing stations having moderate or major alterations.

A key difference between the two classification analyses of stations was that two reference stations clustered somewhat closely with the two largest station groups in the analysis based on the three benthic groups. Station 2440 clustered somewhat with Station Group 3, and Station 2243 clustered somewhat with Station Group 2. The benthic community at Station 2440 was dominated by members of Benthic Group 2, whereas Station 2243 was dominated by members of Benthic Groups 1 and 3.

The spatial distribution of shipyard stations from the various station clusters based on benthic groups is presented in Figure 8-19 and was similar to the spatial patterns based on all benthic taxa (see Figure 8-18):

- **Station Group 2:** All 17 stations (SW02, SW09, SW11, SW13, SW15, SW18, SW25, SW27, NA01, NA03, NA05, NA06, NA07, NA09, NA11, NA16, and NA19) were located in a relatively continuous band along the northwest and central parts of the site

- **Station Group 3:** Six stations (SW03, NA04, NA12, NA15, NA17, and NA20) were scattered throughout the site, but three stations (SW21, SW22, and SW23) were clustered in a confined area in the northwest part of the site

- **Station Group 4:** The single station from this group (SW17) was located in the northwest part of the site

- **Station Group 5:** Both stations from this group (SW04 and SW08) were located adjacent to each other in a shallow protected area in the northwest part of the site.

- **Station Group 6:** The single station in this group (NA22) is located at the southeast boundary of the site and is the station closest to Chollas Creek.

8.1.3.8 Identification of Potential Benthic Indicator Species

Indicator species are those that are typically found in a specific set of environmental conditions. For the purpose of assessing environmental health, indicator species are generally classified as either pollution sensitive or pollution tolerant. Lowe and Thompson (1999) conducted a literature review of benthic studies from throughout the world, eliminating studies that focused exclusively on the effects of organic enrichment. This section presents a summary of the potential tolerances to chemical contamination for the species constituting the three major benthic groups discriminated in the present study. The summary is based largely on the study by Lowe and Thompson (1999), but other references (primarily for California) are cited in some cases. Each benthic species is listed as sensitive or tolerant to chemical contamination, unless no information was found.

- **Benthic Group 1**
  - *Musculista senhousei*—No information found
  - *Exogene lourei*—No information found
  - *Pseudopolydora paucibranchiata*—Considered sensitive by Olsgard (1999); *P. kempi* is considered tolerant by Lowe and Thompson (1999)
- **Benthic Group 2**
  - *Theora lubrica*—Considered tolerant by Ferraro and Cole (1997) and Lowe and Thompson (1999)
  - *Pista percyii*—No information found
  - *Leitoscoloplos pugettensis*—No species-specific information found, but *Leitoscoloplos* spp. are considered tolerant by Lowe and Thompson (1999)
  - *Scolelepis* sp.—No information found
- **Benthic Group 3**

  Subgroup A

  - Nematodes—Considered inconclusive by Lowe and Thompson (1999)
  - Oligochaetes—Considered tolerant by Anderson et al. (2001) and Hunt et al. (2001)
  - *Neanthes acuminata*—No species-specific information found, but *N. succinea* is considered tolerant by Lowe and Thompson (1999)

Subgroup B

- *Mediomastus* sp.—Considered tolerant by Fairey et al. (1996) and Lowe and Thompson (1999)
- *Prionospio heterobranchia*—No species-specific information found, but *P. cirrifera* is considered tolerant by Lowe and Thompson (1999) and sensitive by Olsgard (1999)

Subgroup C

- *Dorvillea longicornis*—Considered tolerant by Fairey et al. (1996); also Dorvilleidae and *D. rudolphi* are considered tolerant by Lowe and Thompson (1999)
- *Harmothoe imbricata*—Considered sensitive by Fairey et al. (1996)
- *Grandidierella japonica*—Considered tolerant by Swartz et al. (1994), Fairey et al. (1996), and Lowe and Thompson (1999)

Subgroup D

- *Podocerus fulanus*—No species-specific information found, but Podoceridae and *P. spongicolus* are considered sensitive by Lowe and Thompson (1999); in addition, crustaceans are generally considered sensitive (Lowe and Thompson 1999; Long et al. 2001; Anderson et al. 2001; Hunt et al. 2001)
- *Paracerceis cordata*—No information found, but crustaceans are generally considered sensitive (Lowe and Thompson 1999; Long et al. 2001; Anderson et al. 2001; Hunt et al. 2001)
- *Synaptotanais notabilis*—No information found, but crustaceans are generally considered sensitive (Lowe and Thompson 1999; Long et al. 2001; Anderson et al. 2001; Hunt et al. 2001)

Outliers

- *Scyphoproctus oculatus*—No information found
- *Scolanthus spB*—No information found.

8.1.3.9 Benthic Community Composition at Selected Stations

In this section, benthic community characteristics are described for the stations not included in the two largest station groups based on all benthic taxa (i.e., Station Groups 3 and 4).

Station Group 1 (NA22)—In addition to being the only station to exhibit alterations of more than four benthic metrics, this station near the mouth of Chollas Creek and Storm Drain SW9 was the only shipyard station at which the relatively ubiquitous mussel *Musculista* was absent (Table 8-6). In addition, it was the only shipyard station at which the relatively ubiquitous polychaete *Exogene* was nearly absent. Given the major alterations of benthic metrics and relatively unique benthic community found at this site, and the proximity of two pollutant sources, it appears that that the community is adversely affected by toxic chemicals from these sources.

Station Groups 2 and 7 (Reference Stations)—The 10 most abundant species at the 4 reference stations (Table 8-7) exhibited a gradient with respect to the three benthic groups:

- **2441 and 2433:** Only members of Benthic Group 2 were abundant at 2441 (the polychaetes *Leitoscoloplos* and *Pista*) and 2433 (*Leitoscoloplos* and the bivalve *Theora*)
- **2440:** All three species from Benthic Group 2 were abundant, as well as one species from Benthic Group 1 (the polychaete *Exogene*) and two species from Benthic Group 3 (the polychaete *Mediomastus* and the tanaid *Synaptotanais*)
- **2243:** No species from Group 2 was abundant, whereas all three species from Group 1 were abundant, as well as five species from Group 3

(nematodes, the capitellid polychaetes *Mediomastus* and *Scyphoproctus*, the isopod *Paracerceis*, and the amphipod *Podocerus*).

The only species from the three benthic groups that was widespread among the reference stations was the polychaete *Leitoscoloplos*, which was one of the three most abundant species at 2441, 2433, and 2440. However, this species was not among the 10 most abundant taxa at 2243.

Station Group 5 (SW13 and SW15)—These stations are two of only three shipyard stations at which the relatively ubiquitous mussel *Musculista* was not among the 10 most abundant taxa (Table 8-8). In addition, these two stations were the only ones at which the bivalve *Ostrea conchaphila* was among the 10 most abundant taxa. In fact, both the abundance of molluscs and number of molluscan taxa at these two stations were higher than at most other shipyard stations. Both of these stations were located in a dry dock area and review of the field log showed that the substrate at both stations included large amounts of fragmented oyster shells. It therefore is likely that the unusual physical characteristics at these two sites largely accounted for the relative uniqueness of their benthic communities.

Station Group 6 (SW04 and SW08)—Both of these stations were dominated by taxa from Benthic Groups 1 and 3, which comprised 8–9 of the 10 most abundant taxa at these stations (Table 8-9). By contrast, no member of Benthic Group 1 was among the 10 most abundant taxa. In addition, both stations shared the same three most abundant taxa (i.e., the polychaetes *Exogene* and *Pseudoplydora* from Benthic Group 2 and the tanaid *Synaptotanais* from Benthic Group 3), which reached their highest abundances in the study at these stations. The capitellid polychaete *Scyphoproctus* was also very abundant at Station SW08.

Major alterations of benthic metrics were due only to SDI at SW04 and H′ at SW08, which were likely the result of the unusually high abundances of the three and four most numerous taxa, rather than reductions in the numbers or abundances of other taxa. In fact, mean values of taxa richness at the two sites (35.6 and 41.0 taxa, respectively) were similar to the mean reference value of 40.2 taxa, and within the reference range of 34.8–47.8 taxa. In addition, if the three and four most abundant taxa were not considered for SW04 and SW08, respectively, the abundances of the remaining major taxa were generally similar to those at the reference stations.

Based on the patterns described above, it is questionable whether the benthic communities at SW04 and SW08 were adversely affected by toxic chemicals.

8.1.3.10 Assessment of Differences in Benthic Macroinvertebrate Communities

The results of the benthic metrics comparisons, classification analyses, MDS analysis, and evaluations of distributions of distinct groups of taxa were integrated to identify stations at which differences from reference stations are potentially related to toxic chemicals. This analysis did not include consideration of physical factors (e.g., grain size, TOC content, water depth) that can affect benthic communities, nor does it incorporate the results of the other triad analyses. Therefore, the results of this analysis are conservative (protective) with respect to determination of potential effects due to shipyard chemicals.

Stations at which some kind of effect on the benthic communities were classified as having minor, moderate, or major differences from reference area conditions, based on the following criteria:

- **Minor Differences:** A difference was found for only one benthic metric and the station clustered closely with one or more stations at which no differences based on benthic metrics were found.
- **Moderate Differences:** Differences were found for one or two benthic metrics and the station clustered closely with one or more stations with major differences based on benthic metrics. Alternatively, differences were found for two benthic metrics and the station did not cluster closely with any other station.
- **Major Differences:** Differences were found for three or more benthic metrics.

In some cases, the abundance and numbers of species of crustaceans at a station were used to provide additional support for its classification, because that group of organisms is known to be particularly sensitive to chemical toxicity. As shown in Table 8-10, differences from reference areas based on the aforementioned criteria were found at 13 of the 30 shipyard stations. Minor, moderate, and major differences were found at three, two, and eight stations, respectively. The spatial distribution of these differences is shown in Figure 8-20.

Stations SW04 and SW08 are unusual because of the large total abundances relative to reference stations. At both of these stations, the most abundant taxa are polychaetes and a tanaid. The polychaete *Exogone lourei* is the most abundant species at Station SW08 and the second most abundant species at Station SW04, and is also the most abundant species at Station 2243. The tanaid *Synaptotanais notabilis,* which is among the three most abundant species at both Stations SW04 and SW08, is a crustacean, a group generally considered to be sensitive to polluted sediments. In addition, the abundance of the polychaete *Pseudopolydora paucibranchiata*, which is also among the three most abundant species at both Stations SW04 and SW08, has been observed to be negatively correlated to available copper in sediments (Olsgard 1999).

8.1.4 Benthic Response Index

The recently proposed BRI for southern California bays (Smith et al. 2003) has been calculated for the shipyard stations and for the final reference pool (Barker 2003, pers. comm.). This BRI was developed in a manner similar to a BRI developed previously for evaluating benthic communities on the mainland shelf of southern California (Smith et al. 2001). The BRI is the abundance-weighted average of the pollution-tolerance values (i.e., p_i values) that have been assigned to individual benthic species found in the bays of southern California. The p_i value for each species was assigned by Smith et al. (2003) based on the abundance distribution of the species along a gradient of communities that is defined by multivariate ordination analysis. This gradient was interpreted as representing evidence of pollution effects, based in part on association with amphipod bioassay results. Information from 170 stations was used to develop the BRI for southern California bays for benthic communities sampled during the summer using a 0.1-m^2 van Veen grab sampler and a sieve mesh size of 1.0 mm. All of the same sampling conditions applied to the shipyard study.

Smith et al. (2003) derived taxon-specific pollution tolerance scores using data from three different studies of southern California bays. These data were divided into northern and southern subsets, and independent p_i values were calculated for each subset. The authors divided the overall data into two subsets because they concluded that the numbers and kinds of benthic organisms vary between the two areas and comparisons to determine altered benthic communities should vary accordingly. San Diego Bay and the shipyard sites are within the area of the southern subset of data. Those p_i values were developed from benthic data collected from Dana Point Harbor southward to the United States–Mexico border. Using various selection criteria for including or excluding benthic species, Smith et al. (2003) developed p_i values for 162 species from the southern data subset, with p_i values ranging from −112 (i.e., least pollution tolerant) to 227 (i.e., most pollution tolerant).

To interpret the magnitude of the BRI values found at individual stations, Smith et al. (2003) defined a set of assessment thresholds based on changes in biodiversity along the pollution gradient defined by the index values. According to the authors, the thresholds for the bays BRI were functionally and ecologically equivalent to those used for the continental shelf BRI, but were not identical because of differences in the fauna between the two environments. Four response levels were defined according to the magnitude of predicted losses of reference species. The reference conditions and response levels for the bays BRI are as follows:

- **Reference Conditions:** BRI values ≤31
- **Response Level 1:** BRI values >31 but ≤42
- **Response Level 2:** BRI values >42 but ≤53
- **Response Level 3:** BRI values >53 but ≤73
- **Response Level 4:** BRI values >73.

The meaningfulness of these distinctions is discussed later in this section. In the following section, however, these different response levels are used, in part, to present the results of the BRI calculations for the shipyard data set.

8.1.4.1 Calculation of BRI Values

In this section, the following items are discussed:

- Taxonomic issues related to two abundant species with elevated BRI values
- An overview of the shipyard data set with respect to the numbers of species that had p_i values (and could therefore be used in the BRI analysis), as well as the range of those p_i values
- Comparisons of large-scale patterns among the reference stations and shipyard sites with respect to the distribution of species and individuals among the various benthic response levels
- Descriptions of station-specific patterns of BRI values at the reference stations and shipyard sites with respect to magnitude of benthic response levels and spatial distribution of BRI values
- Comparisons of station-specific BRI values with major benthic community metrics.

This section is followed by a critical evaluation of the BRI approach, based on its application to the shipyard benthic data.

Taxonomic Issues—In applying the BRI to the benthic data set for the NASSCO and Southwest Marine shipyards, taxonomic uncertainties were found for two abundant taxa: the polychaete *Pista percyi* and the isopod *Paracerceis cordata,* neither of which has a p_i value for the southern bays. However, two closely related species (*Pista alata* and *Paracerceis sculpta*) have p_i values for the southern bays but were not identified in the Phase 1 shipyards investigation. By contrast, *P. alata* and *P. sculpta* were identified in San Diego Bay during a large-scale benthic survey conducted in 1992–1994 (BPTCP; Fairey et al. 1996), but *P. percyi* and *P. cordata* were not found in that earlier data set. After discussions with benthic macroinvertebrate experts at the taxonomic laboratory used for the shipyard study, as well as the Southern California Coastal Water Research Project, it was found that considerable taxonomic uncertainties currently exist in the identification of these two species. Although *Pista percyi* is currently considered a shallow-water species, and *P. alata* an offshore species, this distinction was not recognized at the time of the BPTCP or Bight '98 studies. Therefore, it is possible that the p_i value assigned to *P. alata* is based on misidentification of *P. percyi*, and should be applied to the *P. percyi* data from the shipyard studies. The two *Paracerceis* species are distinguished by characteristics of mature males, and in samples containing females and immature individuals, *Paracerceis* species identifications may not be accurate. Given these uncertainties, the p_i value for *Pista alata* was applied to *P. percyi* and the p_i value for *Paracerceis sculpta* was applied to *P. cordata*. Because the p_i values for *P. alata* and *P. sculpta* are relatively high (i.e., 66 and 57, respectively), the substitution of those two species in the current data set provided an environmentally conservative evaluation.

Overview of the Shipyard Benthic Data—The majority of the species occurring at shipyard and reference stations[1] had p_i values for southern bays, and those species represented most of the total numbers of individuals found in each of the three study areas. However, at each station, typically 40–50 percent of the species did not have p_i codes (Table 8-11).

The species having p_i values that were present at the reference stations and shipyard sites included most of the abundant taxa from the three areas. For example, of the 20 most abundant taxa at the reference stations, NASSCO site, and Southwest Marine site (which represented 79–85 percent of the total numbers of individuals sampled in the three areas), p_i values were available for 18, 16, and 17 taxa, respectively (Table 8-12). The abundant taxa without p_i values at one or more of the three sites included Nematoda, Oligochaeta, *Scolanthus* sp. B, three polychaete species (*Dorvillea longicornis*, *Harmothoe imbricata*, and *Protocirrineris* sp. A), and one molluscan species (*Lyonsia californica*).

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[1] Benthic data from reference station 2231 are not considered in this evaluation because that station has a highly altered benthic community dominated by an invasive tanaidacean.

The p_i values for the benthic taxa found at the reference stations and shipyard sites ranged from −112 to 150 (Table 8-13). The three most abundant taxa at the shipyard sites with higher p_i values (i.e., >65) were the polychaetes *Leitoscoloplos pugettensis* (p_i value = 94) and *Pista alata* (p_i value = 66) and the mussel *Musculista senhousia* (p_i value = 70). Together, these three species accounted for 23.4 and 16.4 percent of total abundance at the NASSCO and Southwest Marine sites, respectively.

Areawide Patterns—Comparisons of species representative of the various benthic response levels among the reference stations and shipyard sites are presented in Figures 8-21 and 8-22. Most of the species having p_i values in all three areas were representative of reference conditions, ranging from 47 and 52 percent at the shipyard sites to 58 percent at the reference stations (Figure 8-21). Similar percentages of species having p_i values were found among all three areas for Response Levels 1–4, ranging from 8 to 14 percent.

With respect to abundances of individual organisms, the largest numbers of individuals for all three areas were found for Response Level 2, ranging from 33 percent at the Southwest Marine site to 47 percent at the NASSCO site, with an intermediate value of 40 percent for the reference stations (Figure 8-22). The distributions of abundances for the other response levels exhibited differences among the three study areas, with the largest differences found for the reference conditions and Response Level 1. Approximately 32 percent of individuals at the reference stations fell within the "reference" response level, whereas 12–17 percent of individuals at the shipyard sites were grouped in that category. By contrast, 2 percent of individuals at the reference stations were representative of Response Level 1, whereas 8–22 percent of individuals at the shipyard sites were representative of Response Level 1. Similar percentages of individuals were found in the combined Response Levels 3 and 4, ranging from 26 percent for the reference stations to 33 percent for the NASSCO site, with an intermediate value of 28 percent for the Southwest Marine site.

The information discussed above indicates that there were no strong patterns of increasing percentages of species or individuals in higher benthic response levels at the shipyard sites, compared to the reference stations. The patterns observed for numbers of species at the shipyard sites were very similar to the pattern found for the reference stations. For numbers of individuals, the patterns were similar among the three areas for Response Levels 2–4, but differed somewhat for the "reference" level and Response Level l, with the reference stations having a higher percentage of individuals with low p_i scores. These results suggest that conditions at the shipyard sites are not substantially different from conditions at the reference stations.

Station-Specific Patterns—The BRI values for the five reference stations ranged from 17 to 45 and generally increased in magnitude with increasing distance from the mouth of San Diego Bay (Table 8-11). The BRI values for Stations 2441, 2433, and 2231 fell within the "reference" response level, whereas the values for Stations 2440 and 2243 were indicative of Response Levels 1 and 2, respectively. This gradient is consistent with variations in community composition with increasing distance from the mouth of the bay that is revealed by the MDS analyses described previously, and indicates that the BRI scores may be responding to large-scale changes in the physical and hydrodynamic environment within the bay. This gradient can also be observed in the distribution of BRI scores among final reference pool stations (Table 8-14).

For the 30 stations sampled in both shipyards, most stations were classified as Response Level 2 (19 stations), whereas smaller numbers of stations were classified as Response Level 1 (5 stations) or Response Level 3 (6 stations). No station at any shipyard station was classified as Response Level 4 (i.e., the category for the most altered communities). In addition, the BRI values at all six of the stations classified as Response Level 3 were marginal values (i.e., 54–56) that only slightly exceeded the lower bound of that level (i.e., 53), and all six values were well within the lower part of the BRI range for that classification (i.e., >53 but ≤73). Therefore, none of the benthic communities found at the shipyard stations can be considered substantially altered based on the BRI analysis.

The station classifications within the two shipyard sites exhibited differences, with a broader distribution found at the Southwest Marine site. At the NASSCO site, only two response levels were represented: Response Level 2 (12 stations) and Response Level 3 (3 stations). By contrast, at the Southwest Marine site, three response levels were represented: Response Level 1 (5 stations), Response Level 2 (7 stations), and Response Level 3 (3 stations).

At the NASSCO site, all three of the stations classified as Response Level 3 (i.e., NA06, NA17, and NA20) were located in the inner parts of the shipyard (Figure 8-23). The elevated BRI values and the taxa that were largely responsible for the elevated values were as follows:

- **Station NA06:** BRI = 55; responsible taxa were the cnidarian family Edwardsiidae (p_i value = 77), the mussel *Musculista senhousia* (p_i value = 70), two polychaetes (*Neanthes acuminata* complex [p_i value = 120] and *Leitoscoloplos pugettensis* [p_i value = 94]), and the decapod *Pyromaia tuberculata* (p_i value = 96)

- **Station NA17:** BRI = 56; responsible taxa were the cnidarian family Edwardsiidae, the mussel *M. senhousia*, two polychaetes (*N. acuminata* complex and *L. pugettensis*), the amphipod *Mayerella banksia* (p_i value = 150), and the decapod *P. tuberculata*

- **Station NA20:** BRI = 55; responsible taxa were the cnidarian family Edwardsiidae, the mussel *M. senhousia*, the polychaete *L. pugettensis*, and the decapod *P. tuberculata*.

At the Southwest Marine site, two of the three stations classified as Response Level 3 (i.e., SW21 and SW22) were located adjacent to each other in the inner part of the shipyard (Figure 8-23). The third station (SW09) was located midway between the inner and outer parts of the shipyard. The elevated BRI values and the taxa that were largely responsible for the elevated values were as follows:

- **Station SW09:** BRI = 54; responsible taxa were the cnidarian family Edwardsiidae, the mussel *M. senhousia*, five polychaetes (*Aphelochaeta/ Monticellina* complex [p_i value = 97], *Capitella capitata* complex [p_i value = 88], *Pherusa capulata* [p_i value = 122], *N. acuminata* complex, and *L. pugettensis*), the amphipod *M. banksia*, and the decapod *P. tuberculata*

- **Station SW21:** BRI = 54; responsible taxa were the cnidarian family Edwardsiidae, the mussel *M. senhousia*, two polychaetes (*N. acuminata* complex and *L. pugettensis*), and the decapod *Ambidexter panamensis* (p_i value = 121)

- **Station SW22:** BRI = 56; responsible taxa were the cnidarian family Edwardsiidae, the mussel *M. senhousia*, two polychaetes (*N. acuminata* complex and *L. pugettensis*), and the decapod *P. tuberculata.*

The patterns described above for the six stations classified as Response Level 3 showed that similar taxa were responsible for the elevated p_i values in most cases. The responsible taxa at all six stations were Edwardsiidae, *M. senhousia*, and *L. pugettensis*.

Several of the species that strongly influence BRI values at the shipyards are known to prefer shallow water, muddy bottoms, or quiescent areas, such as are found near shore and in the shipyards. These species include the amphipod *Mayerella banksia*, which is found primarily in intertidal to sublittoral habitats (Laubitz 1970); the mussel *Musculista senhousia*, which is found primarily in protected areas (Crooks 1996); and the polychaetes *Leitoscoloplos pugettensis* and *Capitalla capitata*, which are found primarily in muddy sediment (Hartman 1969). In contrast, the amphipod *Ampelisca cristata*, which has the second lowest pollution tolerance score for southern bays (–105), is likely to be a stray from the open ocean coastal habitat that it prefers (Barnard and Reish 1959) and was found only at Stations 2441 and 2433 in the shipyard study.

At the NASSCO site, the BRI values at the various stations generally exhibited a gradient of increasing magnitude from the outer to the inner parts of the site (Figure 8-23). Values at the five outermost stations (NA01, NA03, NA05, NA12, and NA19) ranged from 43 to 47, whereas values at six of the seven innermost stations (NA06, NA09, NA15, NA17, NA20, and NA22) ranged from 51 to 56, although a value of 45 was found at the seventh innermost station (NA07). Values at the three middle stations (NA04, NA11, and NA16) ranged from 46 to 50.

Although a gradient of BRI values was present throughout most of the Southwest Marine site, it was reversed in direction at the northern extremity of the site, where the value of 42 at the two innermost stations (SW04 and SW08) was considerably lower than the values of 50–54 found at the three middle to outer stations (SW02, SW03, and SW09). In the remainder of the site, the values at the five outer stations (SW11, SW15, SW18, SW25, and SW27) were relatively low (38–43), whereas the values at the five innermost stations (SW13, SW17, SW21, SW22, SW23) were higher (44–56).

Comparisons with Major Community Characteristics—To evaluate how the results of the BRI analysis related to the major benthic community metrics used for the detailed data evaluation described previously, BRI values at the shipyard stations were compared with total taxa richness, total abundance, and species diversity (Shannon–Wiener Diversity Index [H′]). In making the comparisons with the BRI values, emphasis was placed on evaluating the degree to which stations with higher BRI values fell outside the reference envelope determined for each of the three benthic metrics. The reference envelopes were defined by the ranges of mean values of the three benthic metrics found at the four reference stations (Table 8-11). Long and Wilson (1997) concluded that use of a reference envelope was an effective method of evaluating benthic community data as part of the sediment quality triad approach.

Comparisons with BRI values were also made with results of the statistical analyses of benthic metrics, in which the mean value of each benthic metric at each shipyard station was compared statistically with the mean value of the pooled data from the four reference stations. Statistical evaluations are useful because they provide an objective means of determining differences from reference conditions, and they account for the variability inherent in benthic data sets.

The results of the comparisons between the BRI values and the three benthic metrics (abundance, richness, and diversity) are presented in Figures 8-24 through 8-26. In all three comparisons, significant negative correlations ($p \leq 0.05$) were found between the benthic metrics and the BRI values. Although the benthic metrics generally declined with increasing BRI values, numerous stations at higher BRI values remained within the reference envelopes, fell only marginally outside of them, or did not differ significantly ($p>0.05$) from mean reference conditions. The specific patterns found for each of the three benthic metrics are as follows:

- **Total Abundance:** The reference envelope ranged from 440 to 987 individuals per sample, and the mean of the pooled reference data was 643 individuals per sample (Table 8-11). The station-specific total abundance values did not fall below the lower bound of the reference envelope until a BRI value of 50 was reached, and the presence of significant differences ($p\leq0.05$) with the mean value for the pooled reference stations exhibited the same threshold. However, approximately a third of the 13 stations (i.e., 4 stations) with BRI values of 50 or greater had values of mean total abundance that fell within the reference envelope, and over half of the 13 stations (i.e., 8 stations) had values that did not differ significantly ($p>0.05$) from the mean reference value.

- **Taxa Richness:** The reference envelope ranged from 34.8 to 47.8 taxa, and the mean of the pooled reference data was 40.2 taxa (Table 8-11). The station-specific taxa richness values did not fall below the lower bound of the reference envelope in relatively large numbers until a BRI value of 46 was reached, and the presence of significant differences ($p\leq0.05$) with mean taxa richness of the pooled reference stations exhibited the same threshold. However, approximately a third of the 18 stations (i.e., 6 stations) with BRI values of 46 or greater had values of mean taxa richness that fell within the reference envelope, and over half of the 18 stations (i.e., 10 stations) had values that did not differ significantly ($p>0.05$) from the mean reference value.

- **Species Diversity:** The reference envelope ranged from 2.49 to 2.80, and the mean of the pooled reference data was 2.65 (Table 8-11). The station-specific diversity values did not fall below the lower bound of the reference envelope in relatively large numbers until a BRI value of 45 was reached. However, significant differences ($p\leq0.05$) with the mean species diversity for

> the pooled reference data were not found until a BRI value of 51 was reached, with the exception of the unusually low value of diversity found for Station SW04. Over half of the 20 stations (i.e., 11 stations) with BRI values of 45 or greater had values of mean species diversity that fell within the reference envelope, and over three-quarters of the 20 stations (i.e., 16 stations) had values that did not differ significantly (p>0.05) from the mean reference value.

The results of the comparisons of BRI values with major benthic community metrics suggest that the community metrics tended to decline with higher BRI values. However, based on use of a reference envelope and statistical comparisons with mean reference values, it appears that meaningful departures from reference conditions did not occur until BRI values generally exceeded 45−50. This suggests that the benthic response thresholds developed by Smith et al. (2003) are too low; they imply that communities with BRI values less than 45−50 are altered when, in fact, their major characteristics are similar to reference conditions. Even when BRI values exceeded 45−50, one-third to one-half of the affected stations had benthic metrics that fell within the reference envelopes, and one-half to three-quarters of those stations had metrics that did not differ significantly (p>0.05) from the mean reference values.

The results of the detailed analyses of the benthic community described previously were used to characterize the extent of alterations of benthic communities at each shipyard station; alterations were characterized as absent (none), minor, moderate, or major. Figure 8-27 shows the relationship between these comprehensive assessments and the BRI scores. Only when major alterations of the benthic community are observed does the distribution of BRI scores become skewed to higher values. However, even so, the range of BRI scores for major alterations is completely within the range of scores for unaltered communities. Thus, there is no BRI threshold that can be used to unambiguously distinguish major differences in benthic communities from unaltered communities.

Because p_i scores for individual taxa were assigned, in part, on the basis of observed toxicity to amphipods (Smith et al. 2003), computed BRI scores for the shipyard stations were also reviewed with respect to observed amphipod toxicity. The results (Figure 8-28) show that the BRI scores for the San Diego shipyard stations are not predictive of amphipod toxicity. This is another indication that biological effects in the data used to develop the pollution tolerance scores are not representative of conditions at the shipyards, and that BRI scores therefore do not accurately characterize shipyard conditions.

The inappropriateness of the definitions of the response levels is shown by review of the shipyard stations that are classified as Response Level 3. This level is defined by Smith et al. (2003) as having a 50 percent biodiversity loss. Total abundance, taxa richness, and Shannon-Wiener diversity for each of the six shipyard stations categorized as Level 3 are shown in Table 8-15. For two of these stations (SW09 and NA06), all three of the benthic metrics are within the reference area range. For the other four stations, two or three of the benthic metrics are outside the reference area range, but none of them are so far outside as to represent a 50 percent biodiversity loss. Overall, stations that are categorized as “Level 3” do not exhibit impairment as great as is implied by the Level 3 definition of Smith et al (2003).

The results of these evaluations of BRI scores relative to more detailed interpretations of biological effects suggest that even if the benthic thresholds were adjusted upwards, their accuracy in predicting meaningful alterations of benthic communities would be very low.

8.1.4.2 Summary of BRI Applicability

BRI values were computed for the San Diego Bay shipyard data set following resolution of some taxonomic issues. The application of the BRI approach to the benthic macroinvertebrate data collected at the NASSCO and Southwest Marine shipyards showed that a small majority of the species found at reference and shipyard stations had p_i values available for the southern bays.

In comparisons of shipyard stations to reference stations, there were no strong patterns of increasing percentages of species or individuals in higher benthic response levels at the shipyard sites, compared to the reference stations. The patterns observed for numbers of species at the shipyard sites were very similar to the pattern found for the reference stations. These results suggest that conditions at the shipyard sites are not substantially different from conditions at the reference stations.

On a station-specific basis, most of the shipyard stations exhibited only small to moderate differences from the predicted reference conditions, with all but six stations classified as Response Levels 1 or 2. No stations were classified as Response Level 4 (i.e., the category for the most altered communities). Although six stations were classified as Response Level 3, their BRI values (i.e., 54–56) only slightly exceeded the lower bound of that level and were well within the lower part of the BRI range for that classification (i.e., >53 but ≤73).

From a spatial perspective, the BRI values exhibited a general trend of increasing magnitude from the outer to the inner parts of the shipyard sites, although this pattern was reversed at the northern boundary of the Southwest Marine site. BRI values for the reference stations also generally increased with distance from the mouth of the bay. These variations may be attributable, in part, to variations of the benthic community in response to differences in physical conditions such as depth, bottom type, temperature, salinity, and tidal currents.

Comparisons of BRI values with major community metrics (i.e., taxa richness, total abundance, and species diversity) showed that the major community metrics tended to decline with higher BRI values. However, based on use of reference envelopes and statistical comparisons with mean reference values, meaningful departures from reference conditions did not occur until BRI values generally exceeded 45–50. Comparison of the BRI values to the results of a comprehensive analysis of benthic macroinvertebrate data, and to amphipod toxicity data, fails to show any distinct relationship between BRI scores and biological effects. This indicates that the benthic response thresholds developed by Smith et al. (2003) were too low; the thresholds imply that communities with BRI values less than 45–50 are altered when, in fact, their major characteristics are similar to reference conditions. Even when BRI values exceeded 45–50, one-third to one-half of the affected stations had benthic metrics that fell within the reference envelope, and one-half to three-quarters of those stations had metrics that did not differ significantly (p>0.05) from the mean reference values. These results show that even if the response level thresholds were adjusted upwards, their accuracy in predicting meaningful alterations of benthic communities would be very low.

A critical evaluation of the BRI approach (Appendix M) shows that it is affected by numerous uncertainties and defects, including:

- Subjective assignment of benthic thresholds.
- Inconsistencies in species-specific p_i values among different aquatic habitats.
- Lack of p_i values for half the species in the development data set (and approximately 40–50 percent of the species in the shipyard data set).
- Lack of consideration for study-specific reference conditions.
- Lack of sensitivity to some kinds of highly altered benthic communities.
- Absence of quantitative estimates of uncertainties in pollution tolerance scores, which could be used to determine uncertainties in BRI values.
- An assumption that any departures from reference conditions result in adverse affects to benthic communities when, in fact, it is possible that species can be replaced along an environmental continuum without measurably affecting community function.
- Absence of any independent peer review or validation of the derivation method, calculation algorithm, and pollution tolerance scores used for the southern California bays BRI. To the extent that the evaluation presented here can be considered a practical validation exercise, it indicates that the current BRI approach is not necessarily an accurate tool for distinguishing different levels of biological effect.

Given the uncertainties inherent in the BRI approach and its arbitrary and unvalidated classification of response levels, it cannot be considered to be a superior method to assess the status of benthic communities at the shipyard sites or reference stations, particularly in comparison to the detailed analysis of species distributions that was conducted.

8.1.5 Summary of Benthic Macroinvertebrate Community Conditions

Stage III benthic communities are found throughout the shipyards, as shown by SPI photographs of organisms in place in the sediment. Bioturbation of sediments is occurring at all shipyard stations at an intensity comparable to that in the reference areas, as indicated by similarity of the apparent RPD depths. Although surface-dwelling species representative of Stage I communities are found at numerous stations, the disturbance that they signify generally has not prevented the establishment of mature (Stage III) benthic macroinvertebrate communities.

Evaluations of the taxonomic composition of benthic communities using multiple community metrics and using multivariate analysis of the abundances of individual species produced similar results. Most of the shipyard stations could be placed into one of two groups: one of these contained 12 stations that generally exhibited major or moderate alterations of benthic metrics relative to reference conditions, and the other contained 13 stations that generally exhibited no alterations or only minor alterations of benthic metrics. The remaining five shipyard stations were included in three small groups from relatively unique habitats: one consisting of only the single station with major benthic alterations located near the mouth of Chollas Creek, one consisting of two adjacent stations with no benthic alterations located in a dry dock area at Southwest Marine, and one consisting of two adjacent stations with minor or no benthic alterations located in a shallow protected area at Southwest Marine.

Numerous benthic organisms and taxa are present at most shipyard stations, including those at which major alterations of benthic metrics were found. For example, total abundances of benthic communities at four of the six stations with major benthic alterations (i.e., Stations NA04, NA15, SW21, and SW23) were 44–49 percent of the mean reference value of 6,400 organisms per square meter (Table 8-16). Although significantly different from reference conditions, those percentages nevertheless represent relatively large numbers of benthic organisms (i.e., approximately 2,900–3,100 benthic organisms per square meter). In addition, the numbers of benthic taxa at the four stations were 60–66 percent of the mean reference value of 40 taxa. However, those percentages represent relatively large numbers of taxa (i.e., 24–27 taxa). Therefore, although both total abundance and taxa richness at the four stations were significantly lower than mean reference values, the stations still support large numbers of benthic organisms and taxa. For stations with minor or moderate differences from reference conditions, abundance and richness of organisms at the shipyards are even higher.

In summary, substantial alterations of benthic macroinvertebrate communities are present at only a single station sampled in the vicinity of the shipyards: Station NA22 near the mouth of Chollas Creek. Of the remaining 29 stations sampled in the shipyards, no significant ($p>0.05$) alteration of any kind of benthic metric was found at 16 stations. Although significant ($p\leq0.05$) alterations of one or more benthic metrics were found at the remaining 13 stations, all of the benthic communities at those stations were represented by relatively large numbers of benthic organisms and taxa. The presence of mature benthic macroinvertebrate fauna, with substantial numbers of individuals and taxa, indicates that functional benthic communities are present throughout almost the entirety of the shipyard leaseholds. If substantial alterations of benthic communities were occurring in response to shipyard chemicals, those alterations would be manifested as communities with very few numbers of individuals and taxa, most or all of which would be relatively tolerant of chemical toxicity. Such conditions were not observed at any of the shipyard stations, with the exception of Station NA22.

8.2 Fish Histopathology

Fish histopathology was evaluated as one indicator of potential exposure of fishes to chemical contaminants near the NASSCO and Southwest Marine shipyards. Fish histopathology has been used as an indicator of contaminant exposure in fishes in numerous studies throughout the United States (e.g., Murchelano and Wolke 1985; Malins et al. 1987; Myers et al. 1987, 1994; Stehr et al. 1997). In addition, fish histopathology is a key component of the ocean monitoring programs for municipal sewage outfalls on the continental shelf off Southern California (OCSD 2003). The target species for most fish histopathology studies are usually demersal fishes that prey primarily on benthic macroinvertebrates, because these fishes have a higher potential for exposure to contaminants in sediments than do pelagic species or species that prey primarily on other fishes. Most fish histopathology studies focus on the liver because:

- Many contaminants become highly concentrated in the liver
- The liver is the site of metabolism for some contaminants (e.g., chlorinated pesticides, PCBs, PAHs), and the metabolic products of some contaminants are known to have mutagenic and carcinogenic properties
- Certain liver lesions have been linked to contaminant exposure and are therefore useful indicators of such exposure.

Although the liver is the primary focus of most studies of fish histopathology in relation to sediment contamination, histopathological evaluations of other organs can also provide useful information for assessing potential exposure to chemical contaminants.

In the present study, the spotted sand bass (*Paralabrax maculatofasciatus*) was selected as the target species because it preys primarily on benthic macroinvertebrates, exhibits limited spatial movements, and is abundant in numerous kinds of habitats within San Diego Bay (Allen et al. 1995). The limited spatial movements that have been observed for this species are important because this behavior enhances the probability that individuals captured in a histopathology study have resided in a particular area for a sufficient length of time to be representative of chemical exposures in that area. In a tagging study conducted in southern California, it was found that most tagged spotted sand bass were recaptured within 200 yards of their release point (Allen 1996). The relatively high abundance of spotted sand bass in San Diego Bay was important because it is essential to obtain adequate sample sizes within each kind of habitat evaluated, if meaningful statistical comparisons among habitats are to be made. The target sample size in the present study was 50 fish per habitat type.

8.2.1 Field and Laboratory Methods

A total of 253 spotted sand bass were sampled on September 25–29, 2002, in five locations within San Diego Bay:

- Inside the NASSCO shipyard site (50 fish)
- Immediately outside of the NASSCO shipyard site (50 fish)
- Inside the Southwest Marine shipyard site (51 fish)
- Immediately outside of the Southwest Marine shipyard site (50 fish)
- Within a reference area near Station 2240 (52 fish).

Fishes were collected using nets and by hook and line.

Detailed descriptions of the field and laboratory procedures used to evaluate fishes and individual organs are presented in Marty (2003). Briefly, each fish was held in fresh seawater no longer that 8 hours before processing aboard the sampling vessel. Prior to necropsy, each fish was weighed, measured (fork length to nearest millimeter), and examined macroscopically for grossly visible external abnormalities. The liver, gonads, kidney, and gills of each fish were then removed, examined macroscopically, and preserved in 10 percent neutral buffered formalin. The right otolith of each fish was also removed and stored for later age determination.

In the laboratory, tissues were processed and examined using standardized procedures. Lesions were identified, and the most common kinds of tissue alterations were scored semiquantitatively on a scale of 0 (none), 1 (mild), 2 (moderate), or 3 (severe). To ensure consistency of lesion identifications and diagnostic criteria, all determinations were made by a single experienced pathologist: Dr. Gary Marty of Fish Pathology Services.

8.2.2 Results and Discussion

Detailed descriptions of the various histopathological lesions found during the study are presented in Marty (2003), along with a set of photomicrographs of key lesions and conditions. The prevalences of all lesions evaluated in this study are summarized in Appendix N. The numbers of lesions or other conditions evaluated in livers, kidneys, gonads, and gills were 22, 14, 10, and 13, respectively. In addition, 11 different grossly visible external lesions were evaluated macroscopically. Of the total of 70 lesions and other conditions evaluated in this study, 65 were found in the 253 spotted sand bass that were evaluated histopathologically. Some kind of lesion was found in the livers, kidneys, gonads, gills, and bodies of fish collected from all five sampling locations.

8.2.2.1 Statistical Comparisons with Reference Conditions

The various histopathological lesions found both inside and outside the shipyards during this study were compared with reference conditions using nonparametric ANOVA, based upon the severity score for each lesion in each fish (i.e., scores of 0, 1, 2, 3, and 4). When the ANOVA results were significant, two-tailed *a posteriori* comparisons were made between the results for each shipyard location and the results for the reference area.

Lesions Elevated at Shipyard Locations—Of the total of 70 lesions evaluated, only 3 exhibited significant ($p \leq 0.05$) elevations at one or more shipyard locations relative to reference conditions (Table 8-17). However, a fourth lesion (i.e., abundant hemosiderin in liver) was considered important by the pathologist (Marty 2003) and was nearly significant (i.e., $p = 0.07$). That lesion was therefore conservatively considered significant for the purposes of this study. The four lesions included the following:

- **Liver:** Abundant lipofuscin—greater inside both shipyards than in the reference area
- **Liver:** Abundant hemosiderin—greater outside the NASSCO shipyard than in the reference area

- **Kidney:** Nephritis—greater inside the NASSCO shipyard than in the reference area
- **Gill:** Shiny gill foci—greater inside the Southwest Marine shipyard than in the reference area.

As described above, three of the four lesions were found to be elevated inside one or both of the shipyards, whereas one lesion (i.e., abundant hemosiderin) was found to be elevated immediately outside the NASSCO shipyard.

The results presented in Table 8-17 show that for the fishes in which lesions were found, the severity of the lesions in most individuals was considered mild (i.e., severity score = 1), and relatively few individuals had lesions that were considered severe (i.e., severity score = 3). For abundant hemosiderin in the liver, no fish from either the four shipyard locations or the reference area was considered moderately or severely affected. For nephritis in the kidney, only 5 of 201 fish collected in or near the shipyards were considered moderately (4 fish) or severely affected (1 fish).

Of the three microscopic lesions found to be elevated in the shipyard locations relative to the reference area, only abundant lipofuscin was found at moderate to severe levels in modest numbers of individuals. For example, 16 of the 101 fish collected inside the two shipyards were moderately or severely affected. However, only 1 of the 100 individuals captured outside the shipyards was moderately or severely affected.

For shiny gill foci that were identified macroscopically, nearly all of the 253 fish collected during the study were affected to some degree (i.e., 237 fish or 94 percent of the total sample), and of the affected individuals only 3 were severely affected. However, as noted by Marty (2003), the cause of this lesion is unknown.

In summary, only 4 of the 70 lesions evaluated in this study were considered significantly elevated compared to the reference area. In addition, the severity of the lesions found in most individuals was considered mild, and relatively few individuals had lesions that were considered severe.

Lesions Elevated at the Reference Area—Six of the 70 lesions evaluated in this study were significantly elevated ($p \leq 0.05$) in the reference area relative to one or more of the four shipyard sites. Those conditions included the following:

- **Kidney:** Renal tubular regeneration—greater in the reference area than outside the NASSCO shipyard
- **Gonads:** Atresia of yolked follicles—greater in the reference area than inside the Southwest Marine shipyard
- **Fins:** Caudal fin reddening—greater in the reference area than outside the Southwest Marine shipyard
- **Fins:** Caudal fin fraying—greater in the reference area than inside or outside the NASSCO shipyard
- **Body cavity:** Diffuse opaque epicardium—greater in the reference area than inside the two shipyards
- **Body cavity:** Mean number of Anisakis parasites—greater in the reference area than inside the two shipyards.

With respect to lesion severity, perhaps the most striking feature of the results presented in Table 8-18 is the relatively large number of fish in the reference area (i.e., 67 percent) that were severely affected by atresia of yolked follicles in the gonads. That percentage is more than twice the percentage of severely affected individuals at any of the four shipyard locations. Another striking feature is the relatively large number of Anisakis parasites found in fish from the reference area.

In summary, the results presented in Table 8-18 illustrate the fact that statistically distinct and unusual histopathological conditions in spotted sand bass were not restricted to the shipyard locations. The results also indicate that the study design included sufficient statistical power to detect relative minor differences in the prevalences of the various lesions.

8.2.2.2 Significance of Lesions

Although 65 kinds of histopathological lesions and other conditions were found in the spotted sand bass captured at the shipyard locations and in the reference area, few of them have been documented to be serious pathological conditions that are potentially related to chemical exposure, and would potentially compromise the health of the affected fish. For the liver in particular, almost none of the serious lesions found at other contaminated sites in the United States were found in the present study. For example, Table 8-19 lists the key liver lesions that have been linked to contaminant exposure and potential carcinogenesis in fishes in the United States (Myers et al. 1994; Stehr et al. 1997; OCSD 2003). They include lesions related to the early stages of contaminant exposure (i.e., hydropic vacuolation) and/or carcinogenesis (i.e., specific degenerative conditions), lesions that appear to be precursors of neoplasia (i.e., foci of cellular alteration), and neoplasms (i.e., tumors). All of the fish collected in the present study were evaluated for the lesions listed in Table 8-19, and only two individuals were found to be affected by one of the lesions (i.e., two fish from the reference area were found to have some kind of neoplasm). A tumor was also found in the kidney of one fish from inside the Southwest Marine shipyard and in the internal body wall of one fish captured outside the NASSCO shipyard. Although tumors were found in two fishes during this study, Marty (2003) concluded that tumor development was not related to the shipyard sites.

Instead of serious pathological lesions, most of the conditions identified during the present study may be related to factors other than chemical toxicity or may have an uncertain effect on the health of the fishes. Marty (2003) considered abundant lipofuscin in the livers of the spotted sand bass to be the most consistent and severe lesion found during the study. As described above, this condition was elevated inside the two shipyard sites relative to the reference area. Only two fish at the reference area were found to have abundant lipofuscin in the liver, whereas 13 fish inside each of the two shipyards (i.e., total = 26) were found to have abundant lipofuscin. Lipofuscin is the term given to granular deposits derived from the lipid component of membranous intracellular organelles. It tends to accumulate in hepatocellular lysosomes following degradation of cell membranes. Although lipofuscin has been shown to increase in fish hepatocytes exposed to a variety of chemicals (e.g., Vogelbein 1993; Biagianti-Risbourg and Bastide 1995; Au et al. 1999; Au and Wu 2001; Fahraeus-Van Ree and Spurrell 2000; Nowak and Kingsford 2003), other potential causes of increased amounts of lipofuscin include starvation, nutritional imbalances, disease, and normal aging.

In the present study, livers with abundant lipofuscin tended to weigh more than livers without this condition, suggesting a relationship between the two conditions. For example, only 5 livers of the 253 evaluated had weights greater than 10 g, and all had abundant lipofuscin. It therefore is possible that the abundant lipofuscin found in the spotted sand bass from inside the shipyard sites was due to exposure to chemicals. However, the significance of this condition with respect to predicting serious pathological conditions in the affected organisms is unknown.

Another prevalent condition found during this study was abundant hemosiderin in hepatocytes, which was found to be elevated outside the NASSCO shipyard. Hemosiderin is a breakdown product of hemoglobin (i.e., the red oxygen-carrying pigment of blood), and it is common in hepatocyte lysosomes. Although hemosiderin had been shown to increase in fish hepatocytes following exposure to chemicals (Thiyagarajah et al. 1998; Khan et al. 1994), it may also be related to the other factors identified above for lipofuscin. In the present study, the fact that abundant hemosiderin in fishes captured within the two shipyards was not elevated relative to reference conditions indicates that this lesion was not the result of exposure to shipyard chemicals.

To evaluate whether fish age may have had an effect on the prevalence of abundant lipofuscin or abundant hemosiderin in the spotted sand bass captured in the present study, those prevalences were compared with the fish age (Figure 8-29). The results of these comparisons show that abundant lipofuscin tended to increase with increasing age of male fish. By contrast, there were no apparent relationships with fish age for abundant lipofuscin in female fish, or for abundant hemosiderin in male or female fish. Although, relatively low sample sizes may have biased the results for some of the age classes evaluated in Figure 8-29, the results found for males suggest that abundance of lipofuscin may be related in some manner to fish age.

8.2.2.3 Evaluations of Fish Growth and Condition

To evaluate whether the spotted sand bass collected in the present study exhibited reductions in size in relation to the various sampling locations or the presence or absence of the key liver lesions discussed above (i.e., abundant lipofuscin and abundant hemosiderin), indices of fish growth and condition (i.e., fatness) were calculated and compared in relation to the factors described above. The index for growth was the length at age for each fish and the index of condition was the weight/length ratio for each individual. Most fish collected at the shipyard sites and reference area (i.e., 56 percent of the 253 fish collected) were 4–5 years old. Twenty percent of the fish ranged from 1–3 years old and the remaining 24 percent of individuals ranged from 6 to 13 years old. The age distribution of fish from the reference area was not significantly different ($p \leq 0.05$) from the age distribution at any of the four shipyard sites.

Comparisons of the growth and condition indices were both stratified by the sex of the fish because both indices can vary with that characteristic. Although normalizing the growth data by age and stratifying both the growth and condition data by sex removed potential confounding variables and provided more meaningful comparisons, it sometimes resulted in sample sizes that were considered too low to be evaluated statistically. In those cases, comparisons were made on a descriptive basis.

Spatial Comparisons—Comparisons of mean length at age of spotted sand bass among sampling stations are presented in Figure 8-30. Samples were pooled for the 1–3 and 6–13 year old age intervals to increase sample sizes. The results of the comparisons show that, in most cases, values of mean length at age at the four shipyard sites were similar to the value found in the reference area. Although differences were found for some age intervals, they showed no consistent pattern of mean length at age at the shipyard sites being either greater than or less than the reference value.

Comparisons of mean weight/length ratios of spotted sand bass among sampling stations are presented in Figure 8-31. None of the mean weight/length ratios found at the four shipyard sites were significantly different ($p \leq 0.05$) from the value found for the reference area.

In summary, the results of the comparisons of mean length at age and mean weight/length ratios of spotted sea bass among sampling stations indicated that that neither the growth nor the condition of the fishes was affected by proximity to the shipyards.

Comparisons Based on Liver Lesions—For these comparisons, all fish were pooled across the five sampling areas (i.e., four shipyard sites and the reference area), to maximize sample sizes. Fishes were then stratified by those in which each liver lesion (i.e., abundant lipofuscin or abundant hemosiderin) was either present (i.e., at any level of severity) or absent.

Comparisons of mean length at age of spotted sand bass in relation to the presence or absence of each of the two liver lesions are presented in Figure 8-32. The results of the comparisons show that, in most cases, values of mean length at age were greater (i.e., individuals were larger) for fish in which each lesion was present compared to fish in which the lesions were absent. These results indicate that an adverse effect on fish condition was not associated with the presence of either kind of liver lesion.

Comparisons of mean weight/length ratios of spotted sand bass in relation to the presence or absence of each of the two liver lesions are presented in Figure 8-33. As was found for mean length at age, mean weight/length ratios in most cases were greater for fish in which each lesion was present compared to fish in which the lesions were absent. These results also indicate that an adverse effect on fish condition was not associated with the presence of either kind of liver lesion.

In summary, the results of the comparisons of mean length at age and mean weight/length ratios of spotted sea bass between individuals with and without abundant lipofuscin or abundant hemosiderin indicated that that neither the growth nor the condition of the fishes was affected by the presence of the two kinds of liver lesions.

8.2.3 Conclusions

Although 65 kinds of lesions and other conditions were found in the 253 spotted sand bass collected during the present study, only 4 lesion categories were considered significantly elevated at one or more of the four shipyard locations relative to the reference area. In addition, the severity of the lesions found in most individuals was considered mild, and relatively few individuals had lesions that were considered severe. A greater number of lesions (i.e., 6) were significantly elevated in the reference area compared to the shipyard sites, documenting that pathological conditions occur in parts of San Diego Bay away from the shipyards.

Few of the lesions identified during this study have been documented to be serious pathological conditions that are potentially related to chemical exposure and would potentially compromise the health of the affected fish. Of the serious liver lesions found at other contaminated sites in the United States (e.g., hydropic vacuolation, specific degenerative conditions, foci of cellular alteration, neoplasms), only neoplasms were found in fish evaluated in the present study, and they were found in only two individuals from the reference area. Tumors were also found in the kidney and internal body wall of two fish collected near the shipyards. Furthermore, it was concluded that this tumor development was not related to the shipyard sites.

Abundant lipofuscin and abundant hemosiderin were two prevalent conditions found to be elevated in the livers of the spotted sand bass collected from one or more of the four shipyard locations, and both of these conditions have been related to chemical exposures in previous studies, including exposure to pesticides (Nowak and Kingsford 2003). However, other factors may cause their occurrence; for example, abundant lipofuscin exhibited a potential relationship to fish age in the present study. The relationship of these conditions to individual fish health and to fish population dynamics is unknown.

Comparisons of indices of growth and condition of spotted sand bass among sampling locations and between fish in which abundant lipofuscin and abundant hemosiderin were either present or absent showed that neither growth nor condition of the fish was affected by proximity to the shipyards or by the presence of the two liver lesions.

The results of this study indicate that the health of spotted sand bass is not adversely affected by proximity to the shipyards.

8.3 Fish Bile

PAH compounds are ordinarily difficult to detect in fish tissue because of rapid breakdown of these compounds in the organism. Metabolism of PAH occurs in the livers of fish, and this process produces a variety of polar organic compounds that are excreted in bile. These breakdown products can be measured in fish bile, and can therefore serve as an indication of the fish's exposure to PAH compounds. Elevated concentrations of PAH breakdown products are found in bile of fish from areas of PAH-contaminated sediment, even when only trace concentrations of PAH are found in fish flesh (Krahn et al. 1984).

During Phase 2, bile was collected from all spotted sand bass captured for histopathological examination. Bile samples were composited to produce up to 10 samples from each sampling location (inside and outside each shipyard leasehold, and at Reference Station 2240). The bile samples were analyzed for fluorescent aromatic compounds and total proteins. Three groups of fluorescent aromatic compounds were measured, corresponding to breakdown products from the metabolization of naphthalene, phenanthrene, and benzo[a]pyrene. Total protein was measured to allow the concentrations of PAH breakdown products to be adjusted for differences in the nutritional state of the fish. Compositing information and analytical results are contained in Appendix E, and the data quality assurance report is contained in Appendix F.

Results of the bile analyses are summarized in Figures 8-34 through 8-36. Concentrations of bile breakdown products in fish from the shipyards are not significantly greater ($p<0.05$) than concentrations at the reference area. Concentrations in fish from within the shipyard leaseholds are generally less than concentrations in fish from outside the leaseholds. These results indicate that fish at the shipyards are no more greatly exposed to PAH than fish at other locations in San Diego Bay, and consequently that there are no potential adverse effects on fish from PAH at the shipyards.

9 Assessment of Potential Effects on Aquatic Life

Protection of aquatic life is one of the beneficial uses to be supported at the shipyards, and numerous different types of analyses were conducted during both Phase 1 and Phase 2 of the shipyard investigation to allow an assessment of potential adverse impacts on aquatic life. Direct measurements of biological effects or conditions at the shipyards consisted of:

- Three types of sediment toxicity tests conducted as part of the triad study
- Benthic macroinvertebrate community analyses conducted as part of the triad study
- Fish histopathology analyses
- Analyses of PAH breakdown products in fish bile.

The triad data provide the only direct point-by-point association between measured biological effects and sediment chemistry at the site. Fish histopathology and bile analyses provide more general indications of exposure or potential effects. The triad data and fish sampling data are evaluated in this section of the document with respect to sediment chemistry, and as a means of deriving cleanup levels for chemicals that are based on protection of aquatic life.

9.1 Biological Effects *vs.* Sediment Chemistry

The results of the four station-specific measures of potential biological effects—the toxicity tests and benthic macroinvertebrate community analysis—can be examined relative to the chemical concentrations at those stations. This comparison can identify any chemical, or set of chemicals, that are strongly associated with adverse biological effects and that are therefore potential causes of those effects. Statistical correlations between chemicals and biological effects can identify chemicals that are potential causes of adverse effects. The magnitude and statistical significance of such correlation coefficients can indicate the relative impact of different chemicals, and the square of the correlation coefficient (the coefficient of determination, or R-squared) indicates what proportion of the variation in biological responses may be attributable to each chemical. However, it should be noted that such statistical analyses only show potential causal relationships and do not, by themselves, prove that a substance causes an effect.

Correlations between measures of biological effects (amphipod survival, echinoderm fertilization, bivalve development, total benthic taxa abundance, and total benthic species) and sediment chemicals (metals, butyltins, LPAH, HPAH, total PCB homologs, PCTs, DRO, RRO, percent fines, and percent clay) are shown in Table 9-1. With a few exceptions, there are no statistically significant correlations between chemical concentrations and biological effects. The absence of a relationship between variables can also be seen in a scatter plot such as Figure 9-1, which shows amphipod survival in relation to sediment copper concentrations: there is no decrease in survival at increasing copper concentrations. These results demonstrate that there is no correspondence between concentrations of most of the putative shipyard chemicals and any of the measures of biological effects. Only for selenium, DRO, and RRO are the correlation coefficients significantly different from zero. However, even the statistically significant correlation coefficients are low. For example, Figure 9-2 shows a scatter plot of benthic macroinvertebrate richness against RRO—the strongest correlation of any biological effect with any shipyard chemical. The correlation in this case is primarily driven by relatively large richness values in three replicates of two samples, and species richness at the highest concentrations of RRO is equivalent to that at the lowest concentrations of RRO. The square of the correlation coefficient (the coefficient of determination, or R-squared) represents the amount of variation in the measures of effects that are associated with variation in chemical concentrations. These values range from 2 percent to 8 percent, indicating that even the statistically significant relationships are very weak. In other words, over 92 percent of the variability in species values is associated with factors other than RRO.

Biological effects are more consistently, and generally more strongly, related to the fraction of fine particles in the sediment than they are to any of the chemicals measured. Although chemical concentrations are usually higher on finer particles, the stronger association of biological effects with fine particles than with any of the measured chemicals suggests that the fine particles themselves—or some other unmeasured chemical that is attached to them—have the greatest effect on biological responses at the shipyards.

The absence of any consistent relationships between most of the putative shipyard chemicals and any of the measures of adverse biological effects may be due to one or more of the following causes:

- The magnitude of the biological effects may be too small to allow significant effects to be identified
- The tests of biological effects may be inaccurate
- Complex variations may occur among concentrations of causative chemicals
- Chemicals other than the putative shipyard chemicals may be producing the observed biological effects
- Effects other than chemical toxicity may be producing the observed biological effects.

These possibilities are each discussed in the following sections.

9.1.1 Magnitude of Biological Effects May Be too Small to Be Significant

The echinoderm fertilization test identified no adverse effects at any shipyard station, so the magnitude of effects for this test is very likely to be too small to observe any relationship between effects and sediment chemistry. The amphipod survival tests identified several adverse effects, but even those effects that were statistically significant were relatively minor (survival above 70 percent in every case). Therefore, it is possible that relationships between chemistry and adverse effects were also not observed for the amphipod test because of the relatively small range of responses in the test.

In contrast, larger, significant effects were observed in both the bivalve development test and the benthic macroinvertebrate community assessment. Therefore, in the absence of other limitations to the tests, it would be expected that relationships between adverse effects and causative chemicals would be identifiable for these tests.

9.1.2 Tests of Biological Effects May Be Inaccurate

Inaccurate test results may explain the lack of correspondence with chemical concentrations for the bivalve larval development test. There are three reasons for this:

1. Larval tests as a whole are not always reliable (U.S. EPA 1992b)
2. The bivalve development test used for this study incorporated a non-standard modification (specified by Regional Board staff; RWQCB 2001), and the method has not been fully developed or approved by EPA
3. The results of the test showed widely varying measures of effects among the reference stations evaluated.

The bivalve test may therefore be inaccurate, and this would explain why there is no relationship between the results of this test and chemical concentrations.

There are no issues of accuracy with the other tests—all are standard and widely used methods, and none of them produced inconsistent results among the reference stations used for this study. All of the toxicity tests except the bivalve test were conducted following standard protocols[2] and included both negative and positive controls. Positive controls were included to assess the responsiveness of the test organisms to reference toxicants. Quality control criteria for the negative and positive controls were met for all tests. In addition, a dilution series was conducted at two stations for the amphipod test; this analysis confirmed that the amphipods responded in a dose-dependent manner to changes in the concentrations of chemicals in the sediment. Counts of benthic macroinvertebrates were verified by re-counting to ensure the consistency and quality of these data. Consequently, the likelihood that spurious results were

[2] The bivalve development test included an experimental method for isolating the larvae from the sediment surface, but other aspects of the test were conducted in accordance with standard protocols.

obtained for every one of these independent biological effect assessments is considered to be negligible.

9.1.3 Complex Variations May Occur among Concentrations of Causative Chemicals

The general absence of associations between chemical concentrations and adverse biological effects could potentially be because different chemicals, or combinations of chemicals, cause toxicity in different samples. Thus, for example, chemical A might cause toxicity in one sample, but chemical B cause toxicity in the next sample, and chemical C cause toxicity in the next sample, and so forth. If this type of situation was responsible for the poor relationships between chemistry and toxicity, the concentrations of the chemicals would have to be poorly correlated with one another. Otherwise, for example, chemicals B and C would also be present at high concentrations—and cause toxicity—wherever chemical A is present at high toxicity. However, in fact, the concentrations of all chemicals except selenium are significantly—and in most cases, highly—correlated with one another (Table 9-2). Thus, there is no strong evidence for complex variation among chemical concentrations that can explain the poor association between chemical concentrations and adverse biological effects. The correlation analysis suggests that there might be, at most, two groups of unrelated chemical causes of adverse effects: selenium and everything else. However, selenium was detected in only one-third of the sediment samples—the lowest detection frequency of any of the chemicals in Table 9-2—and the detected concentrations are approximately equal to the quantitation limit. Therefore, the absence of correlations between selenium and other chemicals is likely a consequence of the fact that there are only minor, and possibly meaningless, fluctuations in observed selenium concentrations. Had a lower quantitation limit been obtained, it is possible that correlations of selenium with other chemicals would have been found.

Another possible type of complex relationship that could occur among chemicals is the presence of combined, or synergistic, effects. However, the relationships observed between chemical concentrations and toxicity in this data set do not support a conclusion that synergistic effects are occurring at the shipyard site. At this site, for essentially all chemicals, some stations without any amphipod effects have concentrations higher than stations with effects. For example, chemical concentrations at the six stations showing amphipod toxicity are all generally lower than concentrations at other stations that do not exhibit toxicity (e.g., SW02, SW04, and SW08). Thus, any functions of multiple chemical concentrations that are potentially responsible for toxicity would also be lower at the effect stations than at the no-effect stations, and such functions of multiple chemicals would not explain the observed presence of amphipod effects.

A quantitative analysis of interactions between chemicals has been carried out, and the results confirm that amphipod toxicity cannot be attributed to interactions among chemicals. This analysis consisted of a principal components analysis (PCA) to identify groups of covarying chemicals, followed by multiple linear regression to identify possible interactions between chemicals in relation to amphipod toxicity. PCA was used to reduce the number of terms used in the linear regression, because the number of chemicals and interactions between them is otherwise too large for analysis. The first four principal components accounted for 85 percent of the variability among chemical concentrations; these principal components were represented mainly by PCBs, HPAH, LPAH, and butyltins and some metals, respectively. A single chemical was chosen to represent each of these components for use in the multiple linear regression. The chemicals chosen were the sum of PCB congeners, total HPAH, total LPAH, and TBT. These chemicals and all pairwise interactions between them were included in the multiple linear regression model. The interaction terms in the regression model accounted for possible multiplicative (synergistic) interactions among chemicals. This model had a multiple R-squared value of 0.127, indicating that interactions among chemicals cannot account for any more than 13 percent of the observed variation in amphipod mortality. Complex interactions among chemicals therefore are not an explanation of the observed toxicity.

9.1.4 Other Chemicals May Be Responsible for Biological Effects

Because analysis of the triad data following Phase 1 revealed the lack of correspondence between shipyard chemicals and adverse biological effects, and because of the proximity of a source of pesticides (Chollas Creek), sediment samples from four stations were analyzed for organochlorine and organophosphorous pesticides during Phase 2. The four stations selected are listed in Table 9-3. The data resulting from these analyses is tabulated in Appendix B. Five organochlorine pesticides were detected at one or more stations: alpha-chlordane, gamma-chlordane, 4,4′-DDT, 4,4′-DDE, and 4,4′-DDD. 4,4′-DDT was detected at all four stations, and all four pesticides were detected at NA22 (the closest to Chollas Creek). The small number of samples and the fact that not all of these pesticides were detected in all samples limit the amount of weight that can be placed on these results. However, there are significant negative correlations between these pesticide concentrations and biological effects, specifically, bivalve development, echinoderm fertilization, and benthic macroinvertebrate richness. These correlations are shown in Table 9-4. The strongest correlation ($r = -0.77$) is with the results of the bivalve development test. Although pesticide concentrations were not correlated with total benthic macroinvertebrate abundance, they were negatively correlated with benthic mollusc abundance ($r = -0.74$). Thus, there is clear evidence that pesticides may be responsible for adverse biological effects at the shipyards—particularly, adverse effects to bivalve molluscs.

9.1.5 Non-chemical Conditions May Be Responsible for Biological Effects

Because of ship traffic, dry dock movements, and engine tests within the shipyards, physical disturbance may also account for the general absence of relationships between chemical concentrations and adverse biological effects. This effect is very likely to occur in the southeast portion of the NASSCO site, where engine tests are conducted after vessel construction has been completed. The SPI data provide confirming evidence for this possibility: all of the stations along the shoreline in this area had Stage I benthic communities, which are indicative of recent disturbance. Stations farther from the shore in this area had a combination of Stage I and Stage III communities, indicating that surface sediment may be frequently disturbed (e.g., resuspended and resettled), although the disturbance is not so great as to prevent the establishment of deeper-dwelling organisms characteristic of mature benthic communities. Visible erosion of the shoreline in this area is also evidence of the physical disturbance caused by engine tests.

Physical disturbance may also have an effect at other locations in the shipyards. A mixture of Stage I and Stage III communities is found in over half (52 percent) of the shipyard area, indicating that some physical disturbance may be affecting the surface sediment over a broad area of the shipyards. The signs of mixing of near-surface sediment apparent in the vertical profiles of percent fines data (discussed in the section titled *Grain Size Distribution*) are another indicator that physical disturbance may be affecting sediment in some areas down to a depth of 1–2 ft or more. The presence of physical disturbance is likely to affect the benthic macroinvertebrate community, even in the absence of chemical toxicity. There are 14 stations at which both triad measurements were made and cores taken; for these 14 stations, Figure 9-3 shows the relationship between sediment disturbance (as indicated by percent fines profiles) and alterations of the benthic macroinvertebrate community. Wherever alteration of the benthic community is the only biological effect found, apparent physical disturbance is also found. There are no cases where sediment is apparently undisturbed and only the benthic community was found to be altered. This strong association between apparent disturbance and benthic community alteration indicates that physical disturbance may be responsible for many of the apparent effects on benthic macroinvertebrates, and thus explains the lack of correlation between benthic macroinvertebrate effects and shipyard chemicals.

9.2 Integrated Evaluation of Biological Effects Data

As described in the work plan (Exponent 2001), the multiple indicators of sediment toxicity and benthic macroinvertebrate community structure are to be used for a biological assessment of sediments, using decision matrices to combine the results of the major types of biological tests assessed as part of the triad study. The decision matrix for interpreting the triad measurements of biological effects in terms of potential effects on beneficial uses is shown in Table 9-5. The integrated evaluation of biological effects data, as represented by effects on beneficial uses as in Table 9-5, will be used to evaluate the efficacy of potential cleanup levels. Derivation and evaluation of cleanup levels is described in Section 12.

The data for the multiple toxicity tests must be evaluated in combination to reach an overall assessment of the potential for toxicity. The results of this evaluation are represented by a scoring system, or decision matrix, as shown in Table 9-6. This scoring system is based upon the ecological relevance and relative reliability of the different toxicity tests, specifically:

- A lethal response, as in the amphipod test, is considered to be a stronger indicator of sediment toxicity than a response in either of the sublethal tests
- Pore water exposure (as in the echinoderm fertilization test) is considered to be less relevant for benthic organisms than whole-sediment exposure (U.S. EPA 2000b)
- Larval tests are considered to be of questionable reliability (U.S. EPA 1992b)
- Because the bivalve test used a non-standard method and produced highly variable results even among reference area samples, it is considered to be less reliable than the other two tests.

Therefore, as shown in Table 9-6, if a response is seen in only one of the sublethal tests, the likelihood of sediment toxicity is considered to be lower than if a response is seen in only the amphipod mortality test. When responses are seen in two of the three tests, toxicity is considered to be more likely when one of the two is the amphipod test, and is more likely when the other test is the echinoderm fertilization test rather than the bivalve development test.

The integrated evaluation of the overall toxicity assessment and benthic community conditions (Table 9-5) gives slightly greater weight to the benthic community measure than to the toxicity measure because of greater ecological relevance. The benthic macroinvertebrate data consist of a direct evaluation of the biotic community at the shipyards, whereas interpretation of the toxicity tests is based on an assumption of equivalence between laboratory and field conditions. Because the benthic macroinvertebrate assessment is a direct measure of the community, and the toxicity tests are indirect measures, the benthic data are given slightly more weight.

Using these decision matrices and the results of the biological assessments reported in Sections 6 and 8 produces an assessment of the likelihood of adverse effects on the aquatic life beneficial use. The results of this assessment are summarized in Table 9-7. The spatial distribution of these different categories of potential effect is illustrated in Figure 9-4.

The results of this assessment (Table 9-7) describe only the likelihood of adverse aquatic life effects, and not causative mechanisms or the magnitude of those effects. As noted in the previous section, relationships between potential adverse biological effects and shipyard chemicals are absent or weak, so there is no clear causative link between shipyard chemicals and these effects. In two areas of the NASSCO shipyard, there is good evidence of alternative causes for the observed biological effects. Between NASSCO Piers 4 and 5, the SPI data, the presence of shoreline erosion, the nature of site usage (engine tests), and the absence of any sediment toxicity indicate that physical disturbance is most likely responsible for the moderately altered benthic community that is observed. On the south side of NASSCO Pier 6, the presence of pesticides in the sediment and the correlation of pesticides with adverse biological effects, the nearby source of pesticides and other contaminants from Chollas Creek, and the absence of shipyard construction activities indicate that pesticides from Chollas Creek are most likely responsible for the effects observed. Lower levels of physical disturbance (as suggested by the SPI results) and pesticide concentrations (as suggested by the pesticide analyses and correlations) may also be responsible for possible adverse effects in other parts of the shipyard leaseholds.

9.3 Cleanup Criteria for Protection of Aquatic Life

Cleanup criteria for protection of aquatic life are derived principally from the triad data that directly associate adverse biological effects with sediment chemistry concentrations. Pore water partitioning data, fish histopathology data, and file bile PAH breakdown product data are also all considered in the development of the cleanup criteria.

9.3.1 Selection of Indicator Sediment Chemicals

A subset of the complete suite of chemicals measured in sediment has been selected as indicator chemicals for the purpose of deriving cleanup levels (RWQCB 2001). Indicator chemicals have been selected to represent each of the major classes of sediment pollutants and to include those chemicals with observed relationships to biological responses. Evaluation of these two criteria—representativeness and potential relationships to effects—is carried out in two steps.

The first step in the selection of indicator chemicals is to identify chemicals representative of the major classes of sediment pollutants. The major classes of sediment pollutants measured in this investigation are metals, butyltins, PCBs and PCTs, PAH, and petroleum hydrocarbons. Although several samples were also analyzed for pesticides, the number and spatial coverage of those pesticide samples are not considered to be great enough to allow sediment cleanup criteria to be derived for them (although, as discussed in Section 9.1.4, pesticides are more strongly associated with adverse biological effects than are any of the shipyard chemicals). Indicator chemicals have been selected to represent each of the other major classes of sediment pollutants as follows:

- **Metals—**All metals, with the exception of selenium, are included. Selenium is excluded because of its relatively low detection frequency and because the detected values are equivalent to the quantitation limit. Because of the low concentrations and the narrow range of values, any relationships between selenium and biological effects are not expected to be meaningful. In addition, selenium is not considered to be a shipyard chemical (RWQCB 2001) and was included in sediment analyses only because it was specified for tissue analyses (RWQCB 2001).

- **Butyltins—**TBT is selected as an indicator chemical because it is the form of butyltin that was produced and applied in marine antifouling paint and is the most toxic of the butyltins.

- **PCBs and PCTs—**PCBs are represented by the sum of PCB homologs. The sum of PCB homologs is a more accurate representation of total PCBs in sediment than the sum of congeners (because not all congeners were measured) or the sum of Aroclors® (see Section 4.1.3). PCTs are represented by the sum of the PCT Aroclors® measured.

- **PAH—**PAH compounds are represented by the sum of all HPAH. Because most LPAH compounds were undetected, the accuracy of sums of LPAH compounds and of all PAH compounds is compromised by the inclusion of undetected values. Total HPAH therefore more accurately represents variation in overall PAH concentrations, and includes the most toxic PAH compounds.

- **Petroleum Hydrocarbons—**Petroleum hydrocarbons are represented by both DRO and RRO. GRO was not detected in sediment, and so is not included. Total petroleum hydrocarbons—the sum of all three compounds—is not used because of inaccuracy that would result from inclusion of quantitation limits for the GRO.

The second step in the identification of indicator chemicals is the evaluation of relationships between these chemicals and biological responses. Results of the three toxicity tests, benthic community assessment, and bioaccumulation testing conducted in Phase 1 of this study were all used to evaluate the potential of such relationships. Inclusion of chemicals with relationships to toxicity tests or benthic community conditions addresses possible impacts on aquatic life. Inclusion of chemicals with bioaccumulation potential addresses possible impacts on aquatic-dependent wildlife or human health. Chemicals were selected as indicator chemicals if they had any statistically significant relationship with amphipod mortality, echinoderm fertilization, bivalve development, total benthic macroinvertebrate abundance, total benthic macroinverte-brate richness, or tissue chemical concentrations in *Macoma.* The results of these comparisons are summarized in Table 9-8 (see Section 7 and Table 9-1 for details). Cadmium, chromium, nickel, silver, and PCTs have no relationship to any type of biological effect. Chemicals selected as indicator chemicals, and retained for the calculation of cleanup levels, are arsenic, copper, lead, mercury, zinc, TBT, total PCB homologs, DRO, and RRO.

9.3.2 Triad Data

The paired chemistry and biological effects data collected during Phase 1 allows computation of a no-effect concentration for each chemical and type of biological effect. The no-effect concentrations are known as apparent effects thresholds (AETs). This section presents AETs calculated for each individual type of biological effect. This section also includes a discussion of the following aspects of AET calculations and performance:

- The ranges of chemical concentrations for which biological effects have been measured
- The spacing of chemical concentrations around AET values
- Combining the different AET values
- The performance of AET values for prediction of biological effects.

For the purpose of deriving a chemical-based cleanup level that is maximally protective of aquatic life beneficial uses, the lowest of all of the individual AET values (the LAET) is also presented.

9.3.2.1 Range of Chemical Concentrations Evaluated

The data set used to derive an AET for any chemical should include a wide range of concentrations, so that both effects and the absence of effects can be observed. Concentrations of chemicals potentially associated with shipyard activities that were measured at the NASSCO and Southwest Marine shipyards covered a range of approximately 1 to 2 orders of magnitude; ranges for some chemicals were as much as 3 orders of magnitude (Table 9-9). Previous estimates of biological effect thresholds in San Diego Bay (PTI 1992) are within the ranges of chemical concentrations measured at NASSCO and Southwest Marine, except for TBT (for which recently measured concentrations are considerably lower). Biological effects-based sediment quality standards adopted by the State of Washington are also within the range of concentrations measured at the shipyards. The broad ranges of concentrations observed for all chemicals, and the comparison to similar effects-based sediment quality criteria indicate that the Phase 1 data set for the NASSCO and Southwest Marine shipyards is adequate for the calculation of AET values.

9.3.2.2 Biological Effects and AET Values

Several different types of biological effects were measured as part of the Phase 1 studies. These studies included three types of laboratory toxicity tests and an analysis of the benthic macroinvertebrate community. The results of the amphipod, echinoderm, and bivalve toxicity tests are presented in Section 6, and the results of the benthic macroinvertebrate analyses are presented in Section 8. Biological effects presented in those sections, and this one, were determined solely on the basis of statistically significant differences from reference areas. These evaluations of statistical significance do not include a determination of whether the difference is biologically significant (e.g., has actual ecological effects) or is caused by any specific shipyard-associated chemicals. Thus, basing the AET calculation on any statistically detectable difference is the most protective approach to the interpretation of possible effects.

The presence or absence of a statistically significant adverse biological effect at each triad station was determined for each of the toxicity tests. Effects on the benthic macroinvertebrate community were characterized as either minor, moderate, or major depending on the type and extent of the differences from reference conditions, as described in Section 8. AET values have been calculated for each type of effect. AETs for both major and minor effects have been calculated for the benthic macroinvertebrate community (moderate differences from reference conditions were included with major differences when calculating the AET values). In addition, AETs have been calculated for BRI values, although, as described in Section 8, the BRI-based effect determinations are not considered to be as appropriate as the more comprehensive assessment of the benthic community. All of these AET values are presented in Table 9-10.

Of the 30 stations sampled in Phase 1, 6 stations showed toxicity to amphipods, none showed toxicity to echinoderm gametes, 12 stations showed toxicity to bivalve larvae, and 13 stations showed at least minor differences from reference in the benthic community. For most chemicals, none of these effects, with the exception of minor differences in the benthic community, were associated with the highest concentrations of the chemical. In such cases, when there is no evidence of toxicity (or differences in the benthic community) at the highest concentration of a chemical, then the no-effect concentration, or the AET, is the highest concentration. Because toxicity may not be found at even higher concentrations of this chemical, the actual threshold of effects might be even higher than the maximum measured concentration. For this reason, the AET values that correspond to the maximum observed concentration of a chemical are annotated with a “G” qualifier in Table 9-10. These AET values are customarily referred to as “greater-than” values, indicating that the actual threshold of effects may be greater than the AET.

The occurrence of some “greater-than” AETs in any given data set is not unusual, and indicates that any observed toxicity is not attributable to the corresponding chemical. This reflects the lack of significant correlations between most chemicals and biological effects (Table 9-1). In the Phase 1 data set, the AETs for all chemicals for amphipod survival, echinoderm fertilization, and bivalve development are “greater-than” values. The “greater-than” values in this data set indicate that the effects observed for amphipods, echinoderms, and bivalves are not attributable to any measured chemical.

9.3.2.3 Resolution of Chemical Concentrations around the AET

One means of assessing an AET’s potential reliability is to review the distribution of measured concentrations around the AET (PTI 1988). If other samples without effects have concentrations similar to the AET, these results corroborate the appropriateness of the AET value. The difference between the AET and the next highest measured concentration (a concentration at which a biological effect was observed) represents the range of concentrations within which the threshold of effects actually lies. The next highest measured concentration above the AET is an upper bound on the AET value. If the difference between the AET and its upper bound is large, then there is more uncertainty about the actual threshold of effects, and the AET value is potentially biased low as an estimator of effects.

Because the AET values for all toxicity tests are “greater than” values, the upper bound is not known. The actual threshold of effects may be greater than these AET values, and the AET values for toxicity tests are therefore potentially biased low.

The distributions of concentrations around the AET for any benthic difference from reference are shown in Figure 9-5 for metals, TBT, total PCBs, and petroleum hydrocarbons. The established guideline for assessing the relationship between AET concentrations and the next lowest no-effect concentration is a factor of 3 (PTI 1988; Ecology 1995). That is, if the highest no-effect concentration is no more than a factor of 3 greater than the next lowest no-effect concentration, the AET value is considered to be representative. All of the AET values for minor benthic effects meet this criterion for protectiveness.

9.3.2.4 Lowest AET (LAET) Values

When multiple sets of AET values have been derived for independent measures of potential biological effects, they can be combined to derive a single set of AET values that is protective of all of the types of biological effects. This is done by taking the lowest of any of the individual AET values for each chemical. The result is referred to as a set of "lowest AET" or "LAET" values. The LAET values are the basis for effects-based chemical cleanup levels. As such, LAET values represent very conservative estimates of cleanup levels because higher concentrations of substances would still be protective of other independent measures of effects. Thus, the LAET is the most conservative estimate possible for cleanup levels protective of aquatic life.

Among the set of AETs for different toxicity tests and for benthic community differences from reference, the AET for any benthic community difference from reference is the lowest for every chemical. The LAET is therefore equal to the AET for any benthic community difference from reference.

9.3.2.5 Reliability of AET

Several different quantitative measures can be used to assess the ability of a set of sediment quality values to predict observed effects. These measures include:

- **Sensitivity—**The fraction of all samples with observed effects that are correctly predicted to have effects

- **Specificity—**The fraction of all samples without observed effects that are correctly predicted to have no effects (Shine 2003)
- **Efficiency—**The fraction of all predicted effects that were correctly predicted
- **Reliability—**The fraction of all samples that are correctly predicted either to have effects or to have no effects.

Values of reliability, sensitivity, efficiency, and specificity are all expressed as a percentage, and range between zero and 100. Sensitivity and specificity are complementary measures of the accuracy of predictions of effects and of no effects, respectively. Reliability is an overall measure of predictive accuracy, incorporating both correct predictions of effects and correct predictions of no effects. The relationship among these quantities is illustrated in Figure 9-6; equations for reliability, sensitivity, and efficiency are presented in the work plan (Exponent 2001).

Sensitivity, specificity, efficiency, and reliability have been calculated for each of the sets of values in Table 9-10, in relation to the potential aquatic life beneficial use impairment (Table 9-7).

The two highest categories of potential beneficial use impairment (likely and highly likely) were combined, and the two lowest categories (possible and unlikely) were combined, for the purpose of calculating performance measures. These combined categories were used to represent the likelihood of actual effects due to shipyard-associated chemicals. Sediment chemistry data from the triad samples were compared to the AET values in Table 9-109 to derive a set of predicted effects. Performance measures were then calculated in the manner illustrated in Figure 9-6. The calculated performance measures are shown in Table 9-11.

Different performance measures vary considerably among different sets of AET values. Sensitivity for all of the sets of AET values is low. This is a consequence of the weak or absent correlations between measured chemical concentrations and biological effects (Table 9-1). The possible explanations for the absence of significant correlations (discussed previously) also apply to the interpretation of the relatively low sensitivity values.

The AETs for amphipod toxicity, bivalve toxicity, and moderate to major benthic alterations all have the highest overall reliability, at 70 percent, but their sensitivity is 0 (zero) percent and their specificity is 100 percent. The LAET and the BRI AET both have an overall reliability of 67 percent, and a high specificity (90 percent), but a low sensitivity (11 percent). Unlike the other individual AETs, the LAET and the BRI AET have some balance between overpredicting and underpredicting effects. For use as a cleanup criterion for protection of aquatic life, therefore, the amphipod (or bivalve or moderate-to-major benthic community) AET provides the lowest overall error rate, whereas the LAET provides some balance between false positive and false negative errors, at the cost of a slightly higher overall error rate. The calculated performance measures are shown in Table 9-11.

Sensitivity for all of the effects-based measures is relatively low. This is a consequence of the weak or absent correlations between measured chemicals and biological effects (Table 9-1). The possible explanations for the absence of significant correlations (discussed previously) also apply to the interpretation of the relatively low sensitivity values.

Sensitivity and overall reliability are greater for all AETs when they are tested against the benthic endpoint than when they are tested against the pooled endpoint. This is attributable to the fact that the toxicity test responses are not correlated with chemical concentrations or with the benthic community responses.

In contrast to the amphipod and bivalve AETs, which predict no effects, and the Regional Board-defined background values, which predict an effect in every sample, the benthic effect AETs achieve a better balance between sensitivity and specificity. These results indicate that at this site, benthic AETs are the most generally useful predictors of potential sediment quality effects. Between the two benthic effect AETs, the AET for any benthic effect (equivalent to the LAET) performs better than the BRI-based AET.

9.3.3 Pore Water Data

Pore water data were collected during Phase 2 to assess these data relative to water quality criteria established in the CTR, as specified in the Regional Board's guidance for the investigation (RWQCB 2001). Although there are no water quality criteria for PAH in the CTR, in October 2002 Regional Board staff requested that pore water also be analyzed for PAH (Robertus 2002c, pers. comm.) and identified the PAH concentration that Regional Board staff had selected as a criterion for evaluation of pore water (Alo 2002c, pers. comm.). The following sections describe application of the equilibrium partitioning approach to chemicals with CTR criteria, and evaluation of PAH data with respect to the criterion selected by Regional Board staff.

9.3.3.1 Equilibrium Partitioning Approach

The conceptual basis for applying water quality criteria to assess sediment conditions rests upon equilibrium partitioning theory. Three assumptions underlie this theory: 1) that sediment solids and pore water are in thermodynamic equilibrium, 2) that sediment-dwelling organisms are primarily exposed to contaminants via pore water, and 3) that the sensitivities of sediment-dwelling organisms are equivalent to those of the organisms tested to derive surface water quality criteria. Because rates of particulate sorption and desorption of chemicals can differ, thermodynamic equilibrium can take a long time to become established (Landrum and Robbins 1990). In locations of relatively frequent sediment alteration or disturbance, the consequence can be that sediment and pore water never reach thermodynamic equilibrium. For this reason, the EPA Science Advisory Board specifically notes that the assumption of equilibrium may not be valid in areas of boat and barge traffic (U.S. EPA 1992a). This limitation unquestionably applies to the shipyards. The second assumption is also questionable because there are many benthic macroinvertebrates that neither respire nor ingest pore water. Organisms that burrow in fine-grained sediment typically respire overlying water either directly (as is the case for bivalves), or through irrigation of their burrows with overlying water (as is the case for polychaetes). (Shipyard SPI photographs that intersect deep polychaete burrows typically show a halo of oxidized sediment around the burrow that results from this irrigation process.) Filter-feeding and surface deposit-feeding benthic organisms are not be exposed to pore water by ingestion. Integumentary exposure is limited in the case of burrowing organisms that irrigate their burrows with surface water, or that have hard shells (e.g., bivalves) or exoskeletons (e.g., crustacea). Thus, the validity of the assumptions underlying the equilibrium partitioning approach are questionable.

The equilibrium partitioning approach, as ordinarily applied, is also based on the assumption that chemical concentrations in sediment and pore water are proportionally related. Field measurements have previously shown that this representation of partitioning behavior is inaccurate because of the presence of colloids in the “dissolved” fraction (Baker et al. 1986). This proportional representation of partitioning behavior is demonstrably untrue for most of the chemicals measured during this investigation, as described in the previous section *Pore Water Data*: for some, there is no relationship whatsoever, and for the rest, most have non-zero intercepts and some have non-linear relationships. For chemicals for which statistically significant relationships were found, however, a regression approach can be used instead of assuming a proportional relationship. As described previously, the positive pore water intercepts are most plausibly attributable to fine suspended or flocculent material that was not removed by centrifugation. The bias represented by these positive intercept values was removed by subtracting the intercept value from each of the observed pore water concentrations. Data were also transformed as necessary to meet the requirement for homogeneity of variance (square root transformations of sediment, pore water, or both). The resulting data set for each chemical then did actually have a proportional relationship between sediment and pore water concentrations, and regressions were re-run. Inverse predictions (i.e., of sediment concentrations) were then made for the CTR water quality criterion for each chemical. The measured variability in the pore water to sediment relationship was used to calculate both upper and lower 95 percent confidence limits on the predicted sediment concentrations. These values are shown in Table 9-12. Only sediment concentrations exceeding the 95 percent UCL are associated with pore water concentrations that are statistically significantly greater than the water quality criterion.

The range represented by the upper and lower 95 percent confidence limits of the predicted sediment concentration is very large for all chemicals except mercury. The large ranges result from variability in the paired sediment and pore water data, and indicate that there is a very high level of uncertainty associated with predictions of sediment concentrations from pore water concentrations. The variability of the data may result from several sources. One possible source is variability inherent in field or laboratory methods (all methods used in this investigation followed EPA guidelines). The variation could also be due to inappropriateness of the underlying assumptions that pore water and sediments are at thermodynamic equilibrium and that partitioning occurs solely between dissolved and particle-associated phases (Landrum and Robbins 1990; Baker et al. 1986). Partitioning to colloids, and the inability of current techniques to separate colloidal material from extracted pore water, can greatly complicate efforts to identify the true relationship between truly dissolved and particle-associated contaminants.

The existence of a positive pore water intercept in the fundamental relationship between pore water and sediment for this data set strongly suggests that colloidally bound chemicals are included in the "pore water" fraction. Because the locations sampled varied in physical characteristics, the amount of colloidal material in the "pore water" fraction almost certainly varied also, and this variation could be a major contributor to inaccuracy and imprecision of observed pore water : sediment relationships. Although subtraction of the positive pore water intercept value is likely to have compensated for a major component of the inaccuracy, the bias it represents may not actually be constant across all conditions. Furthermore, there is no similar way to compensate for the imprecision resulting from the presence of colloidal material. Whatever the reason for variability in the data, however, the consequence is that predictions of sediment chemistry concentrations from pore water concentrations are so highly variable as to have no practical utility for the establishment of pore water based candidate cleanup levels.

9.3.3.2 PAH in Pore Water

The Regional Board staff's specification of PAH analyses in pore water were that the sum of 34 individual PAH compounds should be no greater than 2.71 μg/L (Alo 2002c, pers. comm.). The 34 individual PAH compounds specified included alkylated forms of commonly measured PAH compounds. Because Regional Board staff provided their specifications for pore water analyses after all other Phase 1 and Phase 2 sampling had been completed, this suite of 34 PAH compounds was measured only in the sediment at the stations sampled for pore water. These PAH measurements in sediment and pore water are reported in Appendices B and D, respectively.

The sums of the 34 individual PAH compounds in pore water samples are shown in Table 9-13. As for most other compounds measured in pore water, Station SW02 is an outlier for PAH. As described previously, this result is consistent with the visual observation of suspended material remaining after centrifugation. The measured concentration at SW02 therefore cannot be considered representative of actual pore water. At all of the other stations, the sum of the 34 PAH compounds is less than the criterion selected by Regional Board staff. The data therefore indicate that there are no potential impacts on beneficial uses from PAH in pore water. Consequently, no modification of the candidate cleanup criteria is needed to address PAH in pore water.

9.3.4 Fish Histopathology

Fish histopathology data contain some indicators of potential chemical exposure at the shipyards, but none of the neoplasms or preneoplastic lesions found are in fish from highly polluted sites. The total number of lesions that were significantly elevated was less at the shipyard sites than at the reference site. The one type of lesion most strongly associated with the shipyard sites—abundant lipofuscin in liver cells—could be attributable to chemical exposure or to fish age. No overall effects on physiological condition are apparent in fish from the shipyards: the size of fish at a given age, and the weight of fish for a given size, were both equivalent at shipyard and reference sites, or slightly greater at the shipyards. Because no adverse effects to fish can be clearly associated with specific chemical concentrations in the sediment, the fish histopathology data cannot be used to derive specific chemical-based cleanup levels.

9.3.5 Fish Bile Data

Concentrations of PAH metabolites in fish bile from locations inside and outside each of the shipyard leaseholds were compared to reference area conditions. Concentrations of all three classes of PAH metabolites inside both shipyards were not significantly different from reference conditions ($p>0.05$, one-tailed ANOVA using log-transformed protein-normalized concentrations). Concentrations of phenanthrene and benzo[a]pyrene at locations outside both shipyards were significantly greater than concentrations inside the shipyards and greater than reference area concentrations. These results indicate that, although there appears to be a source of PAHs in the local region of San Diego Bay outside the shipyards, fish exposures to PAH at the shipyards themselves are equivalent to that at the reference area. Because there is no significant fish exposure to PAH at the shipyards, no cleanup criteria for PAH need to be derived or modified to protect fish against exposure to PAH at the shipyards.

9.3.6 Summary

Minor alterations in the benthic macroinvertebrate community are the only adverse effects on aquatic life at the shipyards on which cleanup criteria can be based. Moderate to major effects on benthic macroinvertebrates, effects on amphipod survival, and effects on bivalve development all have no-effects levels that are, for most chemicals, the highest concentrations observed at the shipyards. This is a consequence of the general absence of statistically significant relationships between any of these types of effects and measured sediment chemical concentrations. Relationships between chemicals in pore water and sediments are so highly variable that they cannot be used to derive realistic cleanup levels. Histopathology data show some tissue alterations in various locations, including the reference area, but neoplasms and preneoplastic lesions are completely absent from fish at the site, and there is no basis for establishing chemical-specific cleanup levels using the available data. Fish bile data show that fish exposure to PAH at the shipyards is no different than at reference areas, and so do not provide any basis for establishing chemical-specific cleanup levels.

LAET values (equivalent to no-effect values for any alteration of the benthic macroinvertebrate community) therefore are the only cleanup levels for the protection of aquatic life that should be evaluated further. However, it should be noted that these cleanup levels have considerable uncertainty because of the general lack of association between shipyard chemicals and biological effects. This lack of association does not result from a problem with the triad approach itself, but stems from:

- The low levels of effects (or absence of effects) observed for most biological indicators
- Some potentially important causal agents (e.g., pesticides) that were not measured
- The confounding effects of physical disturbance of sediments at some locations.

10 Risk Assessment for Aquatic-Dependent Wildlife

This risk assessment was conducted to evaluate the potential for adverse effects to aquatic-dependent wildlife receptors (marine reptiles, birds, and marine mammals) occurring at the NASSCO and Southwest Marine shipyards. This assessment is consistent with the most recent guidance from EPA for conducting ecological risk assessments (U.S. EPA 1996b, 1997a, 1999a,b, 2000c, 2001b,c). Ecological risks have been evaluated for those receptors and pathways identified in the conceptual site model (Figure 1-3).

10.1 Selection of Assessment Endpoints

EPA defines assessment endpoints as "explicit expressions of the actual environmental values (e.g., ecological resources) that are to be protected" (U.S. EPA 1997a). These environmental values are manifested in endpoints that generally focus on the community or population level of biological organization. The protection of a threatened or endangered species, however, is an assessment endpoint that focuses on individual organisms. The conceptual site model identifies several aquatic life communities and aquatic-dependent wildlife communities that may be exposed to chemicals in sediment, surface water, and food at the shipyards. The assessment endpoints for the risk assessment are the protection of the following communities from adverse ecological effects resulting from exposure to chemicals at the shipyards:

- Submerged aquatic plants
- Benthic and epibenthic macroinvertebrates
- Fish
- Marine reptilian herbivores (plant-eaters)
- Avian piscivores (fish-eaters)
- Aquatic avian invertivores (invertebrate-eaters)
- Marine mammalian piscivores.

10.2 Selection of Measurement Endpoints

Measurement endpoints provide the actual data used to evaluate attainment of each assessment endpoint. The following sections describe the measurement endpoints that correspond to each assessment endpoint addressed in the risk assessment.

10.2.1 Submerged Aquatic Plants

Three measurement endpoints were selected to assess the health of submerged aquatic plant communities at the shipyards:

- Measured chemical concentrations in plants at the shipyards compared to measured chemical concentrations in plants at the reference areas
- Measured chemical concentrations in plants at the shipyards compared to phytotoxicity benchmarks and other literature values
- Spatial distribution of plants at the shipyards relative to bathymetry and other physical factors and in relation to other sites in San Diego Bay.

10.2.2 Aquatic-Dependent Wildlife

For assessment endpoints such as the protection of marine reptile and avian and aquatic mammal communities, appropriate indicator species, or ecological receptors, were selected to represent the broader wildlife communities. Ecological receptor selection is discussed below in Section 10.5.2, *Receptor Selection*. Daily dietary exposures to chemicals in sediment and food were estimated for each representative receptor using a food-web exposure model. Dietary exposures were compared to toxicity reference values (TRVs) derived from the ecotoxicology literature to assess the ecological risks to aquatic-dependent wildlife species and communities. The food-web exposure modeling approach and the development of TRVs are discussed below in Sections 10.5.4 and 10.6.

10.3 Risk Questions

Ecological risk questions are based on the assessment endpoints, and they provide a basis for evaluating the results of the site investigation. This ecological risk assessment for aquatic-dependent wildlife addresses the following risk questions:

- Are submerged aquatic plants exposed to chemical concentrations that will impair their growth and/or survival?
- Are benthic invertivorous fish exposed to chemical concentrations that will impair their reproduction, growth, and/or survival?
- Are aquatic-dependent wildlife receptors exposed to chemical concentrations in their diet that will impair their reproduction, growth, and/or survival?

10.4 Exposure and Effects Characterization for Eelgrass

Chemical concentrations were measured in eelgrass samples from each of the shipyards and the reference area (Station 2240). Samples of emergent vegetation (stems and leaves) were collected by divers, and samples were not washed before analysis so that concentrations would be representative of ingestion by sea turtles. Reported concentrations therefore include the chemicals on the surface of the plants as well as incorporated into plant tissue. Concentrations of detected chemicals in the eelgrass samples were systematically higher at the shipyards than at the reference area. Concentrations in shipyard samples were typically 2 to 5 times higher than concentrations in reference area samples. Because only a single eelgrass sample was collected from each location, statistical comparisons of site and reference were not performed. Nevertheless, the consistently higher chemical concentrations in samples from the shipyards indicate that eelgrass at the shipyards do show increased exposure to chemicals.

A literature search was conducted for studies of chemical concentrations in eelgrass that could potentially cause adverse effects. A few studies examined the effects of metals, and one study examined the effects of TBT on eelgrass. However, a majority of the studies identified examined uptake rates of metals and concentrations of metals in various components of the plant (e.g., rhizomes, roots, and leaves) and/or evaluated the significance of eelgrass in heavy metal cycling in coastal areas (Brix et al. 1983; Drifmeyer et al. 1980; Lyngby and Brix 1984, 1987, 1989). A number of studies reported concentrations of metals in eelgrass populations from areas with known metal contamination. However, none of these studies provided any indications of adverse effects to eelgrass populations associated with chemical exposures.

Williams et al. (1994), in a review of metal accumulation in salt marsh environments, noted that very few studies have assessed the toxicity of metals towards halophytic plants or noted vegetation disorders in natural salt marsh plant communities. Prange and Dennison (2000) demonstrated that changes in photochemical efficiency and amino acid concentrations and composition in various seagrasses (not including eelgrass) were affected by additions of heavy metals to seawater. However, exposures were to metals in seawater rather than in sediment, and the responses were species specific.

Lyngby and Brix (1984) found that addition of 0.35 mg/L (5 μM) of copper or 10 mg/L (5 μM) of mercury inhibits the growth of eelgrass. Cadmium (at 5 μM) and zinc (at 50 μM) also had significant, but less toxic, effects. Toxic effects are linked to metal availability and, therefore, soil and water chemistry (Lyngby and Brix 1987). For example, the availability of mercury to marsh plants increases as organic matter decreases, and cadmium availability in sediment increases under acidic-oxidizing conditions (Williams et al. 1994). Lyngby and Brix (1984) concluded that significant phytotoxicity on eelgrass due to metal contamination probably does not occur because the concentrations observed in the study to have toxic effects are generally much higher than those found in natural and polluted waters.

TBT has been shown to be rapidly accumulated in eelgrass from seawater (Francois et al. 1989), but no adverse effects have been demonstrated (Williams et al. 1994). TBT concentrations of 809 μg/kg (dry weight) in plant tissue did not adversely affect eelgrass productivity (Francois et al. 1989). In contrast, the maximum TBT concentration in eelgrass at the shipyards was 19 μg/kg (dry weight).

The eelgrass distribution at the shipyards was surveyed by divers during Phase 2, and is shown in Figures 2-8 and 2-9. Eelgrass was present in the shallowest water near the shore at the east and west ends of both shipyards. Eelgrass was not present in the center of either of the shipyards, where most ship construction and repair activities take place. Sediment chemical concentrations are also generally highest near shore, and the spatial distribution of eelgrass therefore does not appear to be limited by sediment chemicals.

Because no phytotoxicity benchmarks for eelgrass exposure to sediments are available, quantitative risk or hazard assessments for eelgrass at the shipyards cannot be made. Based on the distribution of eelgrass at the shipyards, sediment chemicals do not appear to be limiting its presence. Overall, therefore, there is no indication of adverse effects on eelgrass at the shipyards.

10.5 Exposure Characterization for Aquatic-Dependent Wildlife

10.5.1 Definition of Assessment Units

To focus the risk analysis, the shipyards were divided into four discrete assessment units: the area inside the NASSCO leasehold (inside NASSCO); the area between the NASSCO leasehold and the shipping channel (outside NASSCO); the area inside the Southwest Marine leasehold (inside Southwest Marine); and the area between the Southwest Marine leasehold and the shipping channel (outside Southwest Marine). Ecological risks to receptors foraging in each assessment unit were evaluated separately to identify areas with a greater likelihood for adverse ecological effects. Therefore, separate chemical exposure estimates were developed for each receptor in each of the four assessment units (see Section 10.5.5, *Quantified Sources of Chemical Exposure*, for exceptions). In addition, exposure estimates were calculated for each receptor in two reference areas: the original reference area consisting of 5 sediment stations sampled during the Phase 1 and Phase 2 sediment investigations, and the final reference pool defined by Barker (2003, pers. comm.).

10.5.2 Receptor Selection

Sediment cleanup levels to be developed for the shipyards are intended to be protective of the species that constitute a guild, such as all piscivorous birds that may forage at the shipyard sites. However, it is not practical to assess risk separately for every individual species in a guild, nor is this required under ecological risk assessment guidelines (U.S. EPA 1998a). Instead, representative species are selected for the guild, and conclusions regarding risk to these species are considered to apply to all ecologically similar species. Ecological receptors were selected to represent guilds of higher trophic-level organisms that may be exposed to chemicals at the shipyards, including marine reptiles, aquatic-dependent birds, and marine mammals. Representative receptors were chosen based on characteristics such as their occurrence at the site, feeding habits, known sensitivity to contaminant exposure, and the availability of pertinent life history information. The following sections describe the ecological receptors evaluated in this risk assessment.

10.5.2.1 Marine Reptiles

One species of marine reptile, the East Pacific green turtle (*Chelonia mydas agassizii*), is known to occur in San Diego Bay. The East Pacific green turtle is generally classified as a small, darkly colored subspecies of the green turtle, which inhabits circumtropical marine waters around the world. This species is a long-lived, migratory sea turtle that roams the coastal waters of North, Central, and South America from Alaska to Chile (NMFS and FWS 1998). Because this species requires warm waters, however, San Diego Bay is generally considered the northernmost extent of its permanent range in western North America (Navy and SDUPD 2000). These turtles nest at various times of the year on sandy beaches in Michoacán, Mexico, the Galapagos Islands, Ecuador, and other secondary sites along the Pacific coast of Central America. There are no known East Pacific green turtle nesting grounds on the west coast of the United States (NMFS and FWS 1998). A small group of East Pacific green turtles resides in southern San Diego Bay, where turtles congregate in the warm effluent of the San Diego Gas and Electric Company power plant, and feed on eelgrass beds (FWS 1998). They are present in the effluent channel throughout the year, though in lower numbers in the summer, when turtles may disperse into the bay to avoid elevated water temperatures (Dutton and McDonald 1992).

Green turtles feed primarily on algae and sea grasses in shallow seawater; often, they will eat exclusively one or the other, perhaps because they develop disparate gut microflora depending on their food (Bjorndal 1985). Green turtles also consume small quantities of marine invertebrates (Whiting and Miller 1998; NMFS and FWS 1998), but the San Diego Bay adult population appears to be herbivorous, feeding on eelgrass and red and green algae species (Dutton and Dutton 1997).

The green turtle is listed as threatened wherever found, except for breeding colony populations in Florida and on the Pacific coast of Mexico, which are listed as endangered (NMFS 2001). The San Diego Bay population is predominantly a part of the Mexican breeding population, and as such, is endangered (Navy and SDUPD 2000). Threats to green turtles include poaching, accidental takes by commercial fishing operations, watercraft collisions, and ingestion of and entanglement in marine debris (NMFS 2001). There is significant ecological information in the literature about green turtles, including studies of the San Diego Bay population.

10.5.2.2 Aquatic-Dependent Birds and Marine Mammals

Numerous bird and mammal species can potentially occur at or near the shipyard sites. For example, 280 species of marine and coastal birds have been recorded in San Diego Bay (Navy and SDUPD 2000). These species belong to various ecological guilds, and some bird species may feed primarily on small forage fishes, whereas other species may use shellfish in sediments as their primary food source.

Fish-eating marine birds and mammals and mollusc-eating birds have been identified as important groups of aquatic-dependent wildlife that could be at risk due to exposure to chemicals in prey species at the shipyard sites. Five species have been selected as suitable representative receptors for assessing potential risk to these groups, specifically California least tern, California brown pelican (*Pelecanus occidentalis californicus*), western grebe (*Aechmophorus occidentalis*), surf scoter (*Melanitta perspicillata*), and California sea lion (*Zalophus californianus*). These five receptors are present in the San Diego Bay area for significant periods annually, and they feed primarily or exclusively on fish and shellfish, both likely exposure media. The California least tern and California brown pelican are federal and state listed endangered species. In addition, there is adequate life history information available for these five species to support their use in an ecological risk assessment. Capsule ecological summaries of these five receptors are provided below.

California Least Tern—The least tern is selected as a receptor representative of marine birds that may feed on small fish at the shipyard sites. The least tern, the smallest of the North American terns (NGS 1987), is a migratory, piscivorous bird whose California subspecies is known to nest at sites in San Diego Bay, nearby Mission Bay, and the Tijuana River Valley during its breeding season, typically April through August in southern California (Keane 2000). California least terns nest on open, flat substrates near the coast (CDFG 2001a), such as beaches and salt flats, and are faithful to their nesting sites, tending to return to their natal colony or the previous year's site (Atwood and Massey 1988). In the fall, the terns depart for wintering grounds in Central and South America (Keane 2000).

During the summer months, least terns usually forage within 3.2 km of their nests in almost any water body that contains suitable prey (Atwood and Minsky 1983). Least terns feed primarily on small, narrow-bodied fresh and saltwater fish, generally 2–9 cm in length and less than 1.5 cm deep, but they may also prey opportunistically on aquatic invertebrates like small crustaceans and insects (Thompson et al. 1997). Atwood and Kelly (1984) reported that northern anchovies (*Engraulis mordax*), topsmelt (*Atherinops affinis*), jacksmelt (*Atherinopsis californiensis*), deepbody anchovies (*Anchoa compressa*), and slough anchovies (*Anchoa delicatissima*) were the dominant prey items of least terns at California breeding colonies. Least terns are not diving birds; instead, they capture fish by swooping down, grasping their prey in an open bill, and lifting the fish from the upper 15 cm of water (Thompson et al. 1997) or by spearing fish with a closed bill (Tomkins 1959).

Industrial and residential development of natural nesting habitat, disturbance of breeding areas by human recreation, and increased predation from animals associated with human encroachment (pets, introduced red foxes, crows, etc.) have all contributed to the decline of the California least tern (Keane 2000; Thompson et al. 1997). The bird has been a federal and California listed endangered species since 1970 and 1971, respectively (CDFG 2001a). Perhaps because of its protected status, the California least tern seems to be well studied, and there is ample life history information about this prospective receptor in the literature.

California Brown Pelican—The brown pelican is selected as a receptor representative of marine birds that may feed on small- to medium-sized fish at the shipyard sites. Although there is some overlap in size of prey used by least terns and brown pelicans, pelicans also consume larger fish than least terns, potentially including higher trophic level species, which could result in different patterns of exposure to bioaccumulative chemicals between the two receptors, and potentially different conclusions regarding risk to these species.

The California brown pelican is a large piscivorous bird that ranges year-round along the North American coast, from the Gulf of California to southern British Columbia, and breeds primarily on undisturbed islands in the Gulf of California and off the coasts of southern California, Baja California, and southern Mexico (NGS 1987; FWS 2001a; CDFG 2001b). There is only one breeding population of California brown pelicans in United States waters, the Southern California Bight population, which nests on the Channel Islands and islands off Baja California (CDFG 2001b). Beginning in mid-May, post-breeding individuals disperse along the coast; the species is fairly common to common in southern California throughout the year (Granholm 2001a). California brown pelicans are known to forage and roost in and around San Diego Bay (FWS 1998).

Brown pelicans feed almost exclusively on fish that are generally less than 30 cm in length (Kaufman and Peterson 2001; FWIE 2001). In California, Pacific mackerel (*Scomber japonicus*), Pacific sardine (*Sardinops sagax*), and northern anchovy dominate the brown pelican diet (FWS 2001a). The pelicans plunge head first after fish in both shallow and deep water and may entirely submerge in the process (Palmer 1962). Brown pelicans typically take fish from the top meter of seawater (FWIE 2001).

Like the California least tern, the California subspecies of the brown pelican is a federal and California listed endangered species (CDFG 2001b). Exposure to the bioaccumulative organochlorine pesticide DDT and its metabolite DDE through the ingestion of contaminated fish caused eggshell thinning and abnormal parental behavior that resulted in massive reproductive failure in California brown pelicans in 1969–1971 (Granholm 2001a; FWS 2001b). The population has since rebounded, but the species is still considered too vulnerable to delist in California (FWS 2001a). There is substantial literature on brown pelican life history.

Western Grebe—The western grebe, a diving, piscivorous bird, is a common winter resident in the bays, estuaries, and marine subtidal waters of the California coast. A closely related species, the Clark's grebe (*Aechmophorus clarkii*), occurs in mixed flocks with western grebes in California but is less common (Ratti 1981). Both species are relatively more abundant in the northern part of San Diego Bay (Navy and SDUPD 2000). The western grebe breeds on freshwater lakes and marshes, rarely on tidewater marshes, in areas of expansive open water and emergent vegetation, which it uses to build its floating nest (Kucera 2001; Palmer 1962; Storer and Neuchterlein 1992). Western grebes are known to nest in colonies at Sweetwater Reservoir in San Diego County (Cogswell 1977; Kucera 2001), but no reference to western grebes breeding in San Diego Bay has been found in the literature. Western grebes migrate to their wintering grounds on the Pacific coast in September and October and remain into early May (Kucera 2001; Palmer 1962; Storer and Neuchterlein 1992). The U.S. Fish and Wildlife Service (FWS) has recommended the western grebe as a year-round resident bird that may feed on fish at the shipyard sites. Although the species is a year-round resident in San Diego County, no information on its relative seasonal abundance in the bay was found. Life history information described above suggests that western grebes would be most common on the bay during winter, with some possible use by breeding birds during the summer. However, for risk assessment purposes, it will be assumed that western grebes could be resident year-round in the bay.

The western grebe's diet varies seasonally (Lawrence 1950) but largely comprises fish up to 20 cm in length (Ydenberg and Forbes 1988), including saltwater fish such as herring, topsmelt, jacksmelt, Pacific staghorn sculpin (*Leptocottus armatus*), and sea perch (*Cymatogaster* sp.; Palmer 1962). Western grebes also feed opportunistically on shrimp, crabs, and other crustaceans, limpets, insects, polychaete worms, and plant material (Lawrence 1950; Palmer 1962). However, they depend more heavily on fish than do other grebe species (Palmer 1962; Cogswell 1977). Western grebes generally dive to 1 m, but may go deeper to pursue prey (Lawrence 1950), which they capture by pinching or spearing with their bills, and they often swallow smaller prey underwater (Storer and Neuchterlein 1992). Bottom-dwelling fish, polychaete worms, and small stones identified in the stomachs of some western grebes suggest that these birds may forage in shallow-bottom sediments (Lawrence 1950; Storer and Neuchterlein 1992).

Surf Scoter—The surf scoter is selected as a receptor representative of diving marine birds that may feed on molluscs in soft sediments at the shipyard sites. The surf scoter is a diving sea duck that breeds in northern Canada and Alaska and winters along the North American coasts (Savard et al. 1998). It inhabits estuaries and coastal waters along the entire California coast from late September to early May (Granholm 2001b). Wintering birds congregate in the open ocean or in large bays (Granholm 2001b), including San Diego Bay. The Office of Migratory Bird Management of FWS reported that in 1994, central and south San Diego Bay harbored 72 percent of the south coast region's midwinter surf scoter population (FWS 1998). Surf scoter is the most abundant waterfowl species in San Diego Bay (Navy and SDUPD 2000). In 1993−1994 weekly bird counts, FWS observed about 80,000 waterfowl (representing 628,000 bird-use days) in the open waters of south San Diego Bay, 95 percent of which were surf scoter and scaup (FWS 1998).

Migrating and wintering surf scoters along the Pacific coast feed mainly on bivalves such as blue mussels (*Mytilus edulis*; Savard et al. 1998). In a study of scoter diets in coastal British Columbia, Vermeer and Bourne (1984) report that surf scoters ate mostly molluscs, including blue mussels, Manila clams (*Tapes philippinarum*), littleneck clams (*Protothaca staminea*), basket cockles (*Clinocardium nuttalli*), and soft-shelled clams (*Mya arenaria*), as well as snails, barnacles, and other crustaceans, and polychaete worms. Surf scoters dive for mussels and other foods to depths of 12 m or more (Cogswell 1977), which is considerably deeper than depths attained by most other surface-diving ducks.

The surf scoter does not have protected status in California nor the rest of the United States. Traditionally one of the least understood sea ducks, the surf scoter has been the object of many ecological and behavioral studies over the last 15 years or so (Savard et al. 1998), and good information is now available on its ecology and life history characteristics.

California Sea Lion—The California sea lion is selected as a receptor representative of marine mammals that may feed on fish at the shipyard sites. The California sea lion is the most abundant pinniped (seal) living along the coast of California (Riedman 2001). California sea lions are known to occur within San Diego Bay, where they use features such as rocks, piers, and buoys as haulout locations. Harbor seals (*Phoca vitulina*) are also common in southern California, but they breed offshore, and no record of harbor seal activity in San Diego Bay was found in preliminary literature research.

The California sea lion breeds from May to August in large rookeries on islands in the Gulf of California and off the Pacific coast south to Mazatlán, Mexico, and north to San Miguel Island. This species does not breed in San Diego Bay, and probably never has because of the access that land predators have to the beach (Navy and SDUPD 2000). After the breeding season, males migrate north as far as British Columbia, while females and their young generally remain in the vicinity of the rookeries (Riedman 2001; Peterson and Bartholomew 1967). California sea lions prefer to haul out and breed at undisturbed sites with easy access to water and food (Riedman 2001), although Peterson and Bartholomew (1967) describe the sea lion as relatively indifferent to humans. Animals haul out and congregate on beaches, rocks, floats, and jetties (Riedman 2001). California sea lions were observed in central and south San Diego Bay during bird surveys conducted by FWS in 1993–1994 (FWS 1998).

California sea lions feed primarily on small fish and cephalopods, including Pacific whiting (*Merluccius productus*), anchovies, herring (*Clupea* spp.), juvenile rockfish (*Sebastes* spp.), Pacific mackerel, squid, and octopus (Peterson and Bartholomew 1967; Keyes 1968; Whitaker 1997). They are capable of diving more than 130 m below the surface, and remaining underwater for 20 minutes at a time, but the animals tend to feed at shallower depths of 26–74 m (Whitaker 1997).

There were about 80,000 California sea lions in United States waters in 1989 (Whitaker 1997), and neither the State of California nor the U.S. government lists the species as threatened or endangered. There is ample life history information about California sea lions available in the literature.

10.5.3 Routes and Media of Exposure

Based on the chemical properties of the shipyard-associated chemicals and the typical foraging behavior of the aquatic-dependent wildlife receptors, the primary routes of exposure to chemicals at the shipyards are through the ingestion of prey items and the incidental ingestion of sediment during foraging. Therefore, the risk analysis was structured to focus on these two routes of exposure.

10.5.4 Wildlife Food-Web Exposure Modeling

Exposure of marine reptiles and aquatic bird and mammal communities to chemicals was estimated using a simple food-web exposure model. Food-web model input parameters were developed for the ecological receptors chosen to represent these wildlife communities.

A standard food-web modeling approach that is consistent with EPA's wildlife exposure guidance (U.S. EPA 1993a; 61 Fed. Reg. 47552) was used to calculate exposure. The food-web model estimates dietary exposure as a body-weight-normalized total daily dose for each receptor species. The general structure of the food-web exposure model is described by the following equation:

$$IR_{chemical} = \frac{\sum_i \left(C_i \times M_i \times A_i \times F_i\right)}{W} \qquad (1)$$

where:

$IR_{chemical}$ = total ingestion rate of chemical from all dietary components (mg/kg body weight-day)

C_i = concentration of the chemical in a given dietary component or inert medium (mg/kg)

M_i = rate of ingestion of dietary component or inert medium (kg/day)

A_i = relative gastrointestinal absorption efficiency for the chemical in a given dietary component or inert medium (fraction)

F_i = fraction of the daily intake of a given dietary component or inert medium derived from the site (unitless area-use factor)

W = body weight of receptor species (kg).

The term $IR_{chemical}$ can be expanded to specify each ingestion medium, which can include one or more primary food items and incidentally ingested sediment:

$$IR_{chemical} = [\Sigma (C_{food} \times M_{food} \times A_{food} \times F_{food}) + (C_{sediment} \times M_{sediment} \times A_{sediment} \times F_{sediment})]/W$$

This model provides an estimated total dietary exposure to chemicals resulting from consumption of food and the incidental ingestion of sediment on a mg chemical/kg body weight-day basis.

10.5.5 Quantified Sources of Chemical Exposure

Browse or prey selection is a function of the receptor's size and method of foraging. Many receptors consume a wide variety of food types, but for assessment purposes, browse or prey selection was restricted to one or several prey species in order to fully use site-specific data and to provide a conservative exposure assessment. Specific dietary exposure assumptions applied in the food-web models were as follows:

- East Pacific green turtle: 100 percent eelgrass and incidental sediment
- California least tern: 100 percent small fish and incidental sediment
- California brown pelican: 100 percent medium-sized fish and incidental sediment
- Western grebe: 100 percent small fish and incidental sediment
- Surf scoter: 100 percent molluscs and incidental sediment
- California sea lion: 100 percent medium-sized fish and incidental sediment.

Browse and prey items were collected at the shipyards and at reference area stations during the Phase 2 field investigation to provide tissue data to support the ecological risk assessment. One sample of eelgrass blades, a favored food of Pacific green turtles, was collected from eelgrass beds inside each shipyard and near Reference Station 2240 to represent dietary exposure to chemicals in the exposure model for the green turtle. Eelgrass was not collected outside either shipyard, where depth of water prevented growth of eelgrass beds; therefore, green turtle exposures were calculated only for the assessment units inside NASSCO and inside Southwest Marine and for the reference area. Chemical concentration data for eelgrass used in wildlife exposure modeling are summarized in Table 10-1.

Composite samples of forage fish were collected inside and outside each shipyard's leasehold and in the vicinity of Reference Station 2240. Forage fish data were used to estimate exposure to chemicals in food for the least tern and the Western grebe, the receptors that may consume small fish at the shipyards. Topsmelt were collected inside NASSCO and at the reference area; anchovies were collected outside NASSCO, inside Southwest Marine, and outside Southwest Marine. Forage fish were analyzed on a whole body basis. Chemical concentration data for forage fish used in wildlife exposure modeling are summarized in Table 10-2.

Five spotted sand bass (*Paralabrax masculatofasciatus*) were also collected inside and outside each shipyard's leasehold and in the vicinity of Reference Station 2240. Chemical concentrations in sand bass whole bodies were used to estimate exposure to chemicals in food for the brown pelican and the sea lion, the receptors that may eat medium-sized fish at the shipyards. Chemical concentration data for sand bass used in wildlife exposure modeling are summarized in Table 10-3.

Composite samples of benthic mussels (*Musculista senhousei*) were collected at Reference Station 2240 and at two stations inside each shipyard. Mussel tissue data (shell removed) were used to calculate the exposure to chemicals in food in the dietary exposure model for surf scoter. Mussels were not collected outside either shipyard; therefore, surf scoter exposures were calculated only for the assessment units inside NASSCO and inside Southwest Marine and for the two reference areas. Chemical concentration data for mussels used in wildlife exposure modeling are summarized in Table 10-4.

Biota station locations (eelgrass and benthic mussels) and collection areas (forage fish and sand bass) are presented in Figure 2-5.

Mean detected chemical concentrations in eelgrass (one sample), mussel tissue, or whole body fish were determined separately for each of the assessment units. For each individual sample, total PCBs are reported as the sum of all detected Aroclors®. For each individual sample, total PAHs are reported as the sum of the concentrations of 17 PAHs (2-methylnaphthalene, acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, phenanthrene, benz[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[j]fluoranthene, benzo[ghi]perylene, benzo[k]fluoranthene, chrysene, fluoranthene, indeno[1,2,3-cd]pyrene, and pyrene) with one-half the quantitation limits used for undetected compounds. When chemicals were not detected in any sample of a specific tissue type from within an assessment unit, chemical concentrations were expressed as the mean of one-half of the individual quantitation limits, or in the case of PAHs, as the mean of the sum totals of each individual sample. Mean chemical concentrations in biota collected near Station 2240 were used to model exposures in both the original and revised reference areas. Concentrations of chemicals in biota were expressed on a dry-weight basis in all exposure models to control for variations in water content among individuals and between biota and sediment. This standardization also permitted direct application to ingestion rates that were determined on a dry weight basis.

Surface sediment (0–2 cm) samples were collected at the shipyards and the original reference area during Phase 1 and Phase 2 field investigations. Surface sediment station locations are shown in Figures 2-1, 2-4, and 8-2. Mean detected values were calculated for surface sediment collected within general areas of the shipyard assessment units where fish were sampled. When chemicals were not detected in any sediment sample from within an assessment unit, chemical concentrations for that unit were expressed as the mean of one-half of the individual quantitation limits. Summations of PCB Aroclors® and individual PAHs in sediment were done according to the procedure described above for biota samples. These sediment chemical concentrations were used to estimate receptors' exposures to chemicals through incidental sediment ingestion at each assessment unit. Mean chemical concentrations across all five original reference stations and across stations in the final reference pool were also applied to estimate exposures in the original and revised reference areas, respectively. Chemical concentration data for sediment used in wildlife exposure modeling are summarized in Table 10-5.

10.5.6 Food Ingestion Rates

The ingestion rate of an organism is a function of its energy requirements, the energy density (energy content) of its diet, and the efficiency of the organism's energy assimilation from the diet. Average daily food consumption for the East Pacific green turtle was estimated from data in Bjorndal (1980). For all avian and mammalian receptors, estimates of prey intake rates were based on the bioenergetic scaling relations of Nagy (1987) and Nagy et al. (1999) and expressed as field metabolic rates (kJ/day). Estimates of assimilation energy and prey energy content reported in Nagy (1987) and Nagy et al. (1999) were applied to convert the metabolic rate to a daily intake rate expressed on a mass of food per mass of body weight basis (kg/kg-day). Body weight estimates for all of the receptors were determined from literature reports; mean values representative of populations indigenous to southern California were preferred, if available; otherwise, estimates for the next nearest population were used. Table 10-6 summarizes the body weights and food ingestion rates used in the food-web exposure model calculations.

10.5.7 Sediment Ingestion Rates

Ecological receptors may be exposed to chemicals in shipyard sediments that are ingested accidentally during foraging. Incidental sediment ingestion rates for ecological receptors were derived from soil ingestion estimates for various wildlife species developed by Beyer et al. (1994) and expressed as percentages of daily food intake. The sediment ingestion rate for the green turtle was based on the percentage of soil in the diet of the eastern painted turtle (Beyer et al. 1994). Sediment ingestion rates for the western grebe and the surf scoter, birds that may feed on invertebrates in the sediment, were estimated using the range of percentage of soil in the diets of duck species reported by Beyer et al. (1994). Sediment ingestion rates for the brown pelican, the least tern, and the sea lion, receptors that tend to feed in the water column were based on the minimum percentage of soil in wildlife diets reported by Beyer et al. (1994). Table 10-6 presents the sediment ingestion rates used in the exposure analysis.

10.5.8 Exposure Duration and Habitat Utilization

Most of the wildlife receptors selected for the risk evaluation are not permanent residents of San Diego Bay; they migrate to other areas for at least part of the year. In addition, all of the species have home ranges or foraging ranges substantially larger than the areal extent of the individual assessment units. In combination, these two factors reduce exposure estimates for receptors from default values derived on the basis of the assumption that receptors forage exclusively at a specific assessment unit and do so on a year-round basis (i.e., $F_i = 1$). Foraging behavior of all receptors is discussed below in the context of developing species-specific area use factors.

10.5.8.1 East Pacific Green Turtle

A small population of green turtles resides in southern San Diego Bay, where they congregate in the warm effluent of the San Diego Gas and Electric Company power plant, and feed on eelgrass beds (FWS 1998). They are present in the effluent channel throughout the year, though in lower numbers in the summer, when turtles may disperse into the bay to avoid elevated water temperatures (Dutton and McDonald 1992). There are no reports that sea turtles feed at the NASSCO or Southwest Marine shipyards. The preferred habitat of green turtles in the bay is the shallow subtidal habitat (Table 2-28, Navy and SDUPD 2000). The total area of this habitat in the bay is 3,734 acres (Table 2-3, Navy and SDUPD 2000). The acreage of the assessment units ranges from 21.0 to 58.8 acres (Table 10-7). Although not all the surface area of the assessment units consists of preferred subtidal habitat, as a proportion of the habitats used by sea turtles, the assessment units range between 0.006 and 0.016 of the total surface area. Therefore, rounding up these estimates for purposes of the risk assessment, an area use factor (i.e., F_i) of 0.02 is used for each assessment unit in the food-web exposure model for green turtles.

10.5.8.2 California Least Tern

Atwood and Minsky (1983) found that in southern California, 97 percent of least tern foraging was within 6.4 km of their colony site. Assuming central place foraging from the breeding site, a foraging radius of 6.4 km results in a potential foraging area of about 129 km^2, which is larger than the surface area of San Diego Bay; however, in this risk assessment, it is assumed that terns may forage anywhere in the bay. The habitats in the bay that are used by least terns are deep subtidal, medium subtidal, shallow subtidal, intertidal, salt marsh, and salt works habitats (Table E-4, Navy and SDUPD 2000). The total acreage of these areas is 13,374 acres (calculated from Table 2-3, Navy and SDUPD 2000). The assessment units occupy areas of 21.0–58.8 acres (Table 10-7), which as a proportion of the habitats used by pelicans, range between 0.002 and 0.005 of the total surface area. Therefore, rounding up these estimates for purposes of the risk assessment, an area use factor (i.e., F_i) of 0.01 is used for each assessment unit in the food-web exposure model for least terns.

10.5.8.3 California Brown Pelican

When breeding, brown pelicans generally range as far as 20 km from their breeding site (Briggs et al. 1981). Assuming central place foraging from the breeding site, a foraging radius of 20 km results in a potential foraging area of over 1,250 km^2, which is substantially larger than the entire surface area of San Diego Bay (43 km^2, as stated in Navy and SDUPD 2000). Because pelicans in the bay are generally non-breeders, their potential foraging range may be even larger; however, for purposes of this risk assessment, it is assumed that pelicans may forage anywhere in the bay. The habitats in the bay that are used by brown pelicans are deep subtidal, medium subtidal, shallow subtidal, and salt marsh habitats (Table E-4, Navy and SDUPD 2000). The total acreage of these areas is 11,219 acres (calculated from Table 2-3, Navy and SDUPD 2000). The assessment units occupy areas of 21.0–58.8 acres (Table 10-7). Although not all the surface area of the shipyards consists of preferred habitats, as a proportion of the habitats used by pelicans, the assessment units constitute 0.002 to 0.005 of the total surface area. Therefore, rounding up these estimates for purposes of the risk assessment, an area use factor (i.e., F_i) of 0.01 is used for each assessment unit in the food-web exposure model for brown pelicans.

10.5.8.4 Western Grebe

The habitats in the bay that are used by Western grebes are open water, deep subtidal, medium subtidal, shallow subtidal, and salt marsh habitats (Table E-4, Navy and SDUPD 2000). The total area of these habitats, excluding open water, is 11,219 acres (calculated from Table 2-3, Navy and SDUPD 2000). The assessment units occupy areas of 21.0–58.8 acres, which as a proportion of the habitats used by Western grebes, constitutes 0.002 to 0.005 of the total surface area. Therefore, rounding up the estimates for purposes of the risk assessment, an area use factor (i.e., F_i) of 0.01 is used for each assessment unit in the food-web exposure model for Western grebes.

10.5.8.5 Surf Scoter

The non-breeding home range of the surf scoter has not been estimated. During breeding, the average home range size is 95 ha (235 acres), as estimated by Morrier et al. (1997, as cited in Savard et al. 1998). The non-breeding foraging range is likely even larger because non-breeding birds are not constrained to foraging around a nest site, and it is assumed that wintering surf scoters may forage anywhere in the bay where suitable foraging habitat occurs. The habitats in the bay that are used by surf scoters are open water, deep subtidal, medium subtidal, shallow subtidal, and intertidal habitats (Table E-4, Navy and SDUPD 2000). The total area of these habitats, excluding open water, is 11,375 acres (calculated from Table 2-3, Navy and SDUPD 2000). The assessment units occupy areas of 21.0–58.8 acres (Table 10-7), which as a proportion of the habitats used by scoters, constitute 0.002 to 0.005 of the total surface area. Assuming that scoters forage equally across all suitable habitats, including the shipyards, then for purposes of the risk assessment, an area use factor (i.e., F_i) of 0.01 is used for each assessment unit in the food-web exposure model for surf scoters.

10.5.8.6 California Sea Lion

No literature references were found that reported a foraging range for sea lions, and it is assumed that sea lions may feed throughout the bay. Sea lions feed in open water habitat where they dive to pursue fish or other prey. The total area of non-vegetated subtidal habitat in the bay is 10,396 acres (as calculated from Table 2-3, Navy and SDUPD 2000). The assessment units occupy areas of 21.0–58.8 acres (Table 10-7), which as a proportion of the habitats likely used by sea lions, constitute 0.002 to 0.006 of the total surface area. Assuming that sea lions forage equally across all suitable habitats, including the shipyards, then for purposes of the risk assessment, an area use factor (i.e., F_i) of 0.01 is used for each assessment unit in the food-web exposure model for sea lions.

10.5.9 Bioavailability

Chemical analysis of prey tissue or sediment measures the total concentration of chemicals but not necessarily the amount that is biologically available to receptors, which may be much lower. The assumption used in the food-web exposure model is that a specific chemical in exposure media is as bioavailable as the form used in the toxicity studies on which the TRV is based (i.e., $A_i = 1$), which likely overestimates wildlife exposures to chemicals.

10.6 Effects Characterization for Aquatic-Dependent Wildlife

To evaluate the potential for adverse effects to aquatic-dependent wildlife, receptor-specific exposure estimates were calculated and compared to TRVs. A TRV is a body-weight-normalized daily intake rate of a chemical that, if exceeded, could potentially result in adverse effects to the ecological receptor. In the course of performing ecological risk assessments, Exponent has conducted comprehensive literature searches to identify relevant toxicology papers that contain data suitable for establishing wildlife TRV values. All relevant papers have been reviewed for technical quality, and based on that review, Exponent has selected the most appropriate avian and mammalian TRVs for different chemicals. These TRVs include no-observed-adverse-effects levels (NOAELs) and lowest-observed-adverse-effect-levels (LOAELs). The selection of TRVs was based on criteria associated with relevance of exposure route, duration of exposure, response endpoint, and severity of response. Preference is given to selection of TRVs from studies where chemicals are administered in the diet using a chronic exposure period and where effects are evaluated for ecologically relevant endpoints, such as growth, survival, and reproduction. Table 10-8 presents the TRVs used in the risk assessment. Avian TRVs were used to estimate potential adverse effects to sea turtles as well as aquatic birds because no suitable reptilian TRVs were found in the literature. The following sections describe the studies selected for derivation of TRVs.

10.6.1 Arsenic

The avian TRVs for arsenic were developed from a study by FWS (1964). Mallard ducks were exposed to 34, 86, or 293 mg/kg-day sodium arsenite (57.67 percent As^{3+}) in the diet for 154 days. Mallards exposed to 86 mg/kg-day sodium arsenite (50 mg/kg-day arsenic) had 60 percent mortality, while mallards exposed to 34 mg/kg-day sodium arsenite (20 mg/kg-day arsenic) had 12 percent mortality. The mortalities reported for three control groups were 0, 8, and 31 percent. Because the average mortality in controls (13 percent) was greater than the mortality observed in mallards exposed to 20 mg/kg-day arsenic, this dose was considered a NOAEL TRV, and 50 mg/kg-day arsenic was selected as the LOAEL TRV.

The mammalian TRVs for arsenic were developed from a study by Schroeder and Mitchener (1971). Mice were exposed to 5 ppm arsenite in drinking water and 0.6 ppm in food over three generations. The treatment had no significant effect on either progeny mortality or fertility, but did produce a slight but significant suppression in productivity. Assuming a drinking water intake rate of 0.0075 L/day (based on the scaling function from Calder and Braun 1983) and a food intake rate of 0.0055 kg/day (based on the allometric equation from U.S. EPA 1993a) for a 30-g mouse, 5 ppm arsenite equates to a LOAEL TRV of 1.3 mg/kg-day. No other study was found to provide a suitable no-effects TRV. Therefore, an uncertainty factor of 0.1 was applied to the LOAEL TRV to derive a NOAEL TRV of 0.13 mg/kg-day.

10.6.2 Cadmium

The avian TRV for cadmium was derived from a mallard toxicity study. White and Finley (1978) exposed mallards to cadmium chloride at three dose levels (1.6, 15.2, and 210 ppm in food) for 90 days through reproduction. The test species had a body weight of 1.153 kg and a food consumption rate of 0.110 kg/day. Mallards exposed to the two lower dosages exhibited no adverse effects, but those exposed to 210 ppm cadmium in food produced significantly fewer eggs than did the other groups. Therefore, 15.2 ppm cadmium in food, or 1.5 mg/kg-day, was selected as the NOAEL TRV for birds, and 210 ppm cadmium in food, or 20 mg/kg-day, was selected as the LOAEL TRV.

The mammalian TRV for cadmium was derived from a rat toxicity study by Sutou et al. (1980). The authors exposed rats to cadmium as $CdCl_2$, at four dose levels (0, 0.1, 1.0, and 10 mg/kg-day) by oral gavage through mating and gestation (6 weeks). Adverse reproductive effects (i.e., reduced fetal implantations, reduced fetal survivorship, and increased fetal resorptions) were observed in the rats exposed to 10 mg/kg-day. Therefore, a dose of 1 mg/kg-day was considered to be a chronic NOAEL TRV, and a dose of 10 mg/kg-day was considered to be the LOAEL TRV for the evaluation of risk to mammals.

10.6.3 Chromium

The TRV for chromium toxicity in birds is based on a study by Haseltine et al. (1985, as cited in Sample et al. 1996). This study, which used chromium(III), was selected in preference to a chromium(VI) study because the TRVs derived from this study were lower than TRVs from chromium(VI) studies that reported effects on ecologically relevant endpoints, and thus is considered protective of potential chromium(VI) toxicity in birds. Black ducks were exposed to chromium(III) (as $CrK(SO_4)_2$) at two dose levels (10 and 50 ppm chromium(III)) in food for 10 months, through reproduction. No effects were observed at the lower dose of 10 ppm chromium(III), but duckling survival was reduced by an unspecified amount in the 50 ppm treatment group. Therefore, 10 and 50 ppm chromium(III) were considered to be no-effects and lowest-adverse-effects doses, respectively. Assuming a body weight for the test species of 1.25 kg from Dunning (1993) and a food consumption rate of 0.0785 kg/kg-day derived from Nagy (1987; based on a reasonable maximum energy requirement of 200 kcal/kg-day, an assimilation efficiency of 80 percent, and metabolic energy content of 3,190 kcal/kg dry weight), 10 ppm chromium(III) equates to a NOAEL TRV of 0.86 mg/kg-day, and 50 ppm chromium (III) equates to a LOAEL TRV of 4.3 mg/kg-day.

The NOAEL TRV for the mammalian receptors was developed from an investigation by MacKenzie et al. (1958). In this study, rats were exposed to chromium(VI) (as $K_2Cr_2O_4$) at six dose levels in drinking water (0.45, 2.2, 4.5, 7.7, 11.2, and 25 ppm chromium(VI)) for 1 year. At the end of this period of exposure, changes in liver, kidney, and bone mass were compared between the treatments. No adverse effects were observed at any of the dose levels. Assuming a body weight of 0.35 kg and a water consumption rate of 0.046 L/day for the test species from U.S. EPA (1988a), the maximum dose (25 ppm chromium in water) equates to a NOAEL TRV of 3.3 mg/kg-day.

The LOAEL TRV for exposure to chromium was based on a study by Gross and Heller (1946). Rats were exposed daily to 1,250, 2,500, 5,000, or 10,000 mg/kg potassium chromate (26.78 percent chromium(VI)) in their diets for 3 months. No reproductive effects were reported for rats exposed to 1,250 mg/kg potassium chromate. However, rats treated with 2,500 mg/kg potassium chromate (669.5 mg/kg chromium(VI)) produced “subnormal” offspring (no definition). Based on an average body mass of 0.168 kg estimated from the study, and a food intake rate of 0.0172 kg/day from U.S. EPA (1988a), this dose equates to a LOAEL TRV of 69 mg/kg-day.

10.6.4 Copper

The avian TRVs for copper were derived from a study by Mehring et al. (1960) in which 1-day-old chicks were exposed to copper in the diet for 10 weeks to examine effects on growth and mortality. Eleven dose levels were tested. At doses up to 570 mg/kg, there were no effects on these endpoints, but at the next highest dose (749 mg/kg), higher mortality and lower growth rates were observed. Based on a mean body weight of 0.534 kg for birds at 5-weeks old (mid-range of the exposure interval) and a food ingestion rate of 0.044 kg/day, as calculated from Nagy (1987), a dietary concentration of 570 mg/kg equates to a NOAEL TRV of 47, and a dietary concentration of 749 mg/kg equates to a LOAEL TRV of 62 mg/kg-day.

The mammalian TRVs for copper were developed from a study by Aulerich et al. (1982). The authors exposed mink to copper sulfate at four dose levels (25, 50, 100, and 200 ppm copper supplement, plus 60.5 ppm copper in base feed) for 357 days through reproduction. The assumptions used in TRV calculation included a body weight of 1.0 kg for mink from U.S. EPA (1993a) and a food consumption rate of 0.137 kg/day based on the observations of Bleavins and Aulerich (1981). The 25 ppm supplemental copper dosage (12 mg/kg-day) resulted in no adverse effects, while mortality of mink kits increased at 50, 100, and 200 ppm supplemental copper. Therefore, 12 mg/kg-day copper was considered to be a chronic NOAEL TRV, and 50 ppm copper, or 15 mg/kg-day, was considered to be a chronic LOAEL TRV for mammals.

10.6.5 Lead

The avian TRV was developed from a study by Pattee (1984). American kestrels were exposed to metallic lead in feed at two dose levels (10 and 50 ppm in food) for 7 months, through reproduction. The assumptions used in the TRV calculation included an average body weight of 0.13 kg from the study and a food consumption rate for European kestrels of 0.01 kg/day from Kenaga (1973). No adverse reproductive effects were observed for kestrels treated at the highest dose level (50 ppm), and a NOAEL TRV of 3.9 mg/kg-day was derived from this dietary concentration. No appropriate study was located from which to derive a LOAEL TRV for birds; therefore, the NOAEL TRV was used to evaluate risk to birds from dietary exposure to lead.

The mammalian TRV for lead was developed from a study by Azar et al. (1973) that examined effects on reproductive performance in rats in a three-generational study. Five dose levels were tested (10, 50, 100, 1,000, and 2,000 ppm). The assumptions used in the TRV calculation included a body weight of 0.35 kg and a food consumption rate of 0.028 kg/day calculated from U.S. EPA (1988a). The results of the study showed that none of the lead dose levels affected the number of pregnancies, number of live births, or other reproductive indices. However, the two highest doses reduced offspring weights and produced kidney damage in young. Therefore, the 1,000 ppm dose level (1,130 ppm measured dietary concentration), or 90 mg/kg-day, was considered to be a LOAEL TRV for mammals. The 100 ppm dose level (141 ppm measured dietary concentration), or 11 mg/kg-day, was considered to be a NOAEL TRV.

10.6.6 Mercury

Methylmercury TRVs were adopted to evaluate the potential effects of mercury on aquatic-dependent wildlife. The avian TRV is based on a three-generation study by Heinz (1974, 1976a,b, 1979) using mallards as test organisms. Mallard ducks were exposed to dietary concentrations of methylmercury dicyandiamide ranging from 0.5 to 3.0 mg/kg dry weight for two generations, with the third generation exposed to 0.5 mg/kg-day. The initial test birds (F1) showed no behavioral or reproductive effects at the lowest methylmercury concentration. However, the second-generation ducklings (F2), demonstrated a 29 percent reduction in 1-week survival rates at 0.5 mg/kg methylmercury (Heinz 1976a). There were no significant effects on productivity in either the F1 or F3 birds at this dose level. The impact over the three generations was reported to be a reduction of 18 percent in productivity overall. Assuming a food intake rate of 128 g/kg body weight from Heinz (1979) for the treated F1 and F2 females, the lowest test dose equates to a LOAEL TRV of 0.064 mg/kg-day. No long-term studies were identified as suitable for the derivation of a no-effects level for methylmercury exposure to birds. Therefore, an uncertainty factor of 0.50 was applied to estimate the no-effects TRV of 0.032 to mg/kg-day, as recommended by U.S. EPA (1995b).

The methylmercury TRVs for mammals were based on a study by Verschuuren et al. (1976). In that study, three generations of rats were exposed to methylmercury chloride (79.89 percent Hg) in their diets at doses of 0.1, 0.5, and 2.5 ppm in food. Exposure to 2.5 ppm methylmercury chloride (2.0 ppm Hg) reduced pup viability, but no adverse effects were observed at lower dose levels. Therefore, 0.5 ppm methylmercury chloride (0.4 ppm Hg) was considered to be a no-effects dose, and 2.5 ppm methylmercury chloride was considered to be a lowest-effects dose. Assuming a rat body weight of 0.35 kg and a food ingestion rate of 0.028 kg/day calculated from U.S. EPA (1988a), a dietary concentration of 0.4 ppm Hg equates to a NOAEL TRV of 0.032 mg/kg-day, and a dietary concentration of 2.0 ppm Hg equates to a LOAEL TRV of 0.16 mg/kg-day.

10.6.7 Nickel

The avian TRVs for nickel were derived from a study by Cain and Pafford (1981) in which mallard ducklings were exposed to nickel in the diet for 90 days to determine effects on mortality, growth, and behavior. Three dose levels of nickel were tested: 176, 774, and 1,069 mg/kg. Birds in the highest dose group showed decreased growth and an increased incidence of mortality, but no effects were observed at lower concentrations. Therefore, 774 mg/kg was considered the no-effects dose and 1,069 mg/kg was considered the lowest-effects dose. Based on a mean body weight of 0.782 kg for birds at 45 days old (mid-range of the exposure interval) and a food ingestion rate of 0.078 kg/day, as calculated from Heinz et al. (1989), these doses equate to NOAEL and LOAEL TRVs of 77 and 110 mg/kg-day, respectively.

The mammalian TRVs for nickel were derived from a multi-generational study by Ambrose et al. (1976) in which rats were exposed to nickel in food at doses of 250, 500, or 1,000 mg/kg. The assumptions used in the TRV calculation included a body weight of 0.35 kg and a food ingestion rate of 0.028 kg/day for the test species from U.S. EPA (1988a). No adverse effects were observed at the 500 mg/kg dose level, but reduced offspring weights occurred when rats were treated with 1,000 mg/kg nickel in the diet. Therefore, 500 mg/kg, or 40 mg/kg-day, was selected as the NOAEL TRV, and 1,000 mg/kg, or 80 mg/kg-day, was selected as the LOAEL TRV.

10.6.8 Selenium

The avian TRVs for selenium were based on the results of a study by Heinz et al. (1989). Mallard ducks were fed a range of selenomethionine from 0 to 16 mg/kg-day in the diet for 100 days prior to egg set. An additional treatment of 16 mg/kg-day selenocystine was also included in the study. Reproductive productivity was significantly reduced at 8 ppm with no significant effects noted at 4 ppm. Assuming an average body weight of 1 kg and a food intake rate of 100 g/day from Heinz et al. (1987), the no-effects dose equates to a NOAEL TRV of 0.40 mg/kg-day, and the lowest-effects dose equates to a LOAEL TRV of 0.80 mg/kg-day.

The evaluation of the impact of selenium exposure on the reproduction of mammalian receptors is based on a study by Rosenfeld and Beath (1954). Rats were exposed to three levels of potassium selenate (1.5, 2.5, and 7.5 ppm) in drinking water over two generations. The treatment group exposed to 2.5 ppm showed no significant difference with regard to reproduction or number of young reared. However, the second-generation female progeny of this treatment group did show a 50 percent reduction in the number of young reared. Therefore, the NOAEL TRV was derived from a dose level of 1.5 ppm potassium selenate, and the LOAEL TRV was derived from a dose of 2.5 ppm. Assuming a water intake rate of 0.046 L/day (based on the scaling function of Calder and Braun 1983) and an average body weight of 0.35 kg (U.S. EPA 1988a), a dietary concentration of 1.5 ppm equates to a NOAEL TRV of 0.20 mg/kg-day, and a dietary concentration of 2.5 ppm equates to a LOAEL TRV of 0.33 mg/kg-day.

10.6.9 Zinc

The avian TRV for zinc toxicity is based on a feeding study performed by Stahl et al. (1990). In this study, 24- or 56-week-old white leghorn hens were exposed to zinc sulfate in the diet from 28 (control) to 2,000 mg/kg in a dehydrated corn and soybean meal diet. After continuous daily exposure to 68 weeks of age, no significant differences were noted in hen weight, feed consumed, egg production, egg fertility, egg hatchability, or progeny growth rates. The authors concluded: "The zinc treatments have no effect on hen performance or reproductive performance." Therefore, based on a no-effects dietary concentration of 2,000 mg/kg, and a measured intake rate of 0.065 kg/kg body weight-day, a NOAEL TRV of 130 mg/kg-day was derived from this study. None of the studies reviewed was appropriate for deriving a LOAEL TRV; therefore, the risk to birds from dietary exposure to zinc was evaluated using a NOAEL TRV.

The TRV used to evaluate risks from zinc exposure in mammals was developed from a study by Schlicker and Cox (1968). In this investigation, adult female Sprague-Dawley rats were exposed to 2,000 and 4,000 mg/kg dry weight zinc oxide in their diets. Exposure commenced 21 days prior to mating and continued throughout gestation. Females exposed to 4,000 ppm showed measured increases in fetal reabsorption. No effect on reproduction (measured as percent reabsorption or difference in rate of fetal growth) was observed at 2,000 ppm. Therefore, 2,000 ppm and 4,000 ppm were considered to be no-effects and lowest-effects dose levels, respectively. Assuming a body mass of 0.35 kg and a food ingestion rate of 0.028 kg/day calculated from U.S. EPA (1988a), a dietary concentration of 2,000 ppm corresponds to a NOAEL TRV of 160 mg/kg-day, and a dietary concentration of 4,000 ppm corresponds to a LOAEL TRV of 320 mg/kg-day.

10.6.10 Polychlorinated Biphenyls

The avian no-effects TRV was developed from a study by McLane and Hughes (1980). In this study, owls were administered 3 mg/kg wet weight Aroclor® 1248 in their diets continuously, with the exception of 60 days during brooding. There was no difference in egg production, eggshell thickness, hatching, or fledgling rates monitored over two breeding seasons between the controls and the PCB-treated birds. Therefore, based on an intake rate of 0.137 kg wet weight/kg body weight from Pattee et al. (1988), this dose equates to a NOAEL TRV of 0.41 mg/kg-day.

The avian LOAEL TRV for total PCBs was developed from a 4-month study by Dahlgren et al. (1972) that examined the effects of Aroclor® 1254 on pheasants. Comparative reproductive studies indicate there is no difference in the reproductive effects of Aroclor® 1254 and Aroclor® 1248 on birds (Lillie et al. 1974; U.S. EPA 1995b). Ring-neck pheasants were dosed once a week with either 12.5 or 50 mg/bird Aroclor® 1254. This weekly dose equated to a daily dose of 1.80 and 7.14 mg/kg-day, respectively, based on a uniform distribution of the dose over 7 days, and an average body mass of 1.0 kg from U.S. EPA (1993a). No impact on chick growth, egg production, or survivability was reported at 1.8 mg/kg-day. However, there was a 5 percent reduction in fertile egg hatchability at this dose. Therefore, 1.8 mg/kg-day was considered the LOAEL TRV.

The mammalian TRVs for total PCBs were based on a study by Aulerich and Ringer (1977). This study consisted of three separate but related investigations. In the first investigation, the reproductive effects of the various Aroclors® on mink were examined. Mink were fed 0, 5, and

10 mg/kg wet weight Aroclor® 1254 in the diet over a 9-month period. All of the females in the treatment groups failed to reproduce. In the second investigation, a cohort of mink was exposed to 0, 1, 5, and 15 mg/kg wet weight Aroclor® 1254 and reproductive success was measured with regard to jill productivity and kit survival. No significant reproductive effects were observed at 1 ppm. In the third study, mink were exposed to 2 ppm of Aroclors® 1016, 1221, 1242, and 1254 over gestation. This study reported a 93 percent reduction in whelped kits per female mated as well as reductions in the average birthweight of kits from females treated with Aroclor® 1254. Based on these results, dietary concentrations of 1 ppm and 2 ppm were deemed to be no-effects and lowest-effects dose levels, respectively. Assuming a body weight of 1 kg from U.S. EPA (1993a) and a food consumption rate of 0.137 kg/day based on the observations of Bleavins and Aulerich (1981), a dietary concentration of 1 ppm equates to a NOAEL TRV of 0.14 mg/kg-day, and a dietary concentration of 2 ppm equates to a LOAEL TRV of 0.27 mg/kg-day.

10.6.11 Polycyclic Aromatic Hydrocarbons

The avian TRVs for total PAHs were derived from a study of benzo[a]pyrene toxicity in pigeons conducted by Hough et al. (1993). Pigeons received weekly intramuscular injections of 10 mg/kg body weight benzo[a]pyrene, or 1.4 mg/kg-day, for 3 to 5 months. Female pigeons treated at this dose level were completely infertile and had gross alterations in ovarian structure. Therefore, 1.4 mg/kg-day represents the LOAEL TRV for birds. No other appropriate study was found to provide a no-effects TRV. Therefore, an uncertainty factor of 0.1 was applied to the LOAEL TRV to derive a NOAEL TRV of 0.14 mg/kg-day.

The evaluation of PAH toxicity to mammals is based on a study by Mackenzie and Angevine (1981) that examined the reproductive effects of benzo[a]pyrene on mice. Female CD-1 mice were exposed to benzo[a]pyrene at doses ranging from 10 to 160 mg/kg-day through daily intubation. Treatment commenced on Day 7 after the best estimated time of conception and continued through Day 16 of gestation. Mean pup weight was observed to be significantly reduced in the 10 mg/kg-day treatment group. This treatment was therefore considered to be applicable as a LOAEL TRV (10 mg/kg-day). A LOAEL to NOAEL uncertainty factor of 0.1 was applied to derive a NOAEL TRV of 1.0 mg/kg-day.

10.6.12 Tributyltin

The avian TRVs for TBT are based on a study by Schlatterer et al. (1993) that examined effects on reproduction in Japanese quail during a 6-week chronic exposure study. TBT (as bis-tributyltin oxide) was administered in the diet at four dose levels: 24, 60, 150, and 375 mg/kg. No consistent adverse effects on reproduction were seen for quail consuming food containing TBT at 60 mg/kg, but egg weight and hatchability were reduced among quail consuming 150 mg/kg in the diet. Based on a body weight of 0.15 kg from Vos et al. (1971) and a food ingestion rate of 0.0169 kg/day calculated from Nagy (1987), a dietary concentration of 60 mg/kg equates to a NOAEL TRV of 6.8 mg/kg-day, and a dietary concentration of 150 mg/kg equates to a LOAEL TRV of 17 mg/kg-day.

The mammalian TRVs for TBT are based on a study by Davis et al. (1987) that examined effects on reproduction in mice exposed via oral intubation during gestation. Six dose levels were used: 1, 2, 3.5, 5.8, 11.7, 23.4, and 35 mg/kg-day. At the highest dose (35 mg/kg-day), mice showed increased signs of litter resorption, and their offspring showed reduced survival and fetal weight. These effects were not seen at lower doses. Therefore, 35 mg/kg-day represents the LOAEL TRV and 23 mg/kg-day represents the NOAEL TRV.

10.7 Risk Characterization

10.7.1 Potential for Adverse Effects to Aquatic-Dependent Wildlife

Risks of adverse ecological effects to reptile, bird, and mammal communities were estimated by integrating the exposure and effects assessments using the hazard quotient approach:

$$HQ = \frac{IR_{chemical}}{TRV} \qquad (2)$$

where:

HQ	=	hazard quotient (unitless)
$IR_{chemical}$	=	total ingestion rate of the chemical (mg/kg body weight-day)
TRV	=	toxicity reference value (mg/kg body weight-day).

The modeled rate of exposure ($IR_{chemical}$) was based on the mean chemical concentrations in prey and sediment for each assessment unit, and exposure was modeled using ecologically realistic area-use factors for exposed receptors. A TRV represents the hypothetical threshold of exposure to a chemical that would not induce any adverse toxicological effect in an individual (NOAEL TRV) or the lowest chemical exposure at which toxicological effects would first be observed (LOAEL TRV). Risk is expressed as a hazard quotient, which is the ratio of the exposure to the TRV.

A hazard quotient below 1.0 indicates that the chemical is unlikely to cause adverse ecological effects. A hazard quotient above 1.0 indicates that the exposure of the modeled receptor has exceeded the TRV, which could indicate that there is a potential that some fraction of the population may experience an adverse health effect as the direct result of the presence of the chemical. However, uncertainties associated with exposure and toxicity assumptions must be weighted in the evaluation and interpretation of hazard quotients (U.S. EPA 1997a), as discussed below in the uncertainty evaluation. The following sections discuss the estimated risks to aquatic-dependent wildlife receptors foraging in each assessment unit. The ecological significance of these food-web model results in discussed below in Section 10.8.

10.7.2 Risks to Aquatic-Dependent Wildlife at the Reference Areas

Risks to aquatic-dependent wildlife were calculated using data for both the original and revised reference areas, as shown in Tables 10-9 and 10-10, respectively. The calculated hazard quotients are virtually identical in both cases, because the same prey species data were used to estimate dietary exposure in both cases, and consumption of chemicals in food is the primary exposure route for wildlife receptors. Under both scenarios, exposure to chemicals in the reference area is unlikely to result in adverse effects to brown pelican, least tern, western grebe, surf scoter, sea lion, or green turtle populations, because hazard quotients are less than 1.0 for all chemicals evaluated, whether using NOAEL- or LOAEL-based TRVs.

10.7.3 Risks to Aquatic-Dependent Wildlife Inside the NASSCO Leasehold

Exposure to chemicals inside the NASSCO leasehold is unlikely to result in adverse effects to brown pelican, least tern, western grebe, surf scoter, sea lion or green turtle populations. Hazard quotients are substantially less than 1.0 for all chemicals evaluated for these receptors, whether using NOAEL- or LOAEL-based TRVs (Table 10-11). For all of the receptors and chemicals evaluated except mercury, hazard quotients are less than 1×10^{-2}. The NOAEL mercury hazard quotient for brown pelican is 1.6×10^{-2}.

10.7.4 Risks to Aquatic-Dependent Wildlife Outside the NASSCO Leasehold

Exposure to chemicals outside the NASSCO leasehold is unlikely to result in adverse effects to brown pelican, least tern, western grebe, or sea lion populations. Hazard quotients are less than 1.0 for all chemicals for these receptors, whether using NOAEL- or LOAEL-based TRVs (Table 10-12). Risks to surf scoter and sea turtle were not evaluated for this assessment unit, because eelgrass and mussel samples were not collected outside the NASSCO leasehold. For all of the receptors and chemicals evaluated except mercury, hazard quotients are less than 1×10^{-2}. The NOAEL mercury hazard quotient for brown pelican is 1.4×10^{-2}.

10.7.5 Risks to Aquatic-Dependent Wildlife Inside the Southwest Marine Leasehold

Exposure to chemicals inside the Southwest Marine leasehold is unlikely to result in adverse effects to brown pelican, least tern, western grebe, surf scoter, sea lion, or green turtle populations. Hazard quotients are less than 1.0 for all chemicals for these receptors, whether using NOAEL- or LOAEL-based TRVs (Table 10-13). For all of the receptors and chemicals evaluated except mercury, hazard quotients are less than 1×10^{-2}. The NOAEL mercury hazard quotient for brown pelican is 1.3×10^{-2}.

10.7.6 Risks to Aquatic-Dependent Wildlife Outside the Southwest Marine Leasehold

Exposure to chemicals outside the Southwest Marine leasehold is unlikely to result in adverse effects to brown pelican, least tern, western grebe, or sea lion populations. Hazard quotients are less than 1.0 for all chemicals for these receptors, whether using NOAEL- or LOAEL-based TRVs (Table 10-14). Risks to surf scoter and sea turtle were not evaluated for this assessment unit, because eelgrass and mussel samples were not collected outside the Southwest Marine leasehold. For all of the receptors and chemicals evaluated except mercury, hazard quotients are less than 1×10^{-2}. The NOAEL mercury hazard quotient for brown pelican is 1.4×10^{-2}.

10.8 Uncertainties Related to Risk Estimates for Aquatic-Dependent Wildlife

Within any step of the ecological risk assessment process, assumptions must be made on the basis of professional judgment in the absence of concise scientific data. Incorporating assumptions in the components of the risk analysis results in a degree of uncertainty surrounding the risk conclusions. The following sections discuss the uncertainties associated with the model approach, site-specific data collected in support of the risk assessment, and with other parameters used in predicting receptor response and behavior.

10.8.1 Model Uncertainty

The exposure estimates for aquatic-dependent wildlife were based on deterministic models that incorporated site-specific chemical concentrations in prey and media with assumptions about the life history characteristics of the receptor species. The risk estimate was based on the ratio of a point estimate of exposure to a literature-based threshold response (i.e., NOAEL or LOAEL), rather than on actual site-specific field data on the populations of these wildlife species. As a result, there are uncertainties associated with the applicability of the resulting hazard quotients in identifying actual effects or risks to the local populations of these wildlife receptors. These uncertainties can stem from numerous factors, including 1) use of assumptions regarding receptor-specific exposure parameters such as body weights, home range size, migratory patterns, prey selection, and ingestion rates; and 2) use of literature-based TRVs derived from laboratory studies, often for species other than those being evaluated. Further analysis of the uncertainties associated with these factors is discussed below.

10.8.2 Parameter Uncertainty

Two classes of parameters were used in the food-web models to evaluate the risk associated with exposure to chemicals at the shipyards. First, site-specific samples of sediment, eelgrass, mussels, and fish collected within the assessment units were analyzed for chemical concentrations to characterize the receptors' exposure. Second, receptor-specific behavioral factors were used as parameters in the food-web model. These parameters were used to best characterize receptor utilization of specific assessment units relative to other surrounding resources. Uncertainty in chemical concentrations and the modeling of receptor behavior is discussed below.

10.8.2.1 Uncertainty in Estimating Chemical Concentrations in Prey Sources

Receptors evaluated in this risk assessment have varied diets that incorporate a range of prey species. For example, Thompson et al. (1997) note that the least tern probably consumes any small surface-swimming, nonspiny fish 2.0–9.0 cm long. However, for this risk evaluation, receptor diet composition was restricted to one or a few ecologically similar species to simplify sample collection and to standardize analytical results across assessment units. Dependent on the extent that uptake of chemicals in species selected for analysis is similar to uptake for other forage species in the diet of selected receptors, dietary exposures using analytical data for the selected species may over- or underestimate the exposure of a receptor consuming a more diverse diet. The magnitude of this uncertainty cannot be quantified with available site-specific data. However, variability between the chemical concentrations in the modeled diet and the actual diet is minimized by selecting prey species that are representative of the broader diet in terms of exposure to chemicals. For example, least terns feed primarily on surface-dwelling fish that have limited exposure to sediment-bound chemicals, and forage species collected for analysis (northern anchovy and topsmelt) have these characteristics. Similarly, sea lions may feed extensively on bottom-dwelling fish with greater exposure to sediment-bound chemicals and the spotted sand bass is representative of these fish species.

In order to provide an adequate sample size for analytical purposes, it was necessary to composite mussel or forage fish samples collected within an assessment unit. While this procedure has the effect of reducing the contribution of the most highly contaminated food items in the exposure assessment, it also reflects the typical dose received by a receptor that integrates foraging across the prey chemical distribution range, and is therefore a realistic estimate of actual exposure.

Mussel and eelgrass samples were not collected outside the NASSCO or Southwest Marine leaseholds and potential for adverse effects to receptors consuming these food sources (i.e., surf scoter and green turtle) could not be evaluated for these areas. However, physical conditions suggest that these food sources are less likely to occur outside the shipyard leaseholds as opposed to inside. Water depth outside the leasehold is likely too deep to permit growth of eelgrass due to insufficient penetration of sunlight. For these reasons, it is unlikely that absence of mussel or eelgrass data from outside the leaseholds constitutes a major uncertainty in this risk assessment.

Food-web exposure models estimate intake based on mean chemical concentrations in food items and incidentally ingested sediment. This approach integrates spatial variation in chemical concentrations across the assessment unit, and is representative of the exposure received by a receptor utilizing the entire assessment unit while foraging for prey. Receptors may ingest prey with maximum measured chemical concentrations at times during foraging, but are not likely to exclusively consume such prey. However, for the uncertainty assessment, food-web exposure models were reanalyzed using maximum chemical concentrations in food and sediment, or one-half maximum quantitation limit in the case of undetected compounds, to represent hypothetical exposure that receptors would receive if they ingested only these items during that part of their foraging time spent within an assessment unit. Results are presented in Tables 10-15 through 10-20. For all chemicals evaluated, and at all assessment units, hazard quotients are still well below 1.0, indicating that there would be no unacceptable risk to receptors in the hypothetical case where they consumed only the maximally exposed prey.

10.8.2.2 Uncertainty in Estimating Ingestion Rates

Estimates of food ingestion rates for avian and mammalian receptors were derived using bioenergetic allometric scaling functions (Nagy et al. 1999; Nagy 1987; U.S. EPA 1993a). These functions relate field metabolic rates to body mass across receptors within a given class (birds or mammals) and therefore are not receptor-specific. Incidental sediment ingestion was estimated as a percentage of food ingestion, because there is a lack of receptor-specific data pertaining to actual sediment ingestion rates. The predictions applied were estimated from soil ingestion rates reported by Beyer et al. (1994) or were based on best professional judgment and consequently are very uncertain. However, conservative estimates were used such that lack of knowledge on incidental ingestion of sediment resulted in an overestimation of risk.

10.8.2.3 Uncertainty in Temporal (Migration) Parameters

All of the avian receptors considered in this risk assessment are migratory in nature; least terns nest in San Diego Bay and are present only during the breeding season, while brown pelicans, surf scoters, and Western grebes are common winter residents of the bay but migrate or disperse away from the bay to breed. Sea lions also breed away from San Diego Bay in offshore rookeries. Hence, each of these receptors would only be exposed to chemicals at the shipyards during a portion of the year. However, temporal variations in chemical exposure were disregarded in the risk analysis, and each receptor was assumed to forage year-round within the bay. This approach incorporates a level of conservatism into the exposure assessments, resulting in an overestimation of risks to migratory wildlife, although risk estimates using this approach are more representative for other species that are permanent residents within San Diego Bay.

10.8.2.4 Uncertainty in Spatial (Foraging) Parameters

In the risk analysis, all receptors were assumed to obtain only a portion of their diet from within the assessment units because their foraging ranges are spatially broader than the area of the shipyards. Estimates of receptor foraging ranges were based on site-specific data on the preferred habitats of receptors and the distribution of those habitats within San Diego Bay, and information from the scientific literature on the foraging ranges of the receptors. Based on this combination of factors, it is unlikely that any receptor would forage preferentially within an assessment unit during the time it is resident in the bay because the units do not provide the preferred habitat or are too small to meet the receptors' foraging requirements. Consequently, food-web models used area use factors that realistically reflect the size of the assessment unit relative to the size of the foraging range of the receptor.

However, for purposes of the uncertainty assessment, food-web exposure models were reanalyzed to determine the hypothetical exposure that receptors would receive if they foraged exclusively within any individual assessment unit. This scenario represents the maximum, but highly unlikely, exposure that a receptor could receive if it were essentially confined to the immediate vicinity of the shipyards and obtained its entire food supply from that area. Results are presented in Tables 10-21 through 10-26. Even under this extreme exposure scenario, hazard quotients are below 1.0 for all chemicals in all assessment units, except for two cases. Hazard quotients for brown pelicans exposed to mercury ranged between 1.0 and 2.0 in all assessment units and the reference area when exposure is compared to the NOAEL-based TRV, but not when compared to the LOAEL-based TRV. The hazard quotient for surf scoter exposure to total PAHs was greater than 1.0 inside the Southwest Marine leasehold when exposure is compared to the NOAEL-based TRV, but not when compared to the LOAEL-based

TRV. In both cases, although the results suggest that unacceptable risk may exist for these receptors, comparison with the LOAEL-based TRVs indicates that adverse effects would not be predicted to occur at a known level for toxic effects. Furthermore, in the case of mercury, the exposure models indicate that background risk (for pelicans foraging in the reference area) is already elevated, although this is again likely due to approach used to estimate exposure. If the background risk from mercury is factored out (by subtracting reference area hazard quotients from assessment unit hazard quotients), then incremental risk associated with exclusive foraging in the assessment units would not be expected to result in adverse effects, because the resultant NOAEL hazard quotients would be less than 1.0. In addition, for both chemicals, hazard quotients are less than 1.0 when exposure is compared to the LOAEL-based TRV. Because the LOAEL represents the dose at which effects would first be expected to occur, these results indicate that the likelihood of adverse effects occurring to pelicans or surf scoters is minimal, even under the highly unlikely scenario where these species would be confined to the immediate vicinity of the shipyards.

10.8.2.5 Uncertainty in Relative Bioavailability

Another factor affecting the uncertainty of chemical exposure estimates is the assumption regarding relative bioavailability. In both the exposure and response models, it was assumed that the form of the chemical present in the environment was absorbed with the same efficiency as the chemical form used in the laboratory toxicity study. Chemical solubility is an important factor in absorption efficiency, and for many chemicals, laboratory toxicity studies are performed using the most soluble form. This is particularly true of the inorganic chemicals, which are themselves natural but partly bound constituents of abiotic media such as sediments and, therefore, not entirely available to biota. The assumption that the concentrations measured from matrices that have undergone strong acid digestion would represent that fraction available for absorption by animals is highly uncertain, and in the case of metals, highly conservative.

10.8.3 Uncertainty Associated with Toxicity Reference Values

The range of toxicity thresholds reported in the literature may be very large, even among those studies deemed suitable for extrapolation to receptor species of interest. Observational errors in conducting toxicological experiments from which a TRV is derived stem primarily from parameter uncertainty. Uncertainty in TRV extrapolation, which may arise due to suspected differences in physiological responses of organisms to chemical exposures under identical conditions, is the result of model uncertainty.

In this risk assessment, where possible, modeled exposures were compared directly with best available NOAELs and LOAELs derived from the literature. Analysis of the available literature provided no reason to assume that receptors evaluated in this investigation would be more sensitive to chemicals than those tested in the respective toxicity studies cited. Therefore, any variance in the sensitivity of the receptor relative to the test species used to develop the TRV would most likely be evenly distributed around the estimated NOAEL or LOAEL and no interspecies uncertainty factors were applied.

For reptiles, no appropriate TRVs could be found for any of the chemicals being evaluated. This data gap is due to the paucity of studies in the literature that have examined adverse effects in reptiles following dietary exposure to contaminants, and is a problem in any risk assessment where reptilian species are included as a receptor. For this risk assessment, avian TRVs were used as surrogate values for evaluating risk to reptiles solely on the basis that birds are more taxonomically similar to reptiles than are mammals. However, the relative sensitivity of birds and reptiles to toxic effects of the chemicals being evaluated is unknown, and risk conclusions for reptiles based on avian effects data may underestimate or overestimate actual risk to marine reptiles occurring in the assessment units.

Some of the chemicals under consideration in this risk assessment are found in multiple chemical forms in the environment. These chemical species may have different rates of exposure and toxicity. The remainder of this section discusses the considerations associated with speciation/composition and describes how they were reconciled.

10.8.3.1 Mercury

When metallic or ionic mercury is released into an aerobic environment, either terrestrial or aquatic, it will tend to accumulate in the Hg(II) oxidation state (Kabata-Pendias and Pendias 1992). When introduced within an anaerobic environment, a portion of it may become methylated, or converted from inorganic mercury to organic methylmercury. This is a biotic process and is limited almost exclusively to sulfur-reducing bacteria, present in aquatic sediments and active under anaerobic conditions (Huckabee et al. 1979). Within anaerobic sediments and the overlying anoxic water column, methylmercury may constitute a small but significant percentage of the total mercury load. Unlike inorganic and metallic mercury, methylmercury is highly bioavailable and tends to bioaccumulate; it can represent the majority of mercury in prey tissues. It is more toxic to wildlife than inorganic forms of mercury.

In this risk assessment, receptors were assessed based on exposure to total mercury concentrations measured in food and sediments. However, dietary exposures to total mercury were evaluated against TRVs based on exposure to methylmercury. Thus, all mercury detected in food and sediments at the shipyards was assumed to be in the more toxic methylated form. This assumption was highly conservative, particularly for mercury in sediments. Assessments of apparent RPD depth and methane in the shipyard sediments indicate that surface sediments are not anoxic, and therefore are not likely to be generating or releasing methylmercury at the sediment surface.

10.8.3.2 Total PCBs

PCBs were analyzed as specific Aroclors®. Because Aroclors® are mixtures of PCB congeners and are subject to chemical weathering that changes their composition, individual Aroclor® measurements are not necessarily consistent from study to study. Therefore, the risk from Aroclors® was evaluated based on total PCB concentrations. Total PCB was computed as the sum of all detected Aroclors® within each sample, or one-half of the highest quantitation limit for any Aroclor® if no Aroclor® was detected in the sample. The following Aroclors® were included in the PCB sum: 1016, 1221, 1232, 1242, 1248, 1254, and 1260. Dietary exposure to PCBs was evaluated against TRVs developed from studies that tested individual Aroclors®

using several of the more potent Aroclors®, specifically 1248 or 1254. To the extent that less potent Aroclors® constitute a significant proportion of the total PCB content, such as in the case of forage fish and spotted sand bass where Aroclor® 1260 was detected in all samples, this approach represents a conservative estimate of the potential toxicity resulting from exposure of receptors to PCBs.

10.8.3.3 Polycyclic Aromatic Hydrocarbons

The availability of toxicity data on individual PAHs, particularly with regard to effects on ecologically relevant endpoints such as reproduction, is extremely limited. Therefore, exposure to PAHs was quantified based upon total PAH concentrations. Total PAH was computed as the sum of the concentrations of the following compounds: 2-methylnaphthalene, acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, phenanthrene, benz[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[j]fluoranthene, benzo[ghi]perylene, benzo[k]fluoranthene, chrysene, fluoranthene, indeno[1,2,3-cd]pyrene, and pyrene. Total PAH concentrations were compared to TRVs developed from studies where animals were only exposed to benzo[a]pyrene. Because benzo[a]pyrene is among the more potent PAHs, comparison of total PAH concentrations to a compound-specific TRV represents a conservative estimate of the potential toxicity resulting from exposure of receptors to PAHs.

10.9 Interpretation of Ecological Significance

Aquatic-dependent wildlife was modeled using conservative, ecologically relevant exposure assumptions to develop representative estimates of risk to receptors foraging near the shipyards. Exposure models indicate that no exposure estimates, for any chemical, exceed either no-effect (i.e., NOAEL-based) or lowest effects (i.e., LOAEL-based) TRVs for any receptor at any of the assessment units. Even under hypothetical, but ecologically unlikely, scenarios that maximize exposure by assuming receptors forage exclusively within an assessment unit, the likelihood of adverse effects is minimal, especially when considering uncertainty associated with exposure estimates and effects thresholds used in the exposure models. Overall, the results of this risk evaluation indicate that chemical concentrations measured in prey and sediment of the

NASSCO and Southwest Marine leaseholds are very unlikely to constitute an unacceptable risk to populations of aquatic-dependent wildlife potentially foraging at these locations. Therefore, the current conditions at the shipyards are protective of beneficial uses associated with aquatic-dependent wildlife.

11 Human Health Risk Assessment

To evaluate potential impairment of the human health beneficial use at the shipyards, a human health risk assessment was conducted to address potential risks associated with chemicals in sediments. The method used to assess human health risks is based on the framework described in the *Guidelines for Assessment and Remediation of Contaminated Sediments in the San Diego Bay at NASSCO and Southwest Marine Shipyards* (RWQCB 2001), and is consistent with California state and EPA guidance (OEHHA 1999; RWQCB 2000; U.S. EPA 1989a,b, 2000d,e).

The following sections describe the method used to conduct the human health risk assessment and provide the results of that assessment.

11.1 Site Setting

As indicated in the conceptual model (Figure 1-3), and as recognized by RWQCB (2001), the most significant potential source of human exposure at the site is through consumption of fish and shellfish that may have bioaccumulated chemicals either directly from site sediments or through the food web. The industrial nature of the site, the lack of a beach, and the absence of public access makes swimming or wading a highly unlikely event. Therefore, consistent with RWQCB guidelines, only the fish and shellfish ingestion pathways were evaluated. Due to access restrictions, fishing and shellfish gathering are also highly unlikely events at the shipyards. Therefore, any exposure estimates developed as part of this risk assessment represent highly conservative estimates and any actual human exposure and risks at the shipyards are considerably lower than the estimates derived herein.

11.2 Screening of Chemicals for Human Health

Prior to conducting the human health risk assessment, chemical concentrations in fish and shellfish (Appendix E) were compared with screening concentrations to identify chemicals in fish or shellfish present at concentrations that warrant further evaluation. This screening step involves the use of highly conservative exposure and risk assumptions and is intended to eliminate any substances from further consideration that clearly have no potential to affect beneficial uses at the site. Any substances that are not screened out at this step are carried through a detailed human health risk assessment that involves site-specific considerations of potential human exposure. In the initial screening, two types of screening levels were applied. First, chemical concentrations in seafood were compared with generic health-protective tissue residue guidelines (TRGs) provided by OEHHA (1999) or derived by using the same method as the Office of Environmental Health Hazard Assessment (OEHHA). Second, because many of the chemicals detected at the site are widely detected in seafood worldwide (e.g., mercury and PCBs), chemical concentrations in seafood were compared with concentrations in aquatic biota from reference sites. The TRGs, which were submitted to RWQCB prior to use in the investigation, are described in more detail below. Only substances exceeding both screening steps were retained for further evaluation.

The general model for deriving health risk-based screening values such as TRGs differs slightly for carcinogenic and noncarcinogenic chemicals because of differences in how the toxicity factors are expressed. For noncarcinogenic chemicals, health-based TRGs were derived using the following algorithm:

$$\mathrm{TRG} = \frac{\mathrm{RfD} \times \mathrm{BW}}{\mathrm{CR} \times \mathrm{FI}}$$

where:

TRG	=	tissue screening level for fish and/or shellfish tissue (μg/kg)
RfD	=	reference dose (mg/kg-day)
BW	=	body weight (70 kg adult in OEHHA's TRGs)
CR	=	fish and shellfish consumption rate (21 g/day in OEHHA's TRGs)
FI	=	fractional intake of seafood consumed that originates from site(unitless, 1.0 in OEHHA's TRGs).

For carcinogenic chemicals, health-based TRGs were derived as follows:

$$TRG = \frac{TRL \times BW}{CSF \times CR \times FI \times ABS}$$

where:

TRL = target risk level (unitless, 10^{-5} in OEHHA's TRGs)
CSF = carcinogenic slope factor $(mg/kg\text{-}day)^{-1}$
ABS = fraction absorbed.

TRG, BW, CR, and FI are as defined for noncarcinogenic TRGs.

When available, TRGs developed by OEHHA (1999) were used to screen chemicals. For those chemicals that do not have TRGs published by OEHHA but do have EPA-derived toxicity factors available, TRGs were derived using the same default assumptions used by OEHHA. Human health TRGs are presented in Table 11-1. The toxicity factors (i.e., reference doses [RfDs] and carcinogenic slope factors [CSFs], as described in the *Toxicity Assessment* section) used to calculate those TRGs are presented in Table 11-2.

The exposure variables used in the OEHHA (1999) TRGs include several assumptions that greatly overestimate any potential exposure of humans at the shipyards. Specifically, individuals are assumed to obtain their entire daily amount of seafood (21 g/day) from the shipyards for their lifetime. This is equivalent to an angler obtaining about one meal of seafood per week from the shipyards for an entire lifetime. As discussed further below in the *Exposure Assessment* section, this assumption is not possible given the access restrictions at the shipyards and the absence of fishing opportunities. Nevertheless, such a highly conservative assumption may be appropriate for use at the screening-level step to eliminate substances from further consideration that clearly pose no threat to beneficial uses associated with human health.

As a further conservative assumption, the maximum chemical concentrations from fish (i.e., spotted sand bass) fillets and the edible portions of lobster were compared to TRGs (Table 11-3). For arsenic, the concentration detected in fish and lobster tissue was multiplied by a factor of 0.04 to account for the relatively small percentage of total arsenic in seafood that is in the inorganic form. The rationale for this assumption is described in the uncertainty section.

In fish tissue, the maximum PCB concentration of 400 μg/kg exceeded the TRG. In lobster, the maximum concentrations of mercury (521 μg/kg) and PCBs (21 μg/kg) exceeded their TRGs of 300 μg/kg and 20 μg/kg, respectively. All other chemicals detected in fish or shellfish (i.e., other metals and PAHs) were present at concentrations lower than the highly health-protective TRGs used in this screening. This indicates that these chemicals in fish and shellfish would not cause unacceptable health risks even if the seafood items were entirely harvested from the shipyards and then consumed at the 21g/day rate over a lifetime.

Chemicals present at concentrations that exceeded TRGs were compared to chemical concentrations in fish and lobster from reference areas. In this screening step, samples were segregated by location (i.e., inside the NASSCO leasehold, outside the NASSCO leasehold, inside the Southwest Marine leasehold, and outside the Southwest Marine leasehold). This was done for two reasons: 1) chemical concentrations differ at the two shipyards because of the different activities carried out over the years of operation, and 2) differences in access restrictions inside vs. outside the leaseholds, in addition to differences in the types of fishing that could occur (from piers/shoreline vs. boat access) and the relative size of the four areas will affect the amount of seafood that could potentially be consumed from each area. Following this second screening step (Table 11-4), the following chemicals were selected for the risk assessment:

- **Inside NASSCO Leasehold—**Mercury in edible lobster tissue was retained for evaluation.
- **Outside NASSCO Leasehold—**No chemicals were retained. PCBs in only one fish sample (57 μg/kg) slightly exceeded the maximum reference concentration (55 μg/kg). The 95 percent UCL (54 μg/kg), as described in

the exposure assessment, was within the reference range and all other sample concentrations fell within the reference range.

- **Inside Southwest Marine Leasehold—**PCBs in fish fillets and edible lobster tissue were retained for evaluation.

- **Outside Southwest Marine Leasehold—**PCBs in fish fillets was retained for evaluation.

11.3 Exposure Assessment

Exposure assessment is the process of identifying human populations that could potentially contact site-related chemicals and estimating the magnitude, frequency, duration, and route(s) of potential exposures. An exposure pathway describes a chemical's transport from its source to a potentially exposed individual and must include a source, transport mechanism, receptor, and point of entry into the body. Only when each of these elements is present can an exposure pathway be complete, and only complete exposure pathways have the potential to result in a health risk. Potential exposures associated with the chemicals retained after screening are evaluated by identifying current and potential future uses of the property, those populations that could be exposed to the chemicals (i.e., the receptors), and the manner in which they may be exposed (i.e., the exposure pathway). The applicable exposure pathways (i.e., human consumption of fish and shellfish from the site) are described in the conceptual site model and site setting sections, above. This section begins with a discussion of potential human receptors and then describes the basis for assumptions used in quantifying exposure.

11.3.1 Potential Human Receptors

The assumed receptors in this assessment are individuals who fish in the waters in and around the shipyards. Because of the current and future activities associated with defense department work, access to the site is highly restricted. Armed personnel are present at all times to ensure that no trespassing, including fishing, occurs at the site. There is no possible access by land to the site and a physical barrier is present to restrict access by boat to the leasehold. Future use of the site is expected to remain the same. Thus, in actuality, the likelihood of recreational or subsistence anglers utilizing the shipyard leaseholds is nonexistent. Furthermore, the potential for anglers in boats to use areas beyond the leaseholds is extremely low given the industrial nature of the site and the proximity to more attractive fishing areas. Nevertheless, the human health risk assessment was conducted assuming that fishing and shellfish collection could occur at the site and that the applicable receptors are recreational and subsistence anglers.

11.3.2 Quantification of Exposures

Consistent with U.S. EPA (1989a) guidance, human exposure to chemicals in fish and shellfish was calculated by combining estimates of fish and shellfish intake with estimates of chemical concentrations in tissue. The daily exposure to each chemical was estimated using the following algorithm:

$$\text{Dose (mg/kg-day)} = \frac{C \times CR \times FI \times ED \times EF}{BW \times AT \times CF}$$

where:

- C = tissue chemical concentration (μg/kg-wet weight)
- CR = fish consumption rate (kg/day)
- FI = fraction ingested from the site (unitless)
- ED = exposure duration (years)
- EF = exposure frequency (days/year)
- BW = body weight (kg)
- AT = averaging time (days)
  - noncarcinogens: exposure duration $\times$ 365 days
  - carcinogens: 70-year lifetime $\times$ 365 days
- CF = conversion factor (1,000 μg/mg).

The assumptions used in the risk assessment are presented in Table 11-5. The bases for critical assumptions used to estimate exposure are described below.

11.3.2.1 Tissue Chemical Concentrations

EPA guidance (U.S. EPA 1989a, 1992c, 2002b) indicates that exposure concentrations used in risk assessment calculations should be the lesser of the 95 percent UCL on the mean concentration or the maximum concentration. EPA recommends the 95 percent UCL as an estimate of mean exposure concentration because of the uncertainty associated with estimating the true average exposure concentration at a site. For normally distributed data, EPA recommends calculating the UCL based on the Student's *t*-statistic. For lognormally distributed data, EPA recommends the Land method using the *H*-statistic (U.S. EPA 1992c, 2002b). In all cases (i.e., PCBs in fish and lobster, and mercury in lobster) the data were determined to be lognormally distributed using the Kolmogorov-Smirnov test on untransformed and lognormally transformed data. Thus, 95 percent UCLs were calculated using the Land method:

$$UCL = \exp\left(\bar{y} + \frac{S_y^2}{2} + \frac{S_y \times H}{\sqrt{n-1}}\right)$$

where:

n	=	number of observations
H	=	H statistic for a given confidence level, n, and Sy
exp	=	exponential function
$\bar{y}$	=	average of the log-transformed data (y = ln(x))
Sy	=	standard deviation of the log-transformed data.

Where a chemical was detected at least once, one-half the quantitation limit for undetected samples was included in the calculation. Consistent with guidance provided by OEHHA (Brodberg 2002), only the edible portions of fish and lobster tissue are evaluated for the main part of the human health assessment. However, PCB data from whole lobster (the only chemical exceeding the TRG in whole lobster) are also evaluated in the uncertainty section. Table 11-4 presents the data and summary statistics by area for the chemicals evaluated.

11.3.2.2 Fish Consumption Rate

OEHHA (2001) recently reviewed national and state fish consumption studies with the aim of identifying the most appropriate data to utilize when assessing human consumption of fish in California. OEHHA concluded that the Santa Monica Bay Seafood Consumption Rate Study (SCCWRP and MBC 1994), a survey of relatively high-end sport and subsistence anglers conducted in 1991−1992, "…provides the best available dataset for estimating consumption of sport fish and shellfish in California." Accordingly, OEHHA uses the median consumption rate of 21 g/day from this study to derive its TRGs.

In the Santa Monica Bay study, a questionnaire was administered randomly to recreational anglers and shellfish harvesters fishing from piers, jetties, private boats, party boats, beaches, and rocky intertidal zones. Of the 2,376 individuals included in the original census, over 1,200 were subsequently interviewed regarding their specific fishing and fish consumption patterns. Of those interviewed, only data from 555 anglers and shellfish harvesters who reported fish consumption at least once during the previous 4-weeks were included in estimates of consumption. A small percentage of interviewees reported fishing everyday. Thus, in reviewing this study, OEHHA (2001) considered the distribution of consumption rates to generally represent a high fish consumption rate population, including sport and subsistence anglers. Therefore, it is appropriate to use a central tendency estimate from this study in deriving screening levels, as OEHHA did (OEHHA 1999), and for estimating risks to similar populations. As with most consumption rate studies, the data were highly skewed. In such cases, the median is a better estimate of the central tendency than the mean.

The median for the entire Santa Monica Bay study population was 21.4 g/day. Median consumption rates for the various ethnic groups ranged from 16.1 to 85.7 g/day, with Hispanics having the lowest median rate (16.1 g/day), followed by Whites (21.4 g/day) and Asians (Filipinos, Japanese, Koreans, Chinese, and Vietnamese) (21.4 g/day), Blacks (24.1 g/day), and those in the Other group (Thai, East Indian, Samoan, Hawaiian, Indonesian, Guamanian, and Malaysian) having the highest median rate (85.7 g/day). The median for the entire population of 21 g/day (rounded, as recommended by OEHHA) was used in the human health risk assessment.

This value was used for both fish and shellfish consumption, as recommended by OEHHA (2001).

OEHHA (2001) recommends that the 95th percentile consumption rate of 161 g/day from the Santa Monica Bay study can be used to "…encompass all potential high-consuming groups, including ethnic groups and/or subsistence fishers." Given the security measures at the site, it is impossible that subsistence fishing would currently occur in the area, and no changes in site use are anticipated. Nevertheless, a subsistence user scenario is included in the uncertainty analysis evaluating hypothetical risks assuming a subsistence user consumption rate of 161 g/day.

11.3.2.3 Fractional Intake from the Site

The fractional intake term describes the fraction of total fish consumed by an individual that is derived from the site being assessed. U.S. EPA (1989a) recommends the use of a fractional intake factor when evaluating risks from homegrown or wild caught foods, including fish and shellfish. There are no default assumptions available for fractional intake. Rather, the value depends on the specific characteristics of the site, including the potential for anglers to use the site when compared with other nearby angling opportunities. In the case of the shipyards, the site represents a part of the overall potential fishing grounds in San Diego Bay. In addition, as discussed previously, the site is under high security and the likelihood of anyone fishing or harvesting shellfish on the site, even on an intermittent basis, is very low. The types of relatively high-end sport or subsistence fish consumption on which the Santa Monica Bay study was based would not occur at the shipyards. Thus, application of a fractional intake is appropriate when assessing fish consumption risks for the site.

The most appropriate method of deriving a fractional intake for the site is to relate the amount of potential fishing access at the shipyards to the fishing access in San Diego Bay as a whole. If restrictions on public access at the shipyards are assumed to be absent (otherwise no risk at all would exist), then a similar assumption applies to all of San Diego Bay. Within the leaseholds, if fishing were to occur, it would likely be from the piers and shoreline, rather than from a boat. Thus, the amount of potential fishing access can be quantified by the shoreline length available to anglers. The length of the shoreline and piers within the NASSCO and Southwest Marine leaseholds is 3.2 km and 2.1 km, respectively, and the length of the shoreline for the entire bay is 93.2 km (Table 11-6). These measures of fishing access give a fractional intake of approximately 0.034 and 0.023 for the NASSCO and Southwest Marine leaseholds, respectively. In other words, it is assumed that 3.4 percent of fish consumption could potentially be derived from fish and/or shellfish from the NASSCO leasehold and 2.3 percent from the Southwest Marine leasehold if there were no access restrictions. These values are conservative (i.e., overestimate consumption from the sites) because the shore length used for the entire bay does not include the length of piers other than in the shipyards, thereby underestimating the shore length available outside the shipyards and overestimating the fraction of shore length available inside the shipyards.

Outside the leaseholds, fishing would occur by boat only. Thus, the amount of potential fishing access can be quantified by the total area available to anglers. The total area outside the NASSCO and Southwest Marine leaseholds is 238,147 m^2 and 92,080 m^2, respectively, and the total area in the entire bay is 44,722,748 m^2. These measures of fishing access give a fractional intake of approximately 0.005 and 0.002 for NASSCO and Southwest Marine, respectively, outside their leaseholds. In other words, it is assumed that 0.5 percent of fish consumption could potentially be derived from fish and/or shellfish from the site area outside the NASSCO leasehold and 0.2 percent from the site area outside the Southwest Marine leasehold.

11.3.2.4 Exposure Frequency and Duration

The exposure frequency describes the assumed number of days per year an individual consumes fish. However, because fish consumption rates are already averaged over 365 days to give a daily average intake, the appropriate exposure frequency to use in a fish consumption risk assessment is 365 days per year.

Exposure duration describes the number of years that an individual might consume fish from the site and is based on the assumption that anglers at the site are people who live in the area. An exposure duration of 30 years was used in the human health assessment. This is consistent with residential exposure assumptions provided by U.S. EPA (1989a, 2001b,c) and is based on the 90th percentile of time that individuals live in one residence.

Using the exposure algorithms and assumptions described above (and listed in Table 11-4) in combination with the exposure concentrations listed in Table 11-5, chemical intake from fish and shellfish consumption were calculated (Table 11-6). Inside the NASSCO leasehold, the estimated mercury intake from edible lobster tissue is 5×10^{-6} μg/kg-day. Inside the Southwest Marine leasehold, the estimated PCB intakes from fish and edible lobster tissue are 1×10^{-6} μg/kg-day and 6×10^{-8} μg/kg-day, respectively. Outside the Southwest Marine leasehold, the estimated PCB intake from fish is 3×10^{-8} μg/kg-day.

11.4 Toxicity Assessment

In the toxicity assessment, the hazards associated with chemicals of concern at the site are evaluated. For noncarcinogenic chemicals, EPA has developed specific toxicity values called RfDs. An RfD is an estimate of the level of daily exposure that is likely to be without appreciable risk of health effects over a lifetime, even in sensitive populations. Potential carcinogenic effects are evaluated through application of a CSF. The primary resource for these toxicity values is EPA's Integrated Risk Information System (IRIS), which is available online (U.S. EPA 2003). RfDs and CSFs from IRIS were used to calculate risk in this assessment. In cases where it was necessary to derive TRGs for the chemical screening, RfDs and CSFs were used from IRIS or, in the case of carcinogenic PAHs, from the California Environmental Protection Agency (OEHHA 2001). The RfDs and CSFs used in the risk assessment are listed in Table 11-2.

11.5 Risk Characterization

In the risk characterization, quantitative exposure estimates and toxicity factors are combined to calculate numerical estimates of potential health risk. In this section, potential cancer and noncancer health risks were estimated assuming potential long-term exposure to substances detected in site fish and lobster tissue. Risk estimates are presented in Table 11-7. For lobster, risks were calculated only for within the leaseholds because lobsters were not found outside the leaseholds. However, even if it were assumed that chemical concentrations in lobster from within the leaseholds were similar to concentrations that could hypothetically be found outside the leaseholds, estimated risks would be lower because the fractional intakes are lower outside the leaseholds. Thus, risk estimates associated with lobster consumption from within the leaseholds would be protective of hypothetical lobster consumption scenarios outside the leaseholds.

11.5.1 Quantification of Carcinogenic Risks from PCBs

Cancer risk is estimated by multiplying the carcinogenic chronic daily intake of the chemical by its cancer slope factor:

$$\text{Risk} = \text{Intake} \times \text{CSF}$$

A 1×10^{-5} cancer risk represents a one-in-one-hundred-thousand additional probability that an individual may develop cancer over a 70-year lifetime as a result of the exposure conditions evaluated. The likelihood that actual risks are greater than estimated risks is very low because of the conservative assumptions used to develop cancer risk estimates; in fact, actual risks may be significantly less than predicted values. EPA's *Guidelines for Cancer Risk Assessment* state ". . . the linearized multistage procedure (typically used to calculate CSFs) leads to a plausible upper limit to the risk that is consistent with proposed mechanisms of carcinogenesis . . . . The true value of the risk is unknown, and may be as low as zero" (51 Fed. Reg. 185:33992, 33998).

In the human health risk assessment, PCBs were the only carcinogen present at concentrations greater than the screening levels, and the maximum concentrations exceeded the TRG in both fish and lobster tissue. Maximum PCB concentrations in lobster exceeded reference concentrations within the Southwest Marine leasehold. Maximum PCB concentrations in fish exceeded reference concentrations within the Southwest Marine leasehold and outside the Southwest Marine leasehold. However, tissue concentrations at the shipyards are not significantly different from the reference area, except for PCBs in spotted sand bass inside the Southwest Marine leasehold (by ANOVA and Kruskal-Wallis tests followed by a Dunnett's test, at $p = 0.05$).

For both fish and lobster, risks calculated using maximum concentrations did not exceed the target risk level of 1×10^{-5}. The estimated risks for each of the areas were:

- **Inside Southwest Marine Leasehold**—2×10^{-6} for fish consumption and 1×10^{-7} for lobster consumption
- **Outside Southwest Marine Leasehold**—6×10^{-8} for fish consumption.

Thus, even under the highly conservative assumptions used in this risk assessment, cancer risk estimates associated with hypothetical future consumption of fish and shellfish would be protective of beneficial uses.

11.5.2 Quantification of Noncancer Risk from Mercury

Unlike carcinogenic effects, other potential adverse health effects are not expressed as a probability. Instead, these effects are expressed as the ratio of the estimated exposure over a specified period to the RfD derived for a similar exposure period. This ratio is termed a hazard index and is calculated through application of the general algorithm:

$$\text{Hazard Index} = \frac{\text{Intake}}{\text{RfD}}$$

A hazard index less than 1.0 implies that exposure is below the level that is expected to result in a significant health risk. A hazard index greater than 1.0 does not necessarily mean that an effect would occur, rather that exposure may exceed a general level of concern for potential health effects in sensitive populations. Exposures resulting in a hazard index less than or equal to 1.0 are very unlikely to result in noncancer adverse health effects. Because EPA states that the range of possible values around RfDs is "perhaps an order of magnitude" (Dourson 1993), the significance of intakes exceeding the RfD by one-half an order of magnitude or less (i.e., hazard indices less than 5) must be carefully considered. However, because of the uncertainties in data supporting RfDs, their use may also underestimate risk.

In the screening stage, mercury in lobster tissue was the only chemical whose maximum concentration exceeded a noncancer TRG. The maximum mercury concentration in lobster from the NASSCO leasehold exceeded the maximum reference concentration. For the risk assessment, the mercury hazard index associated with lobster consumption, however, was 0.05 for the NASSCO leasehold, well below the target hazard index of 1.0.

Although PCBs were evaluated for cancer risk, some PCBs have also been associated with noncancer effects. However, there is no EPA derived noncancer RfD for the only detected Aroclor® (i.e., Aroclor® 1260). Risk assessments protective of the cancer effects of PCBs are typically also protective of the noncancer effects. Nevertheless, to ensure that noncancer risks of PCBs are adequately assessed in the human health assessment, they are addressed in the uncertainty assessment.

11.5.3 Uncertainty Assessment

Because risk characterization serves as a bridge between risk assessment and risk management, it is important that major assumptions, scientific judgments, and estimates of uncertainties be described in the assessment. Risk assessment methods are designed to be conservative to address the uncertainties associated with each step in the risk assessment process. Thus, "true" risks associated with consumption of fish and shellfish are likely to be less than risks estimated using standard risk assessment methods. In the uncertainty assessment, it is appropriate to discuss the most significant uncertainties and, to the extent possible, evaluate their effect on the risk estimates.

Risk assessment is subject to a number of uncertainties. General sources of uncertainty include the site characterization (adequacy of the sampling plan and quality of the analytical data), the exposure assumptions, and estimation of chemical toxicity, background concentrations, and the present state of the science involved. In this section, several key sources of uncertainty specific to this site are evaluated, including alternative consumption rates and fractional intake assumptions, issues related to PCB toxicity assessment, evaluation of hypothetical risks associated with consumption of lobster whole body, and assumed inorganic arsenic levels.

11.5.3.1 Fish and Shellfish Consumption Rates

The consumption rate used in the human health assessment represents the median consumption rate from a study of relatively high-end sport and subsistence anglers. Given the security measures at the site, it is not possible that subsistence fishing would occur in the leaseholds and, in fact, the consumption rate used in the assessment of 21 g/day certainly overestimates fish and shellfish consumption from the site. It is also highly unlikely that sport or subsistence fishing would occur on the portions of the site outside the leaseholds. Nevertheless, risk estimates for a hypothetical subsistence user scenario are included for the areas outside the leaseholds to ensure that highly exposed subpopulations would be adequately protected.

OEHHA (2001) recommends that the 95th percentile consumption rate of 161 g/day from the Santa Monica Bay study be used to "…encompass all potential high-consuming groups, including ethnic groups and/or subsistence fishers." This consumption rate is likely to overestimate actual exposure, even for subsistence fishing populations. OEHHA (2001) notes that in the Santa Monica Bay study "[t]he greatest differences in consumption rates for specific subpopulations were on the order of a maximum of five times greater when comparing the highest-consuming and lowest-consuming ethnic subpopulations in a survey." Furthermore, this consumption rate is nearly twice the median consumption rate for the highest consuming subpopulation in the study (85.7 g/day for the "Other" group).

Using a subsistence consumption rate of 161 g/day in combination with the other exposure assumptions used in the main assessment for the areas outside the Southwest Marine leasehold, the hypothetical PCB risk associated with fish consumption is 4×10^{-7}. This estimate is well below the target risk level of 1×10^{-5}. Therefore, even assuming high consumption rates associated with subsistence fishing or shellfish harvesting at the site, chemicals associated with the site do not appear to pose a significant human health risk.

11.5.3.2 Issues Related to PCB Exposure and Toxicity

Reduction in PCBs during Preparation and Cooking—The human health risks from exposure to contaminants in fish depend on the amount of contaminant actually ingested rather than the amount present in the aquatic species. Contrary to the default exposure assumptions used in many risk assessments that evaluate the potential hazards associated with fish consumption, most anglers are unlikely to be exposed to the concentrations measured in the raw fillet. There is growing consensus within the scientific and regulatory communities that preparation and cooking of edible fish tissue can result in significant loss of lipophilic contaminants, such as PCBs. Cooking methods that result in significant fat loss from fish tissue or allow for the transfer of lipophilic contaminants to cooking oil result in the greatest PCB losses.

Wilson et al. (1998) conducted a review of the literature on loss of contaminants from fish during cooking. These authors reviewed 14 studies published between 1972 and 1996 and found that mean reductions of PCBs ranged from 26 percent for microwaving to 68 percent for boiling. The mean reduction in PCBs for fried fish was found to be 48 percent (Wilson et al. 1998). Consistent with these findings, the recent research and guidance from *The Uniform Sport Fish Consumption Advisory for the Great Lakes Region* proposed by the Great Lakes Sport Fish Advisory Task Force (GLSFATF 1993, as cited in U.S. EPA 1993c) includes an assumed 50 percent reduction factor for PCBs in fish fillets. This reduction factor is more conservative than the one indicated in the draft *Sampling and Guidance Manual* (U.S. EPA 1993b, as cited in U.S. EPA 1993c), which indicates that trimming and cooking can reduce PCB concentrations in fish fillets by 60–90 percent.

A cooking loss factor was not applied in this assessment. However, PCB concentrations detected in fish and lobster samples would likely be reduced during cooking, so that the risk values presented here would be even more greatly overestimated. Available data indicate that

PCB concentrations would be reduced by 50 percent or more. Thus, by omitting a loss factor, risk may be overestimated by as much as 2-fold or more in this assessment.

Uncertainties in the PCB Cancer Slope Factor—The current EPA toxicity values applied for PCBs, including both the CSF and the RfDs, are based on studies in experimental animals. Animal studies indicate that the carcinogenic potential of PCB mixtures varies widely within Aroclor® mixtures (e.g., Aroclor® 1260) and between Aroclor® mixtures (e.g., Aroclors® 1260 and 1016) (IEHR 1991; U.S. EPA 1996a). U.S. EPA (1996a) summarized available studies on the carcinogenic potential for four different Aroclors® in laboratory animals and calculated a number of upper-bound slope estimates based on liver cancers in male and female Sprague-Dawley rats. Despite the wide range of upper-bound slope estimates for the four Aroclor® mixtures, a CSF of 2 $(mg/kg\text{-}day)^{-1}$ based on data for Aroclors® 1254 and 1260, EPA recommends applying the more conservative CSF to all fish consumption risk estimates (U.S. EPA 2003). Although 1260 was the only Aroclor® detected in edible tissue from this study, thus reducing inter-mixture uncertainty, there is also a large amount of variability within mixtures. For example, the upper-end CSFs calculated by EPA for Aroclor® 1260 ranged from 0.2 to 2.2 $(mg/kg\text{-}day)^{-1}$. Use of the most conservative (i.e., highest) of these upper-end CSFs will tend to overestimate risks from PCBs, potentially by an order of magnitude or more.

Noncancer Risks from PCBs—Although PCBs were evaluated for cancer risk, some PCBs have also been associated with noncancer effects. However, there is no EPA derived noncancer RfD for the only Aroclor® detected in lobster and fish tissue (i.e., Aroclor® 1260). Risk assessments protective of the cancer effects of PCBs are typically also protective of the noncancer effects. In some cases, however, estimated noncancer risks may exceed target risk levels when cancer risks do not. To ensure that PCBs are adequately addressed, noncancer risks were also evaluated in this assessment. For evaluation of noncancer effects, EPA has published RfDs for two Aroclors® based on studies in monkeys: an RfD for Aroclor® 1254 of 0.00002 mg/kg-day and one for Aroclor® 1016 of 0.00007 mg/kg-day. For the purpose of this assessment, the more conservative (i.e., health protective) RfD for Aroclor® 1254 was used as a surrogate. This is likely to result in an overestimation of risk.

Using the exposure algorithms and assumptions listed in the main assessment, along with the Aroclor® 1254 RfD of 0.00002 mg/kg-day, the estimated hazard indices associated with fish and lobster consumption were:

- **Inside Southwest Marine Leasehold—**0.14 for fish consumption and 0.007 for lobster consumption
- **Outside Southwest Marine Leasehold—**0.003 for fish consumption.

Thus, even under the highly conservative assumptions used in this risk assessment, estimated non-cancer risks associated with hypothetical future consumption of fish and shellfish are well below the target risk level of 1.0.

11.5.3.3 Assessment of Whole Body Lobster

Consistent with guidance provided by OEHHA (Brodberg 2002), only the edible portions of fish fillets and lobster tissue were evaluated for the main part of the human health risk assessment. However, it is possible that specific subpopulations could prepare lobster in a manner where chemicals in the inedible portions of the lobster could be released. Specifically, it is assumed that whole lobsters could be cooked in soups or stews, potentially releasing some of the chemicals from inedible portions of the lobster. Therefore, to ensure that potential risks associated with consumption of chemicals in the inedible portions of the lobster are adequately addressed, whole lobster bodies were also analyzed for the target substances.

In the screening step, when maximum chemical concentrations from whole lobster bodies were compared to TRGs, only PCBs exceeded the screening levels. The maximum concentrations of PCBs in lobsters from both the NASSCO and Southwest Marine leaseholds also exceeded the maximum reference value. Table 11-4 presents the data and summary statistics for whole body lobster PCBs. Combining the 95 percent UCL exposure concentration for whole body lobster PCBs with the exposure assumptions from the main assessment and the PCB CSF gives an estimated cancer risk of 7×10^{-7} for NASSCO and 3×10^{-7} for Southwest Marine. Both are well below the target risk level of 1×10^{-5}. Therefore, the hypothetical risks associated with the cooking and eating of whole lobsters from the shipyard areas are negligible.

11.5.3.4 Inorganic Arsenic as a Percent of Total Arsenic

The form of arsenic found in fish is critical to evaluating potential adverse effects, if any, in consumers. Arsenic in marine fish and shellfish has long been recognized to occur primarily as organic forms that have reduced or negligible toxicity. Specifically, arsenic is present in almost all marine animal species mainly as arsenobetaine (Edmonds and Francesconi 1993), a stable pentavalent arsenic compound that has been shown to be nontoxic in several studies (Eisler 1994).

Chew (1996) summarized seafood inorganic arsenic levels published in the literature for 30 species of finfish and at least 11 species of shellfish. In these studies, inorganic arsenic ranged from 0 to 9.5 percent of total arsenic in finfish, with 34 of 36 results less than 4 percent. Inorganic arsenic generally constitutes a lower percentage of total arsenic in shellfish. In the Chew (1996) report, 49 of 50 results were less than 3 percent inorganic arsenic. In a review conducted by Donohue and Abernathy (1999), 74 of 77 marine fish specimens had inorganic arsenic levels less than 4 percent of total arsenic. In shellfish, 54 of 57 specimens had inorganic arsenic levels less than 3 percent of total arsenic. The EPA Office of Science and Technology in its document titled *Arsenic and Seafood Consumption* (U.S. EPA 1997b) reviewed four studies that reported speciated arsenic data in fish tissue and concluded that, in general, the data indicate that less than 10 percent of the total arsenic in fish is inorganic. In that document, EPA assumed a maximum value of 4 percent inorganic arsenic content for fish and shellfish to estimate exposure to inorganic arsenic.

In the human health risk assessment for the NASSCO and Southwest Marine shipyards, exposure concentrations for inorganic arsenic in fish tissue were assumed to be 4 percent of total arsenic to account for the percentage of arsenic in fish that occurs in nontoxic forms. This assumption is considered conservative, and thus more likely to overestimate risk, in that it exceeds the percentage of inorganic arsenic found in fish and shellfish reported in all but a few studies.

11.6 Summary and Conclusions

Chemical concentrations in fish and lobster tissue were screened against TRGs protective for human consumption. Two chemicals, PCBs in both fish and lobster, and mercury in lobster only, exceeded screening TRGs. Concentrations of these two chemicals were further screened against chemical concentrations in fish and lobster from reference areas. Within the NASSCO leasehold, maximum concentrations of mercury in lobster exceeded reference concentrations. Within the Southwest Marine leasehold, maximum concentrations of PCBs in fish and lobster exceeded reference concentrations. Outside the Southwest Marine leasehold, maximum concentrations of PCBs in fish exceeded reference concentrations. These chemicals were selected for evaluation in the human health risk assessment.

Estimated cancer risks associated with PCB exposure were:

- **Inside Southwest Marine Leasehold**—2×10^{-6} for fish consumption and 1×10^{-7} for lobster consumption
- **Outside Southwest Marine Leasehold**—6×10^{-8} for lobster consumption.

The estimated hazard index associated with mercury exposure was:

- **Inside NASSCO Leasehold**—0.05 for lobster consumption

In no case do risks exceed target risk levels. The existing conditions at the shipyards are protective of beneficial uses associated with human health. Therefore, it is unnecessary to derive cleanup levels for protection of human health at the site.

12 Development of Candidate Cleanup Levels

Potential adverse effects on aquatic life, aquatic-dependent wildlife, and human health have been evaluated, with the following results regarding candidate cleanup levels for each of these types of beneficial uses:

- **Aquatic life—**Some adverse biological effects were found. The lowest chemical-specific no-effects level across all the toxicity tests and the benthic macroinvertebrate community analysis (LAET) is the effects-based candidate cleanup level that provides the best balance between false positive and false negative errors.

- **Aquatic-dependent wildlife—**Hazard quotients are less than 1.0, and no cleanup levels are required for protection of aquatic-dependent wildlife.

- **Human health—**Risk is at or below the threshold of 1×10^{-5} and hazard quotients are less than 1.0, and no cleanup levels are required for protection of human health.

Based on these results, the current conditions at the shipyards are protective of beneficial uses associated with human health and aquatic-dependent wildlife. For the purpose of conducting the evaluation of candidate cleanup levels specified by the RWQCB (2001), the LAET will be used as a candidate cleanup level for protection of aquatic life. However, a lower overall error rate would be achieved by using any of the other individual AETs.

As required by Regional Board staff (RWQCB 2001), background chemical concentrations must also be considered as candidate cleanup levels. The origin of the background chemical concentrations to be used is described in the previous section titled *Reference Stations and Background Conditions*. The site-specific candidate cleanup levels and background concentrations are summarized in Table 12-1. This table also includes effect range-low (ER-L) and effect range-median (ER-M) values (Long et al. 1995) for comparison. ER-L and ER-M values are often used to screen sediment chemical concentrations in the absence of any site-specific measures of effects or cleanup levels. Concentrations below the ER-L are regarded as highly unlikely to be associated with adverse biological effects (Long et al. 1995). Concentrations above the ER-M are regarded as potentially requiring site-specific assessment of effects ("…ER-M exceedances should only be taken to indicate that further analysis is in order. They should never be taken, by themselves, to mean that sediment is exerting a toxic effect upon the environment or that there would be any benefit to decreasing its chemical content" [O'Connor et al. 1998]). Because ER-L and ER-M values were developed by discarding all no-effects data (as well as some effects data) and then assuming that every detected chemical concentration caused an adverse biological effect, neither set of values bears any definable relationship to actual no-effect levels (that is, the ER-M and ER-L values have a high negative predictive value but a low positive predictive value). However, despite the limited interpretive value that can be placed on exceedance of either the ER-L or ER-M values, they can be used to identify chemical concentration ranges that have a low potential for causing biological effects.

The defined background conditions (95%UPL of the final reference pool) are low concentrations for all chemicals: of the 7 chemicals with ER-L and ER-M values, none of the background values exceed the ER-M and 5 are comparable to or below the ER-L. As a consequence of both their method of definition and their low concentrations relative to effects range values, the defined background conditions are unlikely to have any ability to distinguish impacted from non-impacted areas.

12.1 Exceedances of Candidate Cleanup Levels

Comparison of sediment chemistry data from Phases 1 and 2 to both the LAET and the final reference pool-based chemistry criteria have very different results. For this comparison, surface sediment data from stations that were sampled in both Phase 1 and Phase 2 (triad and pore water stations) were averaged, and half of the quantitation limit was used for those chemicals that were not detected. At every station, at least one chemical, at some depth, exceeded the 95%UPL background values. Because concentrations decreased with depth at most locations, more chemicals generally exceeded the background-based criterion in surface and near-surface sediment than in subsurface sediment. Core stations at which no exceedance of background were found at depth were NA13 (below 2 ft), NA17 (below 4 ft), NA21 (below 6 ft), NA24 (below 2 ft), NA25 (below 2 ft), NA26 (below 2 ft), NA29 (below 2 ft), NA30 (below 2 ft), NA31 (below 2 cm), SW08 (below 6 ft), SW10 (below 2 ft), SW19 (below 4 ft), SW29 (below 6 ft), SW30 (below 8 ft), SW31 (below 2 ft), SW32 (below 2 ft), and SW33 (below 2 ft).

Exceedances of the LAET values occur at fewer than half of the sediment stations sampled at the shipyards. Summaries of the locations, depths, and chemicals exceeding the LAET criteria are shown in Tables 12-2 and 12-3. Twenty of 67 stations have at least a minor exceedance of an LAET value. Petroleum hydrocarbons are the chemicals most frequently exceeding their LAET values; DRO or RRO exceed their LAET at all 20 stations, and at 10 stations only, petroleum hydrocarbon concentrations are above the LAET. Only six stations have LAET exceedances in surface sediment.

Among the shipyard stations that have an LAET exceedance, there are large differences in the extent of the exceedances. Evaluation of the identity of chemicals exceeding LAETs, the extent of exceedance, and the depth and location of exceedance identifies several locations at which exceedances are minor or isolated at depth. These locations, and the conditions at each, are as follows:

- **NA20—**Only DRO exceeds the LAET, and only in the sample from 6–8 ft. The exceedance was relatively low, less than twice the LAET. Because this exceedance is in deeply buried sediment, it is isolated from aquatic life and aquatic-dependent wildlife. Triad analyses were conducted at this station, and no adverse effects were found on amphipod survival, echinoderm fertilization, or bivalve development. Alterations were found in the benthic macroinvertebrate community; however, the SPI data, the grain size profile, and the usage of this area for engine tests indicate that these effects are the result of physical disturbance rather than chemical exposure.

- **SW25—**The only exceedance is for DRO, at a concentration of 500 mg/kg in comparison to a criterion of 490 mg/kg. The exceedance was in buried sediment: it occurs only in the sample from 2–4 ft. Surface sediment does not exceed any LAET value.

- **SW30—**Only DRO exceeds the LAET, and only in the 2–4 ft sample. The exceedance is in buried sediment, and is less than twice the LAET. Surface sediment does not exceed any LAET value.
- **SW36—**Only DRO exceeds the LAET, and only in the 2–4 ft sample. The exceedance is in buried sediment, and is less than twice the LAET. Surface sediment does not exceed any LAET value.

In contrast to the stations that only have minor exceedances of DRO, several stations have exceedances of large numbers of chemicals, some with large exceedance factors. These stations, and their characteristics, are as follows:

- **NA04—**Petroleum hydrocarbons exceed the LAET throughout most of the sediment column by factors up to 11. Unlike most locations at the shipyard, concentrations increase with depth at this station. No exceedances were found in the surface (0–2 cm) sediment, however.
- **NA09—**All classes of chemicals measured, except for TBT, had some exceedances of the LAET. The highest exceedance factors were for petroleum hydrocarbons. Unlike most locations at the shipyard, concentrations increase with depth at this station down to the 4–6 ft sample; below this depth, concentrations are much lower and do not exceed any LAET.
- **NA16—**Petroleum hydrocarbons exceed the LAET at this station by factors of up to 16. The highest concentrations of DRO and RRO in this core are in the 2–4 ft sample; below this depth concentrations are much lower and do not exceed any LAET.

- **SW04—**Every LAET value is exceeded at some depth at this station. Exceedances are found primarily in the 0–2 cm and 2–4 ft samples. The largest exceedance is for PCBs, which exceeds the LAET by a factor of 9.
- **SW08—**Metals, TBT, PCBs, and petroleum hydrocarbons all exceed the LAET at this station. Of the major classes of chemicals, only PAH does not exceed its LAET at any depth. The highest concentrations are found in the 0–2 ft sample, and there are no LAET exceedances below 4 ft.

Remediation to the LAET boundary shown in Figure 12-2 would remove the highest concentrations of most shipyard-associated chemicals. For example, all copper concentrations above 800 mg/kg in surface sediment would be removed, all TBT concentrations above 1.4 mg/kg in surface sediment would be removed, all PCB concentrations above 2.4 mg/kg in surface sediment would be removed, and all HPAH concentrations above 17 mg/kg in surface sediment would be removed. Maximum concentrations of other metals would also be removed. Remediation to this boundary therefore addresses the highest chemical concentrations regardless of the generally weak association between chemical concentrations and biological effects.

12.2 Performance of Candidate Cleanup Levels

Comparison of the stations exceeding the LAET with the results of the integrated assessment of potential effects on aquatic life (Table 9-7) allows an assessment of the predictive accuracy of the LAET as a candidate cleanup criterion. Figure 12-1 contains such a summary, including a contrast between exceedances of the LAET and the final reference pool-based chemical criteria. Positive and negative predictive accuracy can be assessed by contrasting the stations likely to have adverse biological effects, as represented by the integrated evaluation of the biological testing data, with the stations exceeding a candidate cleanup criterion. Sensitivity, specificity, efficiency, and overall reliability of a remedial scenario can then be assessed, as illustrated by Figure 9-6.

The performance of the LAET cleanup criterion is moderate (Figure 12-1b), with a sensitivity of 11 percent, specificity of 90 percent, and overall reliability of 67 percent. However, shipyard chemicals are not the likely cause of effects at Stations NA20 and NA22. At Station NA20, physical disturbance is the likely cause of effects (only alterations of the benthic community were found), and at Station NA22, effluent from Chollas Creek and the city storm drain is the likely cause of effects (benthic alterations and bivalve toxicity, which is correlated with pesticides, were found). Hence, for a remedial scenario focused on the effects of chemicals associated with the shipyards, Stations NA20 and NA22 should be excluded from the set identified with adverse effects. Figure 12-1c illustrates the effect of considering adverse biological impacts due only to shipyard-associated chemicals. Sensitivity and specificity both increase slightly, and overall reliability increases from 67 to 73 percent.

The efficiency value represents the probability that beneficial use impairments will be present at a station that is designated for remediation. For strict application of the LAET, this value is 33 percent (Figure 12-1b). This value can be contrasted with the overall fraction of triad stations at which beneficial use impairments are likely, which is 9 of 30 stations, or 30 percent. The fraction of stations with likely beneficial use impairments that would be removed by remediation to the LAET is essentially equivalent to the overall fraction throughout the shipyard sites. As noted previously, Stations NA20 and NA22 should not be included in the group of stations with likely beneficial use impacts associated with shipyard-associated chemicals only. Reclassifying them appropriately reduces the overall number of triad stations at which beneficial use impacts are likely to only 7 (23 percent of all 30 stations). The corresponding efficiency of remediation to the LAET is also 33 percent, and the difference between 23 percent and 33 percent represents a considerable improvement.

The relative performance of the LAET cleanup criterion and the final reference pool-based chemistry values is also illustrated in Figure 12-1. Because the latter set of values predicts effects everywhere, they are extremely sensitive but completely non-specific. The overall reliability of the final reference pool-based values is lower than that of the LAET values, at 30 percent compared to 57 percent. Focusing on the possible effects of shipyard chemicals only, by excluding Stations NA20 and NA22 from the set identified as having adverse effects, reduces the overall reliability of the final reference pool-based values to 23 percent.

Locations at the shipyards that exceed the LAET cleanup criterion are shown in Figure 12-2. Locations that exceed only the LAET for petroleum hydrocarbons are distinguished from locations that exceed other LAET values. Petroleum products are made up of a variety of compounds, including both aromatic (those in which the carbon backbone forms ring structures) and aliphatic (those in which the carbon backbone does not form ring structures). Aromatic compounds include PAH, and are the most toxic constituents of petroleum. The aromatic compounds are also those that are most rapidly broken down after release (Lee and Page 1997; NOAA 2001; Page et al. 2001). Toxic effects associated with petroleum spills diminish substantially or completely in periods of several weeks to several months. The DRO and RRO present in sediment at the shipyards are not associated with PAH or with toxicity for the following reasons:

- The spatial distributions of PAH and petroleum hydrocarbons are different
- PAH at the shipyards are primarily pyrogenic, rather than petrogenic, in origin
- The LAET for HPAH is exceeded at only three stations (NA09, SW04, and SW24)
- There are no adverse effects on amphipod survival associated with petroleum hydrocarbon concentrations
- There are no adverse effects on echinoderm fertility associated with petroleum hydrocarbon concentrations
- There are no adverse effects on bivalve development associated with petroleum hydrocarbon concentrations.

Although there is a statistically significant correlation between petroleum hydrocarbons and benthic macroinvertebrates, variation in petroleum hydrocarbons can explain only 8 percent of the variation in benthic macroinvertebrates, and the correlation is strongly controlled by a small number of data points (see Section 9.1). Altered benthic macroinvertebrate communities are also strongly associated with physical disturbance (see Section 9.1.5), and physical disturbance is found at most of the locations where LAET values for petroleum hydrocarbons are exceeded (Figures 4-2 and 12-1). Therefore, LAET exceedances for petroleum hydrocarbons do not necessarily represent potential areas of adverse biological effects. In contrast to the petroleum hydrocarbons, the other chemicals for which LAET values are exceeded are not rapidly biodegraded, and those LAET exceedances are therefore more likely to represent potential areas of adverse biological effects. The overall reliability of the LAET for only these chemicals is 60 percent, higher than that for the set of LAETs that includes petroleum hydrocarbons.

Figure 12-2 includes a boundary line around the set of stations that exceed LAET for chemicals other than petroleum hydrocarbons. This boundary was established by contouring the chemical concentrations, using the average of only those concentrations over the LAET for the stations with any LAET exceedance, and the average of all samples for those stations without an LAET exceedance. Contouring was done using an inverse distance weighting method, with an exponent of 4 and an ellipsoidal search distance with major and minor axes of 60 and 50 ft and the major axis parallel to the shore (310 degrees). The boundary shown in Figure 12-1 is the outermost of the contours for all chemicals.

Although the LAET performs better than other effects-based candidate cleanup levels (described in the section titled *Assessment of Potential Effects on Aquatic Life*), because of the lack of correlation between adverse effects and shipyard-associated chemicals, and because of the presence of adverse effects that are attributable to causes other than shipyard chemicals, the LAET has only moderate performance as a basis for remediation at these shipyards.

12.3 Causation Analysis

The absence of statistical correlations between shipyard chemicals and effects on aquatic life shows that shipyard chemicals are not related to those effects. Although the statistical analyses strongly imply the absence of a cause-and-effect relationship, they strictly address only covariation. The causal relationship between shipyard chemicals and aquatic life effects can, however, be directly evaluated using a causation analysis based on rules of formal logic. Potential cause-and-effect relationships at the shipyards can be formulated as hypotheses and tested using the data collected during this investigation. Testing can be carried out using truth tables for exclusive (only cause) and non-exclusive (not necessarily the only cause) causation, as described in Appendix O.

Causation analysis requires potential causes and effects to be formulated as true/false statements that can be evaluated using data from this investigation. The fundamental hypothesis that this investigation is intended to evaluate is: "High concentrations of shipyard chemicals are causing adverse effects." The LAET is the lowest no-effects level for aquatic life effects, so the condition "High concentrations of shipyard chemicals" is equivalent to "Concentrations of shipyard chemicals exceeding the LAET." Because no adverse effects on aquatic-dependent wildlife or human health are present at the shipyard sites, the phrase "adverse effects" is equivalent to "adverse effects on aquatic life." The fundamental hypothesis is therefore equivalent to "Concentrations of shipyard chemicals exceeding the LAET are causing adverse effects on aquatic life." The data set with which such hypotheses can be tested consists of the triad stations, for which both chemical concentrations and aquatic life effects were evaluated using synoptic data.

All possible combinations of LAET exceedance and likely aquatic life effects, with the truth-table values for exclusive and non-exclusive causation, and the stations associated with each combination, are shown in Table12-4. The rows in this table directly correspond to the cells of the table in Figure 12-1. The existence of stations that match the categories in rows 2 and 3 of the truth table show that elevated concentrations of shipyard chemicals are neither an exclusive nor a non-exclusive cause of aquatic life effects. The fundamental hypothesis stated above is therefore false: high concentrations of shipyard chemicals are not causing biological effects.

Other factors, either alone or in some combinations with LAET exceedances, may instead be causing aquatic life effects (either exclusively or non-exclusively). The current data set allows some hypotheses about the effects of other factors to be formulated and tested. For example, the fraction of fine particles in the sediment was significantly (although weakly) correlated with responses in the amphipod and bivalve toxicity tests, as well as with total benthic macroinvertebrate abundance (Table 9-2). A reasonable hypothesis therefore is “A large amount of fine particles in the sediment is causing adverse effects on aquatic life.” This hypothesis has been evaluated and tested, using various threshold values for measured percent fines to define “a large amount of fine particles.” The results of these evaluations show that there is no threshold value at which high values of percent fines are either an exclusive or a non-exclusive cause of aquatic life effects.

Another hypothesis that can be formed is “The combination of LAET exceedances and fine sediment particles is causing adverse effects on aquatic life.” The first part of this hypothesis is a compound statement made up of a conjunction between the two factors of chemical concentrations and fine sediment. The truth table for logical conjunction is shown in Appendix O. The results of evaluating this hypothesis, using a threshold value of 65 percent for percent fines, are shown in Table 12-5. The absence of any stations in the second row of this truth table shows that the data are consistent with LAET exceedance combined with more than 65 percent fines as a non-exclusive cause of aquatic life effects. Of the 30 triad stations, only one exceeds the LAET, has high percent fines, and is likely to have aquatic life effects; this is the only case that links high concentrations of shipyard chemicals (in combination with percent fines) to likely biological effects. In contrast, there are eight stations at which likely biological effects are not associated with LAET exceedances and high percent fines. Effects at those stations are attributable to other causes. Because the likelihood of aquatic life effects was determined by comparison of site conditions to reference station conditions, it should be noted that the threshold value of 65 percent fines also distinguishes site and reference stations. Most site stations had higher values of percent fines, and most reference stations had lower values. The partial association between fine sediment and biological effects that is represented by this hypothesis may be a reflection of the physical differences between site and reference stations, and the fact that those physical differences are expected to lead to biological differences.

A formal causation analysis therefore shows that elevated concentrations of shipyard chemicals are not, by themselves, sufficient to cause adverse effects on aquatic life at the shipyards. A similar conclusion may be inferred from the results of the correlation analyses of chemical concentrations and biological effects. Therefore, the absence of causation by shipyard chemicals is supported by two independent lines of evidence. Elevated concentrations of shipyard chemicals in combination with elevated levels of fine sediment particles—a combination that occurs at one station within the two shipyard leaseholds—may be a non-exclusive cause of adverse effects on aquatic life. Other causes, such as non-shipyard chemicals, must be responsible for aquatic life effects at other shipyard stations. Although differences in sediment grain size among stations within the shipyard leaseholds are not a cause (exclusive or non-exclusive) of apparent biological effects, the differences in grain size between shipyard stations and reference stations may have a role in explaining the apparent biological effects.

13 Site Assessment Summary

The detailed sediment investigation conducted by NASSCO and Southwest Marine shipyards in response to State Water Quality Control Board Resolutions No. 2001-02 and 2001-03 has comprehensively evaluated the potential effects on aquatic life, aquatic-dependent wildlife, and human health beneficial uses due to shipyard-associated chemicals. This evaluation has included analyses of:

- Surface and subsurface sediment chemistry
- Sediment toxicity to amphipods (survival)
- Sediment toxicity to echinoderms (fertilization)
- Sediment toxicity to bivalves (embryonic development)
- Benthic macroinvertebrate abundance and community structure
- Bioaccumulation potential of sediment chemicals
- Mineral forms of metals in sediments
- Histopathological conditions of fish
- Potential exposure of fish to PAH compounds
- Chemical concentrations in indigenous fauna and ecological risks to six representative wildlife receptors
- Risks to human health from consumption of fish and shellfish.

The results of these analyses show that the current status of beneficial uses near the shipyards is as follows:

- Sediment toxicity and alterations of benthic macroinvertebrate communities exist at moderate levels in some locations

- Current conditions are protective of aquatic-dependent wildlife beneficial uses
- Current conditions are protective of human health beneficial uses.

Although concentrations of chemicals potentially associated with shipyard chemicals are above concentrations in the final reference pool samples designated by Regional Board staff, sediment toxicity and benthic macroinvertebrate community effects are generally not correlated with shipyard-associated chemicals. There are demonstrably no causal relationships between shipyard-associated chemicals and observed biological effects. The absence of a cause-and-effect relationship between shipyard chemicals and biological effects is confirmed by explicit tests of causal hypotheses. Shipyard-associated chemicals may not be the cause of sediment toxicity or adverse effects on benthic macroinvertebrate communities for the following reasons:

- Copper and chromium, and likely other metals as well, are chemically isolated in mineral matrices and therefore not bioavailable to sediment-dwelling organisms
- Physical disturbance is strongly associated with benthic macroinvertebrate community alterations where no toxicity is found
- Analyses of several sediment samples for pesticides—which are not shipyard-associated chemicals—show that chlordane and DDT isomers have a much stronger correlation with effects on aquatic life (bivalve development effects and benthic bivalve abundance) than any of the shipyard-associated chemicals.

The presence and potential effects of pesticides, are most likely due to the proximity of the shipyards to Chollas Creek, which is a known source of pesticides and other chemicals. Because the storm water outflow plume from Chollas Creek completely covers the shipyard leaseholds, and the shipyard leaseholds can be entirely within the part of the plume that is significantly toxic (Schiff et al. 2003), Chollas Creek is a likely source of toxic levels of pesticides (and other chemicals) to shipyard sediments. Municipal storm drains within both shipyard leaseholds are another potential source of contaminants.

Remediation of shipyard sediments prior to control of contaminant sources would be premature. Remediation would be ineffective because the shipyard leaseholds would be recontaminated by Chollas Creek and storm drain effluent. Nevertheless, in accordance with Regional Board staff guidance for conducting this investigation, candidate cleanup levels for shipyard-associated chemicals have been evaluated. The candidate cleanup level with the best predictive accuracy is the no-effects level for any alteration of benthic macroinvertebrate communities (equivalent to the LAET for all sediment effect assessments). Because of the absence of a causal relationship between shipyard-associated chemicals and biological effects, however, the predictive accuracy of this candidate cleanup level is only moderate. An LAET-based remedial alternative is evaluated, along with others, in the following feasibility section of this report.

Because even the greatest alterations of benthic communities observed at the shipyard had only about a 50 percent reduction in the abundance of benthic macroinvertebrates, and in most cases effects were substantially less (or absent), current conditions at the shipyard represent, at worst, only a moderate reduction in the aquatic life beneficial use. Mature benthic communities are found throughout the shipyard leaseholds, and abundances of benthic macroinvertebrate organisms at the shipyards are generally in the range of 4,000 to 8,000 individuals per square meter. Considering the likely importance of physical disturbance on benthic communities, and the very low risks to aquatic-dependent wildlife and human health, this comprehensive and detailed sediment investigation has demonstrated that shipyard-associated chemicals have a negligible impact on overall beneficial uses.

Part 2

Feasibility Study

14 Feasibility Study Introduction

The purpose of this feasibility study is to develop and evaluate remedial alternatives to address potential impairments of beneficial uses that are attributable to shipyard chemicals. Part 1 of the report describes site conditions and effects on beneficial uses and develops an effects-based candidate cleanup level. This feasibility study presents alternatives to address those effects. In addition, this feasibility study includes an alternative consisting of remediation of all sediment with chemical concentrations higher than those in the final reference pool (Barker 2003). The feasibility study is organized according to EPA guidance for Superfund studies (U.S. EPA 1988b) and specifically addresses the technological and economic feasibility evaluation criteria contained in Section X of the *Guidelines for Assessment and Remediation of Contaminated Sediments in San Diego Bay at NASSCO and Southwest Marine Shipyards* (RWQCB 2001).

The following subsections describe site-specific constraints. The remainder of the document includes the following primary sections:

- **Remedial Technology Screening—**Candidate remedial technologies are described and screened, and a summary of the retained technologies is provided.
- **Assembly of Remedial Alternatives—**The retained technologies are assembled into meaningful remedial alternatives to address the cleanup goals.
- **Detailed Evaluation of Alternatives—**The alternatives are reviewed in detail relative to a set of criteria to assess their feasibility. The broad classes of evaluation criteria include effects on beneficial uses, technical feasibility, and economic feasibility.
- **Comparative Evaluation and Ranking of Alternatives—**The results of the alternatives evaluation are compiled and the alternatives are ranked relative to one another with regard to the criteria evaluated in the detailed evaluation of alternatives.

14.1 Site-Specific Constraints

The NASSCO and Southwest Marine sites are active shipyards, and ongoing operations, as well as physical conditions, will limit the types of remedial technologies that can be applied at the sites. Consideration of the physical, chemical, and biological characteristics of the shipyard sediments plays an integral role in identifying and evaluating potentially applicable technologies and process options. The physical features of the shipyard sites are also important in determining whether a particular technology or disposal site is feasible. In the following subsections, the physical and chemical properties of the shipyard sediments and the major features of shipyard sites are described, and other potential constraints on the use of remedial technologies are discussed. The known properties of the sediment and features of the shipyard sites are then used in Section 15, *Remedial Technology Screening*, to assist in screening the sediment remedial technologies and process options.

14.1.1 Physical Properties of Shipyard Sediments

Testing of physical properties of sediment at the NASSCO and Southwest Marine sites was conducted during Phase 1 and Phase 2 of the detailed sediment investigation. In general, based on the core logs, sediments at NASSCO and Southwest Marine consist of about 3 to 9 ft of interbedded silts, clayey silts, and sandy silts, with occasional sandy and clayey lenses. They contain an average 70 percent fines content, with a range of 21 to 100 percent (dry weight) and an average total solids of 39 percent (with a range of 29 to 56 percent total solids). Underlying the site sediments is the Bay Point formation, a sequence of intermixed medium dense to dense sands and silty sands, and stiff to hard silty to sandy clays.

14.1.2 Physical Features and Limitations of the Shipyard Sites

Bathymetric survey information within the majority of the shipyard project investigation areas was collected by Racal Palogos, Inc., between September 1999 and December 1999 and is reported in feet below mean lower low water (MLLW). Additional bathymetric information was obtained for the nearshore/inner shipway area of Southwest Marine by URS (2002).

Throughout most of the site, the water depth ranges from 20 to 40 ft below MLLW. Water depths at the shipyard sites range from less than 10 ft near the bulkhead to about 55 to 70 ft in deepened sumps located beneath the shipyard dry docks.

Shipyard-leased land area is heavily occupied and used for operating facilities, materials laydown, administrative offices, and crane and vehicular travel routes. Very little upland area is available for non-operational activities, which imposes significant constraints on the selection of construction and disposal methods for any remedial action. In particular, there is a shortage of onsite space for dealing with dredged sediment (see discussion in Section 15.1.4) and for temporary stockpiling and dewatering to support transfer of dredged sediment to upland disposal facilities (see Section 16.3.1).

Similarly, the waterside leasehold property area presents extremely limited opportunities for storage of sediment onsite in a constructed nearshore confined disposal facility (CDF), because most of the waterside areas are occupied by piers and active berths. For the most part, using existing nearshore water space to construct a CDF would pose insurmountable obstructions to ongoing shipyard operations. The limited exceptions to this are discussed later in this document (Section 16.3.2, *Nearshore Confined Disposal*).

14.1.3 Biological Scheduling Constraints

Dredging is restricted in San Diego Bay from April 1 to September 15, when endangered least terns nest in the area. During this time, proximity to the closest least tern nesting colony limits whether remedial activities such as dredging may move forward. Dredging is not expected to be authorized during least tern nesting season. Turbidity is also an important issue outside of the tern breeding season, and the amount of fine sediment in the dredged material determines whether conditions such as the use of dredging engineering controls (e.g., silt curtains) are imposed on the project. Least terns and brown pelicans, species of concern in the bay, forage by sight; consequently, reducing the impact of dredging on water clarity is a focus of FWS (Kenney 2001, pers. comm.). There are no seasonal dredging restrictions dictated by fish species in the bay (Hoffman 2001, pers. comm.).

15 Remedial Technology Screening

This section outlines the technologies and process options that may be appropriate for containing, treating, removing, and disposing of shipyard sediments, and documents a screening of these technologies based on their predicted effectiveness, implementability, and relative cost. The technology screening serves to focus the feasibility study on technologies that are most suitable for the site by eliminating those that are obviously inappropriate or infeasible.

15.1 Description and Screening of Candidate Remedial Technologies

This section describes, in general terms, the remedial technologies that the RWQCB has required NASSCO and Southwest Marine to review and consider as part of the sediment cleanup and abatement alternatives analyses (RWQCB 2001) and provides a screening of these technologies, which may be used separately or in combination to achieve target cleanup levels at the shipyard sites. For the technology screening, each candidate remedial technology and some process options are evaluated for effectiveness, implementability, and relative cost based on known site-specific constraints and screening criteria and potential applicable or relevant and appropriate requirements (ARARs). Based on this evaluation, the candidate remedial technology is either retained for the detailed feasibility study or eliminated from further consideration.

The effectiveness of a technology includes its demonstrated ability to provide an efficient and permanent remedial solution under constraints similar to those at the shipyard sites, weighed against its potential impacts on beneficial uses in San Diego Bay. The implementability of a technology refers to the technical and administrative feasibility of employing the technology at the shipyard sites (U.S. EPA 1988b). In the screening, cost is appraised qualitatively and is described as low, moderate, or high, relative to other technologies. Table 15-1 presents these technologies and process options and the results of the screening.

15.1.1 Natural Recovery

Natural recovery is an integral part of EPA's contaminated sediment management strategy (U.S. EPA 1998c). As stated in U.S. EPA (1998c):

> In certain circumstances, the best strategy may be to implement pollution prevention measures as well as point and non-point source controls to allow natural attenuation. Natural attenuation may include natural processes that can reduce or degrade the concentrations of contaminants in the environment including biodegradation, dispersion, dilution, sorption, volatilization, and chemical or biological stabilization, transformation or destruction of contaminants, and the deposition of clean sediments to diminish risks associated with the site.

The acceptability of natural recovery for California waters has been affirmed by the State Water Quality Control Board's Office of Chief Counsel (Wilson 2002, pers. comm.):

> Resolution 92-49 allows for consideration of adverse impacts of cleanup as well as natural attenuation if cleanup goals can be met in a reasonable time.

Natural recovery would be an appropriate alternative if other remedial approaches would cause unacceptable environmental effects or would be otherwise infeasible, if the offsite sources have been controlled, if natural processes are sufficiently rapid, and if natural forces and human activities would not disturb the sediment (U.S. EPA 1993d).

The factors to consider when determining whether natural recovery is appropriate for a site include the following (U.S. EPA 1998c):

- The specific chemicals present and their associated risk
- Establishment of source control
- The designated uses impaired during recovery
- The size of the affected area

- The feasibility and costs of remediation
- Site hydrodynamics, including sediment transport
- The time required for natural recovery.

Natural recovery processes include:

- Deposition of new sediment resulting in dilution and burial of existing surface sediment
- Degradation of organic compounds through both chemical and biological processes
- Recolonization of sediment by benthic macroinvertebrates.

If offsite sources were to be controlled, natural recovery of benthic macroinvertebrate communities would be expected to occur within a 3–5 year period. Sediment deposition rates in San Diego Bay have been estimated to be 1 cm per year (Peng et al. 2003). This rate of sediment accumulation will lead to substantial changes in surface sediment conditions in just a few years. Although this sediment accumulation rate will nominally result in complete replacement of the most biologically active surface sediment layer (0–2 cm) in 2 years, physical and biological processes may mix the sediment to a greater depth. The apparent RPD depth at the shipyards generally ranged from 1 to 2.5 cm (Section 8.1.1.1), indicating the depth range over which bioturbation is likely to mix newly deposited sediment. Newly deposited sediment will therefore have a substantial impact on existing surface sediment in a period of 2 to 4 years.

Petroleum hydrocarbons are the chemicals that most commonly exceed LAET values at the shipyards, but petroleum hydrocarbons weather relatively quickly. The most toxic components of petroleum hydrocarbons are broken down in weeks to months in the marine environment (Lee and Page 1997; NOAA 2001; Page et al. 2001). As a result, remediation of subtidal sediments is ordinarily not required even after a major oil spill. A relatively short period of natural recovery is therefore expected to address any effects of petroleum hydrocarbons.

Benthic macroinvertebrate communities become reestablished in disturbed habitats through a series of stages characterized by different functional relationships between the organisms and the sediment. The first organisms to colonize a disturbed area are generally small, opportunistic tube-dwelling polychaetes that feed at the sediment surface or from the water column (Stage I fauna). Irrigation of these organisms' tubes pumps water into the sediment surface, which alters its chemical and physical properties and provides a more favorable environment for burrowing species. Continuing irrigation and bioturbation further modify the sediment and produce favorable conditions for further colonization by other species. The typical endpoint of this succession of colonizing stages is a community characterized by head-down tubiculous or free-living organisms that feed at the RPD (Stage III fauna) (Pearson and Rosenberg 1978; Rhoads and Boyer 1982; Aller 1982). Total abundances and biological productivity of Stage I fauna can be greater than those of Stage III fauna (Rhoads et al. 1978). Initial colonization of a disturbed habitat and development of abundant Stage I fauna may occur in just days to weeks, but the rate of this recolonization may depend on the seasonal availability of larval forms (Rhoads and Boyer 1982). Because of the sequential nature of the succession of stages, restoration of a mature community requires additional time, even under optimum conditions. In areas of frequent sediment disturbance, mature communities may never develop, or, if disturbance is limited to the top few centimeters of sediment, species representative of both pioneering and mature communities may be present (Rhoads and Boyer 1982).

Monitoring of changes in the benthic community on the Palos Verdes shelf (Stull 1995) provides a basis for estimating the rate of benthic macroinvertebrate recolonization in southern California. Treated wastewater from the city of Los Angeles is discharged onto the Palos Verdes shelf. In 1970, changes in treatment practices and source control resulted in a substantial reduction in the discharge of suspended solids (principally organic matter) and chemical contaminants. At that time, benthic macroinvertebrate communities near the outfall were dominated by Stage I fauna. Two decades of monitoring, from 1972 to 1992, showed increasing abundances of burrowing species, decreasing abundances of Stage I species, increasing diversity, and lower dominance—all indicators of progression toward a mature benthic macroinvertebrate community. During this period, the number of species per sample increased from 20–30 to 40–80 after about 10 years (Stull 1995). This length of time probably represents an upper limit for recovery times at the shipyards. Organic enrichment and chemical toxicity on the Palos Verdes shelf had rendered the sediment physically and chemically unsuitable for the burrowing fauna representative of mature benthic communities, and the recovery time therefore likely represents the time needed to reduce the biological oxygen demand and chemical toxicity. In contrast, sediment at the shipyards is currently suitable for mature benthic communities, as is indicated by their current presence throughout the shipyard leaseholds. Consequently, the rate of recolonization of dredged areas at the shipyards is likely to be considerably less than that on the Palos Verdes shelf.

More rapid benthic recolonization of disturbed sediment was observed at a dredged material dump site in Long Island Sound. Following deposition of contaminated harbor sediment covered by a cap of clean sand, the numbers of species at the dump site and a reference site reached parity approximately 18 months after placement of the dredged material (Rhoads et al. 1978). Because the dredged material was itself surface sediment, and some benthic organisms may have survived the dredging and dumping activities, the period of 18 months is likely to represent a minimum estimate of recovery time, particularly for a dredged area, where the newly exposed surface is completely azoic.

Although neither the Palos Verdes shelf nor Long Island Sound is likely to be strictly representative of conditions in San Diego Bay, these two examples provide a basis for estimating limits on the rate of benthic recolonization in San Diego Bay following dredging or the cessation of other physical disturbance. Reestablishment of a mature benthic community in San Diego Bay following complete sediment removal is likely to require an intermediate period of time. Several seasonal spawning cycles may pass before Stage III fauna find appropriately modified sediment conditions, and recovery times in San Diego Bay could therefore take as long as 3–5 years. Recovery times for sediments that already contain populations of both Stage I and Stage III fauna, and are subject to periodic disturbance from shipyard activities, are likely to be much shorter, although such recovery will take place only after the sites are no longer used for ship construction and maintenance.

Natural recovery is often combined with monitoring to evaluate the ongoing effectiveness and protectiveness of the technology as the recovery proceeds. Monitored natural recovery is potentially effective and implementable at the shipyards, especially for those areas affected only by petroleum hydrocarbons. This technology is retained for further evaluation.

15.1.2 Subaqueous Capping

In-place capping is the most straightforward and least intrusive of the active sediment remedial techniques. Capping material, typically clean sediments, sand, silty to gravelly sand, and/or armoring material is placed on top of problem sediments. The availability of appropriate capping material influences the site-specific suitability of capping as a remedial technology.

Capping material is generally brought to the site by barge and put in place using a variety of methods, depending upon the selected remedial action alternative. Placement methods include surface release from barges, tremie-tube or submerged diffuser placement, hydraulic washing, pipeline with baffle box or diffuser placement, and direct mechanical placement. The issues generally associated with in-place capping are 1) obtaining an appropriate cap thickness over the entire problem sediment area, 2) placing the capping material without displacing sediment, and 3) maintaining long-term cap integrity (e.g., caps must be able to withstand seismic events, and disturbance from wave forces, propeller wash).

There are two general categories of capping: thick capping and thin capping. The goal of thick capping is to isolate or physically confine the problem sediment, prevent migration of chemicals from the confined sediment, and in some cases, to replace or create new benthic habitat. The thickness of an in-place, physical containment cap (i.e., a "thick cap") is typically greater than 6 to 12 in. The thickness depends on the nature of the underlying sediments and their chemical characteristics. Thick capping requires that the sediment has sufficient structural strength to support the cap, and that capped slopes are flat enough to maintain the stability of the cap. Because cap placement raises the elevation of the mudline, capping is typically used in areas where navigational depths are not an issue.

The goal of thin capping, also known as enhanced natural recovery, is typically to aid in ongoing natural recovery processes by improving the chemical or physical properties of surface sediments constituting the biologically active zone. The objective is not to isolate the surface sediments, but instead to augment the natural sedimentation rate and biological processes by adding clean material to the existing sediments. With thin capping, surface coverage is expected to vary spatially, providing variable areas of capped surface sediments and amended surface sediment (i.e., where mixing between capping material and problem sediment occurs), as well as areas where no cap is evident.

The cost of capping is low relative to other remedial technologies, and the cost of thin capping would be less than that of thick capping, because less material is purchased and placed. Sediment slopes at the shipyards are generally low, and do not *a priori* rule out capping as a feasible remedial alternative. However, depending upon the cap location and water depth, subaqueous capping may conflict with future navigation, construction, or maintenance dredging at the shipyard sites. Subaqueous capping does not appear to be a feasible option for most areas of the shipyard sediments because of the ever-larger ships being serviced at the shipyards, the associated navigational requirements, and the likelihood of cap disturbance resulting from normal shipyard activities (e.g., prop wash). Therefore, subaqueous capping is not retained for further consideration.

15.1.3 Dredging

Dredging is the removal or excavation of sediments from a water body. The most common purpose of dredging operations is to remove large volumes of subaqueous sediments as efficiently as possible within a specified operational and environmental restriction (Palermo and Hayes 1992). The term “environmental dredging” has evolved in recent years to distinguish dredging operations for the purpose of environmental remediation from maintenance or navigational dredging. Environmental dredging operations must attempt to remove problem sediments as effectively as possible, while minimizing environmental risk and other adverse consequences.

Dredging involves active disturbance of the bed to dislodge sediment by mechanically penetrating, grabbing, raking, cutting, or hydraulically scouring with water jets. After the bed sediment is dislodged, the sediment is transported to the water surface hydraulically (e.g., by pipe slurry) or mechanically (e.g., by clamshell). Most dredges are categorized as either hydraulic or mechanical, depending on the method of transporting the sediment. The selection of mechanical or hydraulic dredging technique for a given dredge project is primarily dependent on access issues, and is often constrained by the method of sediment disposal and the characteristics of the end disposal facility. Selection of a dredging technique also has a significant impact on overall dredging rates. Dredging would be subject to several ARARs, including federal dredge and fill standards, federal and state water quality standards, the California Environmental Quality Act, and location-specific restrictions such as dredging windows. Dredging would also be subject to operational restrictions associated with active shipyard operations.

15.1.3.1 Effects of Dredging

Remedial dredging of contaminated sediments can have several adverse effects during and after dredging. These adverse effects include:

- Resuspension of contaminated sediment during dredging
- Alteration of contaminant bioavailability
- Destruction of the indigenous biotic community
- Alteration of habitat suitability.

Additional adverse effects expected are related to transport and ultimate placement or disposal of the sediment.

Uncontrolled release of sediment during dredging can have adverse impacts on nearby biological communities. These effects can include both immediate and long-term effects from smothering and toxicity. The potential for this type of adverse effect is likely to be greater at certain times of the year than at others. For example, effects may be greatest during spawning or migration periods of sensitive species. The potential for these types of effects is widely recognized and is typically addressed through a variety of operational constraints, including the following:

- Dredging using sealed ("environmental") dredge buckets
- Deployment of silt curtains around the dredging operation
- A prohibition on stockpiling of sediments on the bottom during dredging
- A requirement that hydraulic dredge intakes be operated only at or below the sediment surface
- Limiting dredging to periods of low current (e.g., tidal) flow or to periods when sensitive species or life stages are absent.

Suspended sediment concentrations are ordinarily monitored during dredging, and dredging may be halted if the quantity of suspended sediment exceeds specified limits.

Dredging exposes previously buried sediments, and the chemical conditions in those sediments may result in alterations of contaminant bioavailability relative to the pre-dredging surface. Exposure of previously buried elevated concentrations clearly has the most direct potential for adverse effects. Pre-dredging sampling and post-dredging confirmation sampling are intended to prevent or remedy this situation; however, there is a possibility of exposure of elevated contaminant concentrations by dredging. This possibility is greatest where maximum contaminant concentrations occur in subsurface sediment.

Differences in chemical or physical conditions at the newly exposed surface may also affect bioavailability. A potentially important chemical difference is the oxidation state; previously buried sediments are likely to be in a chemically reduced condition, and would become oxidized upon exposure. Divalent metals, which may be bound to sulfide under reducing conditions, can be released upon oxidation. Other chemical effects may also occur, although the nature and severity of these effects are not well known.

Destruction of the existing biotic community is an immediate impact of dredging. The severity, or importance, of this impact depends upon the value of that community and the time that may be required for it to be replaced. As discussed below, dredging may also alter the habitat in such a way that the original community cannot be restored. Removal of a healthy benthic community can also have harmful impacts on higher trophic level organisms (e.g., fish and birds) that feed on that community.

Soft-bottom benthic communities generally show substantial recovery in 3−5 years. However, if eelgrass, kelp, or other rooted plants are present, more time may be required for them to become reestablished and to mature to a point that they can sustain the original community.

Dredging ordinarily alters habitat suitability in a number of ways that can affect the health or type of biotic community that can become established after dredging:

- Increased water depth, with concomitant changes in pressure, temperature, and light penetration
- An exposed surface that has substantially different physical characteristics than the original surface (e.g., grain size, organic chemical content)
- An increased sediment deposition rate, as a consequence of the stilling effect of deeper water
- Removal of physical structures, such as boulders, logs, and pilings, resulting in an absence of anchoring points or shelter for some fauna.

Thus, the short-term effect of destruction of the biotic community may be accompanied by long-term alterations in habitat suitability. The post-dredging benthic community may therefore differ from the communities found in appropriate site-specific reference locations.

15.1.3.2 Types of Dredging

Hydraulic dredges are usually barge-mounted systems that use centrifugal pumps (which may be either barge-mounted or submersible) to remove and transport the sediment and water mixture via a pipeline to a barge or disposal facility. The cutterhead dredge, a type of hydraulic dredge, uses a mechanical device (called a cutterhead) to dislodge the sediment (U.S. EPA 1993d). Resuspension at the cutterhead is a common problem for a hydraulic dredge working in fine-grained sediment. For example, certain hydraulic dredges, such as the "clean-up," "matchbox," "refresher," or "modified dustpan," typically add an enclosure around the suction end of the dredge to reduce resuspension of sediment. Hydraulic control devices, such as floating silt curtains or containment booms, may also be used in conjunction with dredging to minimize the resuspension and dispersal of sediment particles. These control devices may be arranged in multiple concentric rings or other combinations to improve containment of resuspended sediments. The effectiveness of hydraulic controls depends in part upon wave activity, tides, currents, and other hydrodynamic conditions at the site.

Hydraulic dredging allows for a relatively fast rate of sediment removal, and in particular can remove sediment from areas where structures would impede mechanical dredging. However, it adds a significant volume of water to dredged material, which can complicate *ex situ* treatment and disposal. In particular, because hauling via truck or train to upland disposal facilities requires that the material pass the "paint filter" test (an index measure of overall dryness), hydraulically dredged material requires a lengthy time period and extensive area for stockpiling and dewatering prior to placement onto transport vehicles. Similarly, use of an onsite CDF would be feasible only for hydraulically dredged sediment if it had sufficient retention area to allow time for fines to settle out within the CDF before the water leaves the enclosure as effluent. The NASSCO and Southwest Marine shipyards are poorly suited to any significant amount of hydraulic dredging because they have very limited upland space for sediment storage and dewatering. As such, hydraulic dredging is not retained for further review.

A mechanical dredge uses equipment such as a clamshell bucket to excavate material from the bottom and haul it to the surface, where it is placed directly into a confined disposal area, or placed into a barge for in-water disposal, or placed into a barge and then offloaded and transferred to a truck to be hauled to a disposal site. The mechanical dredging process adds substantially less water to the dredged sediment relative to hydraulic dredging. Mechanical dredging generally operates in a manner that leads to higher resuspension rates in the water column, but the use of engineered controls (i.e., enclosed buckets and silt curtains) can reduce sediment suspension and mobilization.

A variation of the conventional bucket, the enclosed dredge bucket, has been developed to limit spillage and leakage from the bucket (Hartman and Goldston 1994). Enclosed bucket dredges are most effective for dredging very soft sediments because of their relatively lighter weight. They have been used routinely in various Great Lakes ports for the maintenance of navigation channels. They have also been used in sediment remediation projects in the Black River near Lorain, Ohio, in 1990; the Sheboygan River, Wisconsin, in 1990 and 1991; the Brazos River channel in Freeport, Texas, in 1992; the Saginaw River, Michigan, in 2000; and Ward Cove, Alaska, in 2001. Other mechanical dredges, such as backhoes (excavators) or dipper dredges, can be used for removing problem sediments under certain circumstances. Mechanical dredging would not be feasible in underpier areas.

Mechanical dredging is retained for further analysis.

15.1.4 Treatment

Sediment can be treated in a variety of ways, ranging from simple dewatering techniques to more elaborate treatment technologies that are designed to immobilize or eliminate hazardous constituents. Treatment of sediment with elevated concentrations of hazardous substances will depend on the physical characteristics of the sediment, specific contaminants, and levels of contamination. Potential sediment treatment technologies and process options are similar to those used for upland solid waste (either soil, sludge, slag, or debris). The main differences between marine sediment and soil are that marine sediments are mixed with saltwater, and the sediments have much higher initial water content than upland soil.

Treatment technologies considered for use at the shipyard sites include dewatering, thermal desorption, thermal destruction (incineration), immobilization, sediment washing, and biological and chemical treatment. Although soil can be treated in place, *in situ* treatment of sediment is rarely done because of the difficulty in working under water. According to Swatko and Berry (1989), it may not be possible to effectively treat sediment in water depths exceeding 20–30 ft because of the difficulty in accurately controlling the treatment equipment or chemical additions. These treatment technologies are considered infeasible for *in situ* application at this site and are thus eliminated from further consideration. The following sections discuss potentially applicable *ex situ* treatment technologies for sediment that has been removed by dredging.

15.1.4.1 Dewatering

Sediments may be dewatered for remedial alternatives that involve dredging and upland disposal, or as a pretreatment step prior to additional treatment. Dewatering may be conducted using filter presses, centrifuges, settling basins, impoundments, or clarifier tanks. The determination of whether dewatering is needed and, if so, the type of dewatering, is dependent on the characteristics of the sediments, the ultimate disposal or treatment method, and the availability of necessary facilities or space. Dewatering processes remove water from dredged material to prepare it for further treatment or disposal, and the excess liquid may require treatment to meet federal, state, and regional water quality standards.

Total solids in surface sediments from the shipyard sites ranged from 29 to 62 percent (wet weight), indicating high water contents. Thus, dewatering will likely be a necessary pretreatment for dredged sediments. Bench-scale testing could provide useful information for evaluating dewatering technologies. The relative cost of dewatering is generally low to moderate. However, fine-grained material in shipyard sediments may increase the cost and decrease the efficiency of active dewatering processes. This technology is retained for further analysis.

15.1.4.2 Thermal Destruction (Incineration) and Desorption

Common incinerators include the rotary kiln, circulating fluidized bed, and infrared incinerator. Through incineration, chemicals such as halogenated and nonhalogenated volatile and semivolatile organic compounds, PCBs, pesticides, and dioxins and furans are destroyed by combustion, leaving behind heavy metals in the ash, or, in the case of volatile metals like arsenic and mercury, in the flue gas. Solids left over from the incinerator and scrubber system may require treatment and are typically placed in upland disposal facilities. Preparation of dredged sediments for incineration includes dewatering and screening to remove oversized particles from the feed stream (U.S. EPA 1993d).

Thermal desorption evaporates volatile and semivolatile organic compounds and concentrates them as vapors, whereas incineration destroys organic contaminants through combustion. Both processes generate liquid and gaseous waste streams that must be treated to meet water quality or air quality standards. Fine-grained materials in shipyard sediments may pass through thermal desorption and thermal destruction systems, causing particle loading and reducing the efficiency of these technologies, and shipyard sediments would probably require prescreening (to remove oversized material) and dewatering prior to treatment. Both technologies have relatively high costs. Because thermal destruction and thermal desorption are not effective for metals and are more costly than other *ex situ* treatments, they are eliminated from further consideration.

15.1.4.3 Immobilization

Immobilization (solidification/stabilization) reduces the leaching potential of sediment contaminants by solidifying the sediment in cement, silicate, or other fixative, and binding its chemical constituents within the matrix using various reagents. Immobilization can increase the volume of waste by more than 20 percent. Immobilization has been thoroughly tested in *ex situ* applications, where it is effective for metals and potentially less effective for organic contaminants. The salinity of marine sediments may interfere with solidification and stabilization reactions.

Ex situ immobilization is more expensive than other equally or more effective technologies, and therefore will generally not be an appropriate technology. However, solidification/stabilization may be useful for limited use in a CDF, and therefore *ex situ* immobilization is retained for that potentially limited application.

15.1.4.4 Sediment Washing

Sediment washing is a separation technology, wherein a washing process physically separates the finer grain-size fractions, with which contaminants are typically associated. Sediment washing removes metals, PCBs, and other contaminants by mechanically scrubbing dredged sediment in a wash solution containing a leaching agent, surfactant, chelating agent, acid, or base. The wash solution may then require treatment to meet federal, state, and regional water quality standards. The sediment washing process concentrates chemicals into a smaller volume through particle size separation. Sediment washing is most effective on sand and gravel; fine silt may pass through the process, and some chemicals bind strongly to clay particles, making sediment washing inefficient (U.S. EPA 1993d). The cost of implementing this technology is relatively high. Available information indicates that shipyard sediments are relatively high in silt, a characteristic that would make this process ineffective. In addition, biological testing conducted during Phase 1 indicates that sediment contaminants at the shipyards have limited bioavailability, and may also therefore be resistant to sediment washing techniques. Therefore, sediment washing is eliminated from further consideration.

15.1.4.5 Biological and Chemical Treatment

Biological treatment can effectively degrade organic contaminants, but it is ineffective at removing metals, a primary category of indicator chemicals at the shipyard sites. Chemical treatment, including oxidation and dechlorination, is also ineffective at treating metals. Therefore, the *ex situ* applications of these technologies are eliminated from further consideration.

15.1.5 Disposal

Dredged material may be deposited in ocean disposal sites, stored in confined disposal sites such as offsite landfills or nearshore or aquatic confined disposal facilities, or reused in beach replenishment and habitat restoration and enhancement projects. Dredged sediment may require treatment to meet disposal or reuse criteria.

15.1.5.1 Offsite Landfill Disposal

Dredged sediment can be disposed of in approved offsite landfills, pending landfill operator and agency approval. If the dredged material classifies as a hazardous waste under the Resource Conservation and Recovery Act (42 USC §6901 et seq.) or Title 22 of the California Code of Regulations (Division 4.5, Chapter 11), it may be discharged only to Class I waste management units. Offsite landfill disposal would involve sediment removal by dredging, treatment if necessary, stockpiling, dewatering, and loading for transport by truck, rail, or barge (or some combination) to a landfill with the capacity to accept the sediment.

Offsite landfilling has the potential to be an effective and implementable remedial option for the disposal of dredged shipyards sediment. The sediment may require dewatering or other treatment to meet the criteria for specific solid waste facilities; the sediment must be evaluated to determine if it is hazardous, designated, nonhazardous, or inert according to Titles 22 and 23 of the California Code of Regulations, and potentially evaluated against other state regulations, in order to identify which landfill classes encompass appropriate facilities for housing this material. The relative cost of offsite landfill disposal is moderate to high relative to the other disposal technologies. Offsite landfill disposal is retained for further analysis.

15.1.5.2 Nearshore Confined Disposal

CDFs are constructed adjacent to the shoreline. The problem sediment is confined using retaining dike structures that are constructed using earthen berms, steel sheetpiling, or combinations of these techniques, in the nearshore/offshore area adjacent to the uplands. The problem sediment can be placed into the CDF by a variety of methods. These methods include release from a split hull barge, direct mechanical placement, hydraulic placement via a pipeline directly from a dredge, and slurrying of mechanically dredged material in the barge with subsequent pumping over the dike into the CDF. Depending on the placement method, a temporary opening in the retaining dike may be used to allow access by the disposal barges during subsurface placement of the problem sediment. Typical retaining structures are berms (constructed with sand, sandy gravel, or other fill material) and sheet-piling structures. After the sediment has settled, the site can be filled to grade and put to a variety of uses.

CDF sites have been used successfully to contain problem sediment at many sites and are one of the most commonly used disposal options for problem sediments. The long-term integrity of the sites can be ensured with appropriate design. Design factors include physical characteristics of sediment, such as average grain size, moisture content, and settling characteristics; groundwater and tidal elevations; foundation materials for dikes; geotechnical stability of the underlying bed materials; and leachability characteristics of the contaminated sediments. These sediment and environmental properties must be characterized before a CDF can be fully evaluated. Disposal of shipyard sediments at a CDF may require mitigation if there is any loss of habitat, wetlands, and/or eelgrass, and must meet federal dredge and fill standards and comply with applicable water quality standards.

The costs of disposal at CDF sites vary widely depending on the geometry of the CDF area and its containment structure. A key indicator of the cost-effectiveness of a CDF design is the ratio between the length, size, or cost of its retaining structure(s) to its overall capacity for containing sediment. The higher this ratio, the less cost-effective the CDF will be. Although the shipyards have very limited waterside space available for construction of a CDF, this technology is retained for further analysis.

15.1.5.3 Confined Aquatic Disposal

Confined aquatic disposal (CAD) facilities are submerged areas where dredged material is placed, followed by capping material, in an aquatic disposal site. Problem sediment is placed on the bottom either in a mound, in an area enclosed by constructed berms, or within an excavated depression, with clean material then placed as a cap over the problem sediment to create a CAD site. The thickness of the cap is based upon the need to limit transport of chemical contaminants upward through the cap, to prevent biological contact with the underlying problem sediment, and to resist erosion forces. The issues associated with CAD site capping are the same as those for in-place capping: 1) obtaining a sufficient cap thickness over the entire area, 2) placing the capping material without displacing the problem sediment, and 3) maintaining long-term cap integrity. In high-energy environments or areas where navigation may disturb the cap, a suitable armor layer of gravel or rock is required. CADs are distinguished from CDFs in that CDFs are connected to the shoreline, and that the final grade of a CDF is typically high enough to allow future upland use.

CAD sites have been used successfully to contain problem sediment at many sites and, like CDF sites, are one of the most commonly used disposal options. Design factors for CAD sites include water depth, bed slopes, water column velocities, bed stability, and physical and chemical characteristics of problem sediment. Again, these sediment and environmental properties must be characterized before this technology can be fully evaluated. Shipyard sediment must also meet federal dredge and fill standards and comply with applicable water quality standards.

The cost of CAD is comparable to the cost of CDF, but depends upon replacement method, cap material, and local hydrodynamic conditions. However, there are currently no known sites that might be used for CAD in the area of the shipyards, nor in the nearby San Diego Bay area. A CAD facility that is built up above existing mudline using perimeter berms would not be feasible either within the shipyard leaseholds or beyond, because it would interfere with current navigational uses. A CAD facility that is constructed by excavating existing sediments to form a depression would require special temporary stockpiling of sediment removed from the CAD location as well as from the shipyards. Release of shipyard sediment into the CAD site may lead to resuspension of a substantial quantity of the fine sediment; limiting sediment release through placement with a dredge bucket will substantially reduce production rate and increase costs. The availability of suitable sites for construction of a CAD facility in San Diego Bay is unknown, but availability of such sites is likely to be limited. For these reasons, the CAD alternative is eliminated from further consideration.

15.1.5.4 Geotextile Bag Containment

Geotextile bag containment has been used in conjunction with mechanical or hydraulic dredging to provide temporary containment of problem sediment. In a typical process with mechanical dredging, permeable geotextile fabric is placed inside a barge and the sediment is mechanically placed onto the fabric, which is then pulled up and over the sediment and sewn shut to create a bag. For hydraulic dredging, the sediment is pumped directly into pre-sewn bags, the open end is sewn shut, and the bag is ready for disposal. The geotextile bags are custom-made for each project but are generally the length and width of the barge bottom opening. At the disposal site, a bottom dump barge is opened to allow the geotextile bag filled with sediment to fall out.

The primary purposes for using geotextile bags are to reduce spread of contaminated sediments at disposal sites and to reduce short-term water quality impacts during disposal. For geotextile bags to be successful, the material used to fill the bags must contain a sufficient proportion of solids and minimal fine materials so that the bag will dewater over time without significant risk of fabric clogging and loss of permeability. Because of the high percentage of fine-grained material observed in the shipyard sediments, geotextile bags may clog and not dewater properly. In addition, installing the geotextile bags into barges and sewing the bags together when they are full significantly increases cycle time and construction costs. Finally, there is no identifiably suitable location to place the bags on or near the NASSCO/Southwest Marine sites. Based on the limited effectiveness and high cost of applying this technology to the disposal of shipyard sediments, geotextile bags are eliminated from further consideration.

15.1.5.5 Ocean Disposal

Dredged sediment may be appropriate for disposal at a designated open ocean facility such as the LA-5 Ocean Dredged Material Disposal Site, located 11 km southwest of Point Loma (RWQCB 1994). EPA and the U.S. Army Corps of Engineers (Corps) evaluate dredged material for ocean disposal using effects-based testing as described in the "Green Book" national testing manual (U.S. EPA/Corps 1991).

Shipyard sediment must be evaluated for ocean disposal using the effects-based testing outlined in the "Green Book" national testing manual (U.S. EPA/Corps 1991), because there are no national or California sediment quality criteria. If the sediment meets EPA requirements for disposal at LA-5, the closest appropriate open water disposal site to the shipyards, then ocean disposal can be used for qualifying dredged material. Costs are generally lower than other disposal options because the direct transport by barge does not require the sediment rehandling and processing steps required for most other disposal options. Ocean disposal is retained for further consideration.

15.1.5.6 Beneficial Reuse

Dredged sediment may be used to replenish eroding beaches, such as Silver Strand beach (RWQCB 1994), if it is compatible with material on the receiving beach. To qualify for beach replenishment, the sediment must be predominantly sand, gravel, or rock; have low organic matter content; and have low levels of contaminants. The sediment is typically transported from the dredge site to the receiving beach via truck, split-hull hopper dredge, or hydraulic pipeline. Dredged sediment may also be appropriate material for wetland restoration or enhancement projects if it is of acceptable quality, remains water-saturated and reduced, and has a near-neutral pH (RWQCB 1994). Reuse of shipyard sediment in beach replenishment or habitat restoration or enhancement projects may be appropriate for coarse-grained material. The Corps requires that beach replenishment material contain mostly particles greater than 74 μm in size (sand, gravel, and rock; RWQCB 1994), and therefore shipyard sediment proposed for beach nourishment would require physical separation to remove fine-grained particles. The cost of this separation process would be relatively high. In addition, untreated shipyard sediment may not be compatible with reuse because of the presence of chemical contaminants. Therefore, reuse is eliminated from further analysis.

15.2 Summary of Retained Technologies

Based on the results of the technology screening, the following remedial technologies are retained for further evaluation: monitored natural recovery, mechanical dredging, dewatering, immobilization, offsite landfill disposal, nearshore confined disposal, and ocean disposal. Remedial alternatives may be made up of a combination of the retained technologies.

The detailed analysis of remedial alternatives will elaborate on these candidate alternatives and evaluate them for their effects on beneficial uses, and on their technical and economic feasibility, while taking into account site-specific constraints.

16 Development of Technologies

In this section, the retained remedial technologies are developed to provide conceptual-level, site-specific implementation details. These implementation details may include the locations and layouts of any required remediation equipment and support facilities; the expected size and production rates of the remediation equipment; the unit costs of remediation; and any volume constraints, sediment quality restrictions, or significant regulatory requirements associated with a technology.

16.1 Natural Recovery

Natural recovery could be implemented 1) as the entire site remedy under the monitored natural recovery alternative; 2) in combination with active remedial measures to address specific areas where the active remedial technologies are technically not feasible; or 3) to address residual contaminants remaining after the implementation of other remedial technologies.

Implementation costs for natural recovery are related to monitoring the sediments to provide confirmation that recovery is occurring. No significant regulatory requirements (other than RWQCB plan approval) or site restrictions are anticipated. Monitoring costs are expected to be approximately proportional to the area over which natural recovery is implemented. For development of a conceptual implementation model for this remedial technology, the following assumptions are made:

- Monitoring will be performed for physical, chemical, and biological parameters. The monitoring will be conducted in four separate sampling events during years 1, 2, 5, and 10. Further monitoring may be needed beyond year 10, depending on the degree to which natural recovery has occurred over the first 10 years.
- There will be one monitoring station located every 2 to 5 acres, depending on the chemical concentrations currently existing in the sediments. In general,

monitoring stations will be more closely spaced where existing concentrations are higher.

- Monitoring of physical parameters will include bathymetry and core sampling for sediment thickness and physical properties (particle size distribution, total solids, and TOC).
- Monitoring of chemical parameters will include a selected set of metals, butyltins, PCBs, and petroleum hydrocarbons.
- Monitoring of biological parameters will include amphipod toxicity tests and benthic macroinvertebrate community assessments.
- Reports will be prepared and submitted to the RWQCB after each monitoring event.

Based on these conceptual level assumptions, the estimated implementation costs for this remedial technology are $75,000 per 10 stations sampled (present worth, with a discount rate of 3 percent per year).

16.2 Dredging

As discussed previously in the screening section, mechanical dredging was retained as the preferred technology to remove contaminated sediments at the NASSCO and Southwest Marine shipyards. Mechanical dredging would be implemented in combination with other treatment and disposal options to remediate contaminated sediments in open water areas of the site.

Dredging immediately adjacent to existing piers, wharves, bulkhead walls, revetted shoreline slopes, or other structures would require special design considerations so as to avoid potential structural damage caused by removal of supporting sediments. Removal of sediments from around pile-supported structures (piers, mooring dolphins, wharves, etc.) will increase the effective, unsupported lengths of the piles, with a corresponding reduction in the axial load capacity of these elements. Similarly, dredging adjacent to bulkhead walls or revetted slopes will reduce the lateral resistance provided by the sediments, which could cause instability or an overstress of the structure. Dredging will not be used beneath piers because of access restrictions and the potential for structural damage.

Where dredged depths are shallow, a structural analysis may reveal that bulkheads or piles have adequate reserve capacity to resist design loads without the need to add riprap or other measures. However, in order to maintain a safe level of lateral restraint to bulkheads and piles where more significant dredging depths occur, the dredged sediments will (at the least) need to be replaced with a riprap berm of sufficient size to provide an equivalent lateral restraint to the sediments that were removed. In some cases, it may be necessary to offset dredging limits slightly from marine structures. In such cases, where sufficient draft depth is available, the contaminated sediments that are left behind would need to be capped with sand, and armored to protect against erosive forces.

For this feasibility-level analysis and development of cost estimates, it has been assumed that a riprap berm will be required along all sections of piers and bulkheads that are adjacent to dredging activity. More detailed design-level structural analysis will confirm this assumption, or will identify structural measures (i.e., structural upgrades or submerged sheetpile bulkheads) or modifications to dredging activity that would be required to protect existing structures.

Other site restrictions include the need to coordinate dredging schedules with ongoing shipyard operations and with seasonal dredging restrictions. The implementation of this technology will likely require regulatory approvals by the San Diego Unified Port District, the RWQCB, the California Coastal Commission, and the Corps. The conceptual implementation model for this remedial technology is based on the following assumptions:

- Dredging will be performed via mechanical means using a clamshell bucket, with BMP used to control suspended sediment transport. Floating silt curtains are a typically used BMP for dredging, although their effectiveness is limited in marine conditions, where tidal fluctuations and currents interfere with their ability to stay fully deployed throughout the entire water column.

- There are no obstructions in most of the leaseholds within the waterway (e.g., cable crossings, buried pipelines, abandoned piles) that would inhibit dredging operations. Significant obstructions exist in the leaseholds in the vicinity of former marine railways 2 and 3 at Southwest Marine, and in the vicinity of existing railways for building ways 3 and 4 at NASSCO.

- Overdredging of the sediment by approximately 1 ft will occur to account for inaccuracies in dredging and in positioning the dredging equipment.

- Sediment removed by mechanical means would be placed on a haul barge, which is transported to the disposal area. If the disposal site is upland, the dredged material would be off-loaded to an onshore staging and sediment transfer area, dewatered, and then transported to the disposal facility.

- Water quality monitoring will be conducted during dredging activity. Turbidity measurements will be taken on an ongoing basis to verify compliance with discharge permit requirements.

- Post-dredging confirmational sampling will be performed and will involve the collection of two samples per acre and prompt analysis for indicator chemicals (metals, PCBs, and petroleum hydrocarbons).

- An additional foot of dredged volume is expected to occur above and beyond the 1-ft overdredge allowance, representing the potential removal of additional material following post-dredging confirmational sampling.

- Long-term monitoring after completion of dredging will not be required.

The unit cost for open-water mechanical dredging depends on a number of factors, including water depth, sloping vs. flat mudline bathymetry, and density and hardness of sediment. For the purposes of this evaluation, and based on the assumptions cited above, the unit cost is estimated to be $6 per cubic yard for open-water dredging outside the shipyard sites, and $12 per cubic yard for dredging within the more constrained and trafficked environment of the inner shipyard. Both unit costs are estimated average rates encompassing the various mudline conditions that might be encountered at the site. Estimated costs are based on both direct quotes from dredging contractors and Anchor Environmental LLC's sediment remedial design and construction experience.

16.3 Disposal Options

Retained disposal options for sediments removed from the shipyards, including offsite landfill disposal, confined disposal, and unconfined ocean disposal, are presented below.

16.3.1 Offsite Landfill Disposal

As discussed previously in the technology screening section, dredged sediment that does not meet the requirements for open ocean disposal could be placed into approved upland landfills for disposal. The sediment must first be dewatered in order to pass the paint filter liquids test (EPA Method 9095A, Revision 1, December 1996). Given that the average percent fines identified at NASSCO and Southwest Marine is approximately 70 percent, it is anticipated that sufficient dewatering will not occur on the barge immediately following dredging. Thus, it will be necessary to identify an adequately sized, waterfront upland staging area for the offloading, stockpiling, dewatering, and subsequent reloading into trucks. The area will need to include a loop road to allow trucks to queue for loading.

In general, typical production rates for mechanical dredging require stockpile and staging areas that are on the order of 1 to 2 acres in size. The NASSCO and Southwest Marine site configurations have very limited space available for stockpiling and dewatering dredged sediments. Because of the space constraints, the staging operation would require either taking the sediment offsite for dewatering and loading, or constructing additional land space (e.g., a nearshore CDF). Also, the production rate of the dredging operation may need to be slowed to accommodate space constraints, regardless of where the staging area is established.

The temporary stockpile/dewatering area will need to be enclosed by a suitable barrier able to withstand the loading applied by the weight of stockpiled sediments, and will need to contain a dewatering collection and disposal system (as identified in permit conditions). In addition to allowing time for drainage of the dredged sediments, free liquids can also be reduced by adding lime or flyash to the sediment to help meet the paint filter liquids test requirement. The water drained from the sediment will be treated onsite to attain appropriate water quality standards and then discharged into San Diego Bay.

Dewatered sediment will be transported to landfills via surface streets by trucks. It is unknown at this time whether the sediments will be chemically suitable for disposal at an in-state landfill, because such landfills require total threshold limit concentration and soluble threshold limit concentration testing on the dredged material in accordance with California waste discharge requirements, to determine concentrations of indicator chemicals and suitability for disposal. For this feasibility evaluation, it is assumed that all dredged materials will be disposed of at the nearest regional disposal site with available landfill capacity. Accordingly, conceptual cost estimates assume a unit price of $50 per ton for transport and disposal at a regional facility (e.g., the Laidlaw landfill at Buttonwillow, California) (Figure 16-1).

Rail transport to a suitable landfill was also evaluated. Southwest Marine does not have a rail siding, and NASSCO's rail sidings are in the center of its shipyard. Because these sidings are actively used, additional rail traffic cannot be accommodated. Also, there is insufficient room for staging, stockpiling, and loading at those sidings. There are no known waterfront properties in the area that have both the requisite rail spur and sufficient area for staging. Transport to an offsite rail spur would require trucking and a secondary handling step, as well as the requisite staging space at that spur. This would result in truck traffic through the neighboring community, which would have similar impacts on the community as would trucking the material directly to an offsite landfill. As a result of these considerations, rail transport is considered infeasible for the landfill disposal technology.

Construction of an uplands staging, offloading, and dewatering area is expected to cost approximately $40,000. This is a construction cost only for the staging operation itself, and does not include the cost of construction of a nearshore CDF at the shipyards, nor does it include fees or site use charges that might be incurred if an offsite area is obtained (e.g., through leasing of a property in the area). The costs of offloading, rehandling, stockpiling, and placing the dewatered sediments into trucks, are estimated to total $14 per cubic yard.

16.3.2 Nearshore Confined Disposal

A nearshore CDF involves placing sediments designated for confinement into an enclosure constructed in shallow subtidal areas adjacent to the shoreline. A berm or sheetpile wall is constructed, dredged sediments are placed within the enclosure, and then a clean cap is placed over the sediments to isolate them. The cap would provide usable upland area for the shipyards.

In general, the selection of an appropriate confining structure for the facility is based on the following site-specific considerations: shape and layout configuration, volume capacity needs, site operational constraints (both waterside and landside), geometric characteristics of the site, and cost-effectiveness relative to other CDF designs and remediation alternatives. Given the highly active and heavily trafficked nature of the NASSCO and Southwest Marine shipyards, and the presence of numerous piers, facilities and dry dock structures, additional considerations factor into selection of conceptual designs for CDFs at the shipyards. In particular, the proximity of vessel maneuvering areas and marine facilities to any given water space at the NASSCO and Southwest Marine shipyards necessitates minimizing CDF areal size. Because a CDF can significantly alter the use of marine- and land-based facilities, it is important that a CDF be sized in such a way as to minimize this impact on operations. Furthermore, space occupied by a CDF detracts from overall habitat value of the area, another factor that weighs toward minimizing its overall size.

The high degree of use of the shipyards and their numerous marine structures also make it desirable to maintain, as much as possible, the draft depths adjacent to the CDF. This consideration gives vertical sheetpile retaining walls certain advantages over earthen berms, which slope outward from the CDF and thus occupy a larger footprint and more water depth. Sheetpile walls also have the added advantage of providing additional volume capacity within the CDF than would an earthen berm, because the berm slope would extend inward, thereby occupying volume that could otherwise be occupied by placed sediments.

Altogether, the active nature of the shipyards offers only limited feasible options for CDF construction and sediment containment. In general, existing operational berthing areas and piers do not serve as viable locations for onsite sediment confinement in a CDF, owing to the profound effects that the construction, and existence, of such CDFs would have on operations. As a result, the most viable placement options for a CDF at the NASSCO/Southwest Marine sites are those areas that are farthest removed from ongoing vessel traffic and marine-side operations. Thus, the area best suited to CDF construction at the shipyards is the region located at and near the property line separating the two shipyards, because this area is not currently used for marine operations to the same degree as other areas within the shipyards. Interruption of the current operational uses of the adjacent land area in this location would adversely affect shipyard operations during construction, but result in less severe impact than would construction of a CDF in other areas of the shipyards.

While operational requirements do not preclude construction of a CDF in the area near the shipyards' shared boundary, they do place certain limitations on the areal extent that such a facility could feasibly occupy. Specific constraints on CDF size in this area include the following:

- The need for at least 150 ft of berthing width adjacent to NASSCO's Berth 10 on Pier 12

- The need to avoid CDF construction within Southwest Marine's dry dock sump between Piers 4 and 5, to allow the dry dock to continue to be functional

- Limitations on the extent to which the CDF is extended outward from the shore, so that the ability to maneuver vessels into and out of the adjacent berths and dry dock areas are preserved.

Based on these considerations, the conceptual implementation model for this remedial technology at the shipyards is to construct one upland CDF at the property boundary of the two shipyards, extending the shoreline out by approximately 150 ft for a length of approximately 600 ft, providing approximately 62,000 yd^3 capacity. In addition, a bulkhead extension would be constructed on the Southwest Marine property. The bulkhead extension could provide an option for disposal of some additional sediments (approximately 7,500 yd^3) if the timing of the bulkhead construction coincides with dredging disposal needs. The locations and conceptual design details of these two CDFs are presented on Figure 16-2.

The overall sequence for construction of a CDF in the NASSCO and Southwest Marine shipyard areas is envisioned as follows:

- First, existing marine structures would be demolished in the area that will be occupied by the CDF. In the region discussed as most suitable for CDF construction, Southwest Marine's Pier 5 (currently non-functional) and a relatively short, unused pier on NASSCO property would require demolition. It would not be necessary to demolish the existing armored shoreline revetment, nor the majority of existing upland facilities and installations.
- The confining structure would be constructed to form the perimeter of the CDF enclosure. Although this structure could potentially take the form either of an earthen berm or a sheetpile wall, water draft needs in the adjacent areas would likely make a driven sheetpile wall option more desirable.
- Sediment would be placed from barges into the CDF enclosure by mechanical means, using a clamshell and crane. Alternatively, the material could be pumped into the CDF from the barge with a high solids pump.
- It would be possible to add cement or similar admixtures to some of the sediment prior to its placement within the CDF, to improve its structural properties. The cement or other admixtures could be added and mixed into the sediment while it is within the barge. This option would be most appropriate for sediments near the surface of the CDF, particularly in the zone of water table fluctuation (where leaching might otherwise be most likely without stabilization).

- During sediment placement within the CDF, some water would likely seep slowly out of the retaining structure over time as material is added to the enclosure, either through the granular berm or through the sheetpiling. Additional overflow water, if it occurs, would be allowed to discharge only from controlled locations along the retaining structure. The entire CDF enclosure would be monitored throughout the placement process to make sure that applicable water quality standards are maintained.

- Sediment can be placed within the CDF until it reaches the maximum height at which the potential for leaching is insignificant. Typically, this is the height at which complete and permanent water saturation is maintained, because the sediment's anoxic state in this zone helps limit leachability of most metals. At tidally influenced elevations where water saturation is transient, leaching can be a factor. However, treatment of the sediment with cement or similar admixtures can lessen the potential for leaching to a degree that allows placement of sediment at higher elevations, independent of tidal or groundwater elevations.

- When the placed sediment reaches its maximum height, clean fill material can be placed as a cover over the sediment, to complete its isolation from the environment. The cover fill would likely be placed using land-based equipment (loaders and dozers), which would push the initial lift of cover fill out ahead and over the underlying sediment, building outward from the shore to cover the entire CDF area. A contractor could elect to place the cover material from the water side of the CDF using either a clamshell, conveyor, or high solids pump. The approach the contractor uses to place the cover material would be very dependent on how the material is brought to the site. If the cover material is brought by truck, the land-based approach would likely be used; if the material is brought by barge, a water-based approach would likely be used.

- An impervious surface would be placed on top of the CDF area to prevent infiltration of surface water.

The implementation costs for this remedial technology vary considerably based on the size, geometry, and structural type of the CDF. Given the assumption that a CDF is constructed in the area of the NASSCO/Southwest Marine property line, and assuming that space constraints in this area (as discussed above) limit its areal size to approximately 750 ft long (measured along the shoreline) by about 150 ft wide (measured out from the shore), the estimated implementation costs are expected to range from approximately $6 million to $8 million for fixed costs (e.g., construction of the perimeter retaining structure and fill cover), and about $6 per cubic yard for placement of dredged sediments within the CDF.

It may be advantageous, both for structural integrity and for chemical containment, to mix cement or other admixtures into the sediment to improve its structural and chemical properties. Although the actual mix design would be determined through additional bench-scale testing, it is expected that an admixture of approximately 8 percent cement would be sufficient to help bind contaminants to the sediment, while at the same time strengthening the sediment to better support subsequent use of the land space for staging and disposal activities. The cement would be added and mixed into the sediment in the barge, and after a day of set-up time has elapsed, placed within the CDF area using a clamshell. As the stabilized material is added to the CDF, land-based equipment would be used to spread and compact the material to construct a stable base for subsequent sediment placement. This procedure would be used only for that portion of sediment volume that is used to fill the portion of the site CDF at elevations in the intertidal zone and higher, because sediment placed at depths below the intertidal zone would be fully submerged and less amenable to this treatment.

Similar to sediment caps, confinement of contaminated sediment within a CDF entails a long-term monitoring program for groundwater and surface water quality around the facility. The estimated present worth implementation costs for this remedial technology are $25,000 to $50,000 per monitoring event, covering sampling, analysis, and reporting, for a total of four sampling events (at years 1, 2, 5, and 10).

16.3.3 Ocean Disposal

Sediments eligible for ocean disposal would be disposed of at the closest open-water disposal site, the LA-5 disposal site located 11 km southwest of Point Loma (Figure 16-3). The sediment must comply with EPA's ocean dumping regulations and the Corps' permitting regulations as described in the "Green Book" national testing manual (U.S. EPA/Corps 1991). Although the existing data are not fully sufficient for a rigorous comparison against "Green Book" guidelines, it is expected that some sediments would be suitable for this disposal technique. The conceptual implementation model for this remedial technology is based on the following assumptions:

- The dredged sediment sent to the LA-5 disposal site will not require treatment before disposal
- The dredged sediment will be transported via barge to the disposal site
- Based on conversations with local contractors, and on past experience dredging and open-water disposal in this and other areas, a unit cost of $8 per cubic yard is assumed for transport and disposal of dredged sediment at the LA-5 disposal site.

17 Assembly of Remedial Alternatives

The assembly of remedial alternatives to be evaluated in the feasibility study involves matching appropriate remediation technologies with potential remedial scenarios. Three alternatives have been reviewed:

- Alternative A—Monitored natural recovery
- Alternative B—Remediation to LAET criteria (the LAET-based area is shown in Figure 12-2)
- Alternative C—Remediation to final reference pool chemical conditions.

For each of the alternatives, a set of one or more remediation technologies is assembled to address the remediation of that area. The remediation technologies selected are generally the lowest cost options available that achieve protection of beneficial uses and can be hypothetically implemented (e.g., there would be no technical or administrative restrictions that prevent implementation). Potential restrictions that would prevent the selection of an otherwise effective and less expensive technology include excessive disruption of harbor activities, disposal area capacity limits, permit restrictions, and physical site constraints.

17.1 Alternative A—Monitored Natural Recovery

Alternative A, monitored natural recovery, includes sampling to assess naturally occurring changes in sediment conditions and biological communities. This alternative also serves as a baseline alternative for comparison purposes in evaluating the costs and benefits of the other alternatives. Long-term monitoring is included in the conceptual implementation of this alternative to track sediment quality and benthic community conditions over time. The monitoring program included in Alternative A involves periodic surveys and sample collection throughout areas of the shipyard sites not otherwise subject to disturbance. Natural recovery monitoring stations would be regularly spaced throughout areas being monitored, with one sampling station for each 2 to 5 acres. Monitoring stations would be more closely spaced within leasehold boundaries. Natural recovery monitoring would be conducted in four separate events over a 10-year period (i.e., 1, 2, 5, and 10 years).

When the physical disturbances of the sediment benthic communities (i.e., in the active shipyard areas) cease, monitoring can be used to evaluate the progress of natural recovery in these areas.

17.2 Alternative B—Remediation to LAET Criteria

Alternative B addresses sediments that exceed the site-specific LAET criteria. This alternative incorporates both dredging and natural recovery. Sediments that exceed LAETs for metals, TBT, PCBs, or PAHs would be dredged. Toxic effects associated with petroleum releases diminish substantially or completely in periods of several weeks to several months (Lee and Page 1997; NOAA 2001; Page et al. 2001). Petroleum hydrocarbon compounds at the shipyards showed no toxicity to amphipods, echinoderms, or bivalves, and only a weak association (driven by three data points) with benthic community conditions. Petroleum hydrocarbons are therefore not expected to be, or remain, a concern at the shipyards. Thus, natural recovery is an appropriate method for addressing those sediments that exceed the LAET only for petroleum hydrocarbons.

The volume of sediment to be addressed in Alternative B is estimated from the depth to which sediment core sample results exceed LAET criteria plus 1 ft to account for overdredging, with an additional 1 ft contingency volume to account for potential redredging as a result of sediment resuspension (subject to post-dredge verification sampling). The total estimated dredgeable sediment volume is 75,850 yd^3, which includes formation of side slopes around dredged areas and for required dredging offsets around existing structures (as described in Section 16). As with Alternative C, the dredged sediment volume does not include sediment from areas beneath piers or within 10 ft of structures because of stability concerns.

For these sediment volumes, the conceptual level costs for disposal at an offsite, uplands landfill are comparable to those for disposal in onsite CDFs. Because both of these disposal options are generally considered viable but differ significantly in their potential effects, two different remedial alternatives are developed to address remediation to LAET criteria: Alternative B1, which relies primarily on offsite disposal of sediments, and Alternative B2, which relies on onsite disposal of sediments.

The remedial actions to be conducted under Alternative B1 (remediation to LAET criteria with offsite landfill disposal) would be as follows:

- Dredging would be performed using a mechanical dredge.
- 7,500 yd^3 of dredged sediment would be disposed of behind the Southwest Marine bulkhead extension (with an additional 1,000 yd^3 of sediment covered in-place by construction of the bulkhead extension).
- Sediment dewatering and stockpiling activities would be performed on land.
- 67,350 yd^3 of dredged sediment would be disposed of at a suitable upland disposal facility such as the Laidlaw landfill located near Buttonwillow, California. Staging of sediment for stockpiling, dewatering, and truck loading activities would be performed on a nearby property leased for the purpose.
- Areas with elevated petroleum hydrocarbon concentrations would be monitored to assess future concentrations and potential effects on aquatic life.

A conceptual layout of the Alternative B1 remedial actions is presented in Figure 17-1. Conceptual design details for each of the components that make up this remedial action are presented in Section 16. Additional details are presented in subsequent sections as needed to support the detailed evaluation of this alternative.

The remedial actions to be conducted under Alternative B2 (remediation to LAET criteria with onsite CDF disposal) would be as follows:

- Dredging would be performed using a mechanical dredge
- 7,500 yd^3 of dredged sediment would be disposed of behind the Southwest Marine bulkhead extension (with an additional 1,000 yd^3 of sediment covered in-place by construction of the bulkhead extension)
- 60,050 yd^3 of dredged sediment would be disposed of in a nearshore CDF constructed near the NASSCO/Southwest Marine property line (with an additional 7,300 yd^3 covered in-place by construction of this CDF)
- Areas with petroleum hydrocarbon concentrations over the LAET would be monitored to assess future petroleum hydrocarbon concentrations and potential effects on aquatic life.

A conceptual layout of the Alternative B2 remedial actions is presented in Figure 17-2. Conceptual design details for each of the components that make up this remedial action are presented in Section 16. Additional details are presented in subsequent sections as needed to support the detailed evaluation of this alternative.

17.3 Alternative C—Remediation to Final Reference Pool Chemical Conditions

Results of the Phase 1 and Phase 2 investigations indicate that some chemicals exceed the 95%UPL for the final reference pool designated by the RWQCB throughout the entire leasehold area of both shipyards, and outside the leasehold boundaries to the shipping channel. Remediation to final reference pool chemical conditions therefore would require that the entirety of the leaseholds (approximately 60 acres) and the entire region between the leasehold boundaries and the shipping channel (approximately 82 acres) be addressed.

The volume of sediment to be addressed in Alternative C is estimated from the depth to which sediment core sample results exceed reference criteria plus 1 ft to account for overdredging, with an additional 1 ft contingency volume to account for potential redredging as a result of sediment resuspension (subject to post-dredge verification sampling). The total estimated dredgeable sediment volume is approximately 1,200,000 yd^3, which consists of 570,000 yd^3 from within the leasehold boundaries and 630,000 yd^3 from between the leasehold boundaries and the shipping channel. This volume does not include sediment from areas beneath piers or within 10 ft of structures because of stability concerns.

Assumptions made to help develop a conceptual model of Alternative C remedial actions include the following: 1) all sediments outside the leasehold boundary except for those in the vicinity of sediment core NA-21 would be of acceptable quality for open ocean disposal, 2) a nearshore CDF would be constructed in the area of the NASSCO/Southwest Marine property line to contain dredged sediments, and 3) all remaining dredged sediments would be dewatered and disposed in an upland landfill. The remedial actions to be conducted under Alternative C therefore would be as follows:

- Dredging would be performed using a mechanical dredge.
- 575,000 yd^3 of dredged sediment would be transported by barge and disposed at the LA-5 disposal site.
- 7,500 yd^3 of dredged sediment would be disposed of behind the Southwest Marine bulkhead extension (with an additional 1,000 yd^3 of sediment covered in-place by construction of the bulkhead extension).
- 62,000 yd^3 of dredged sediment would be disposed of in a nearshore CDF constructed near the NASSCO/Southwest Marine property line (with an additional 13,900 yd^3 covered in-place by construction of this CDF).
- The remaining 537,600 yd^3 of dredged sediment would be disposed of at a suitable upland disposal facility such as the Laidlaw landfill located near Buttonwillow, California. Staging of sediment for stockpiling, dewatering, and truck loading activities would be performed either on the land area created by the property boundary uplands CDF, or on a nearby property leased for the purpose.

A conceptual layout of the Alternative C remedial actions is presented in Figure 17-3. Conceptual design details for each of the components that make up this remedial action are presented in Section 16. Additional details are presented in subsequent sections as needed to support the detailed evaluation of this alternative.

18 Detailed Evaluation of Alternatives

In this section, each set of response actions associated with the remediation alternatives is evaluated with respect to effects on beneficial uses, technical feasibility, and economic feasibility. Evaluations performed within the first two categories rely on criteria traditionally used in conducting feasibility studies under the Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA). The current guidance document used in conducting a CERCLA feasibility study is *Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA* (U.S. EPA 1988b). Evaluations performed within the third category (economic feasibility) are required by the RWQCB guidelines (RWQCB 2001), consistent with SWRCB Resolution 92-49.

Effects on Beneficial Uses. Evaluation of the effects on beneficial uses incorporates the traditional CERCLA feasibility study criterion associated with "overall protection of human health and the environment." This evaluation criterion is used to measure how an alternative would achieve and maintain human health and environmental protectiveness. It assesses whether the risk posed to potential receptors is eliminated, reduced, or controlled by the proposed remedial actions. The primary beneficial uses to be protected are aquatic life, aquatic-dependent wildlife, and human health (RWQCB 2001). The evaluation of the effects on beneficial uses is performed on both the short-term and long-term effectiveness of the proposed remedial actions:

- Short-Term Effectiveness. This criterion addresses the short-term risks posed during implementation of an alternative, the immediate environmental effects of the remedial alternative, and the potential effects on workers during remedial action.
- Long-Term Effectiveness and Permanence. Alternatives are assessed for the long-term effectiveness and permanence they afford, along with the degree of certainty that the alternative would prove successful. The assessment includes the consideration of the magnitude of the residual risk remaining at

the conclusion of the remedial activities and the adequacy and reliability of controls.

Some short-term (temporary) or even long-term (permanent) adverse effects on beneficial uses can occur as a result of remediation, and some beneficial uses may not be fully restored by remediation. These effects include the following:

- Destruction of healthy benthic communities
- Destruction of eelgrass beds (if the water depth is increased so that eelgrass cannot be reestablished, the effect is likely to be permanent)
- Resuspension and redistribution of contaminants
- Recontamination of an area by ongoing sources.

These potential adverse effects are balanced against the potential improvements that remediation would accomplish in considering the short- and long-term effectiveness of the alternatives.

Technical Feasibility. The evaluation criteria for technical feasibility are:

- Compliance with ARARs
- Implementability
- Cost.

Alternatives are evaluated to determine whether they attain chemical-specific, location-specific, and action-specific ARARs under federal and state environmental laws. A detailed review of ARARs and to-be-considered criteria is provided in Appendix P. Remedial alternatives that do not meet relevant ARARs are not considered to be technically feasible.

Implementability includes technical aspects such as constructability, operability, and reliability of technology; the ability to schedule and complete work in a reasonable time; the availability of equipment, services, and materials; and administrative feasibility in terms of permits, and rights-of-way access, etc. Remedial alternatives that are not implementable are not considered to be technically feasible.

Costs include all capital costs and operation and maintenance costs that are expected to be incurred as a result of implementing a remedial alternative. Capital costs include both direct capital costs (such as construction and disposal costs) and indirect capital costs (such as administrative, engineering/ design, permitting, and contingency costs). Costs shown here are reasonable estimates obtained from an experienced remedial design firm (Anchor Environmental LLC) and are based on quotes from dredging contractors. Comprehensive cost estimates for each remedial alternative cannot be obtained from dredging contractors until remedial design is completed and put out to bid. Costs are calculated on a present worth basis, assuming a 3 percent discount rate.

Economic Feasibility. The economic feasibility evaluation focuses on tangible (explicit), intangible (implicit or opportunity), and social costs of implementing the remedial alternatives. These effects may include the following:

- Financial and logistical effects on the shipyards and dependent economic activities
  - Effects on employment
  - Effects on taxable revenue
  - Reduced service to customers, with possible effects on transportation and national defense
- Financial, noise, safety, and quality-of-life impacts on neighborhoods
  - Effects on businesses, including both advantages from participation in remedial activities and disadvantages from side effects of remedial activities
  - Effects on local and area traffic

- Positive or negative effects on sport or commercial angling and shellfish harvesting/aquaculture.

The economic feasibility of conducting remedial activities in actively used portions of the shipyards is linked to technical feasibility. In the following sections, each alternative is evaluated relative to the criteria described above, including interactions between technical and economic feasibility. Technical and economic feasibility are contrasted with incremental changes in beneficial uses in Section 19.

18.1 Alternative A—Monitored Natural Recovery

Alternative A includes monitoring of sediment quality and benthic communities, but does not require active remediation. Chemical biodegradation; sediment accumulation, mixing, and burial; and benthic fauna recolonization are the processes that will lead to changes in aquatic life conditions. Alternative A is evaluated below relative to the criteria described above.

18.1.1 Effects on Beneficial Uses (Alternative A)

For Alternative A, the effects on beneficial uses represent baseline conditions. These effects were evaluated in detail in the investigation and risk assessment portions of this report, and are summarized below.

As discussed in Part 1 of this report and summarized in Section 13, current conditions at the site are as follows:

- Current conditions are protective of human health beneficial uses
- Current conditions are protective of aquatic-dependent wildlife beneficial uses

- Sediment toxicity and moderate alterations of benthic macroinvertebrate communities are present at several locations, but are generally not correlated with shipyard chemicals.

Sediment toxicity is not statistically associated with any shipyard-associated chemicals, and a causation analysis demonstrates that elevated concentrations of shipyard chemicals (LAET exceedances) are not the cause of current reductions in the aquatic life beneficial use. Possible causes of sediment toxicity include contributions of pesticides and potentially other contaminants from offsite sources (e.g., Chollas Creek and storm sewer discharges from areas outside the shipyard property).

Alterations of benthic macroinvertebrate communities are not related to shipyard chemicals except for a weak relationship with petroleum hydrocarbons. Benthic community alterations are associated with physical disturbance of the sediments from normal shipyard operations, and such disturbance may be an important cause of benthic macroinvertebrate community alteration.

Under the monitored natural recovery alternative these conditions are expected to remain unchanged unless and until offsite sources of contaminants are controlled and physical disturbances cease. Conditions at the shipyards are expected to remain protective of human health and aquatic-dependent wildlife.

18.1.1.1 Short-Term Effectiveness (Alternative A)

In the short term, there would be no additional risks to workers, the public, or the environment under this alternative, because no new construction activities would occur.

18.1.1.2 Long-Term Effectiveness and Permanence (Alternative A)

This alternative would provide reduction of contaminant levels by the process of natural recovery. Natural recovery will occur through breakdown of organic chemicals and through burial and dilution of chemical concentrations by newly deposited sediment. Sediment deposition rates in San Diego Bay are approximately 1 cm per year (Peng et al. 2003). Because toxic effects on the benthic community are evidently associated with offsite sources, natural recovery is not expected to be complete unless and until offsite sources of contaminants are controlled. In addition, physical disturbance of the benthic community in areas of active shipyard operations are expected to continue indefinitely.

As discussed in Section 1, the shipyards have incorporated extensive pollution prevention mechanisms to eliminate the possibility of direct releases of contaminants. These measures include collection and treatment of all rainwater and other liquids released within the shipyards paved areas, with subsequent discharge to the sewer system; onsite treatment of bilge and ballast water; the implementation of BMPs; and training of all personnel in ongoing pollution prevention practices. Therefore, future contribution of contaminants from shipyard sources is unlikely.

If offsite sources were to be controlled, natural recovery of the benthic community would be expected to occur in as little as 3–5 years in areas where physical disturbance is absent, but potentially longer depending on the persistence of pesticides in the sediment.

18.1.2 Technical Feasibility (Alternative A)

In the following sections, Alternative A is evaluated relative to ARARs, implementability, and cost criteria.

18.1.2.1 Compliance with ARARs (Alternative A)

No short-term or long-term exceedances of chemical-specific ARARs (e.g., water quality standards) are anticipated. Also, because no active remediation measures would be implemented, Alternative A is expected to comply with action-specific and location-specific ARARs.

18.1.2.2 Implementability (Alternative A)

Alternative A is readily implementable, because the primary activities are sampling, monitoring, analysis, and reporting, with no remedial construction activities (such as dredging, disposal, CDF construction) required. It is expected that sampling can be effectively accomplished at the shipyards.

18.1.2.3 Cost (Alternative A)

Estimated costs for this alternative were independently prepared by Anchor Environmental LLC and are presented in Table 18-1. Alternative A is estimated to cost approximately $900,000. It comprises four separate long-term monitoring events, assuming the overall distribution of sampling stations described in Section 17.1.

18.1.3 Economic Feasibility (Alternative A)

Alternative A represents baseline conditions with respect to economic feasibility. It would create neither a positive nor a negative effect on area jobs, tax base, or commercial, recreational, or industrial use of aquatic resources. This alternative would have no major financial or logistical impacts on the shipyards.

- Financial and logistical effects on the shipyards and dependent economic activities:
  - No changes in employment at the shipyards are expected with this alternative
  - No significant changes in taxable revenue are expected with this alternative
  - No effects on shipyard customers is expected with this alternative.

- Financial, noise, safety, and quality-of-life impacts on neighborhoods:
  - Because there is no active remediation, no significant effects on local and area traffic and area businesses are expected.
- No effects are expected on sport or commercial angling or on shellfish harvesting/aquaculture. Commercial and sport fishing, shellfish harvesting/aquaculture, and recreational uses are all prohibited within the security boom at the shipyards and are not impaired in any case.

18.2 Alternative B1—Remediation to LAET Criteria with Offsite Disposal

Alternative B1 involves dredging areas of sediment where exceedances of LAET-based criteria occur and disposing of the dredged material in an offsite landfill. Exceedances based upon petroleum hydrocarbons only (not other site chemicals) would be addressed by natural recovery mechanisms. This alternative is evaluated below.

18.2.1 Effects on Beneficial Uses (Alternative B1)

For Alternative B1, effects on beneficial uses are evaluated by assessing the changes to baseline conditions that are expected to occur resulting from the implementation of these remedial measures.

The removal of sediments from portions of the leasehold would eliminate some areas shown to have moderate toxicity to the benthic community and some areas where benthic macroinverte-brate communities differ from reference conditions.

This alternative would have a positive effect only if after completion of dredging, those locations recover to the equivalent of reference conditions. However, because existing sediment toxicity is very likely caused by continuing offsite sources, and because current alterations of the benthic community are attributable, at least in part, to physical disturbance, any benefits from sediment removal are likely to be temporary. A gradual return to approximately the baseline conditions is to be expected as sediment from neighboring areas is redistributed and contaminants from urban runoff in Chollas Creek and storm water discharges are continually introduced to the site.

Because there are currently no adverse effects on aquatic-dependent wildlife or human health at the site, the sediment removal would not result in any improvement of beneficial uses.

Potential negative effects of sediment removal in areas where exceedances of LAET-based criteria occur, are as follows:

- Immediate destruction of many of the existing mature benthic macroinvertebrate communities. Long-term alterations in benthic communities may result from different physical characteristics of the sediment surface after dredging.
- Immediate and potentially permanent destruction of many of the eelgrass beds in both shipyards.
- Remediation activities would pose some risk to human health, primarily from transportation through the community needed to transport the sediment to an offsite landfill. This risk is discussed in further detail in the following subsections.

18.2.1.1 Short-Term Effectiveness (Alternative B1)

The evaluation of short-term effectiveness considers both beneficial and adverse changes during and immediately after the period of active remediation.

In the short term, this alternative would result in destruction of many of the benthic macroinvertebrate communities and eelgrass beds. Epibenthic organisms (e.g., fish and lobsters) that feed on benthic macroinvertebrates or that use the eelgrass beds as nurseries might also be affected, because the site might not provide the resources they need. The destruction of benthic macroinvertebrate communities and likely absence of epibenthic fish might also cause short-term effects on some aquatic-dependent wildlife that feed at the site.

Dredging is expected to release some sediment to the water column. Water quality protection measures and monitoring would need to be implemented during remediation to minimize potential effects to the environment. Special procedures and equipment may need to be used to reduce the resuspension of sediments (i.e., slower production rates, removal of debris only if no portion is buried in the sediments). Silt curtains may be used around the dredging zone to contain suspended sediment.

Under this alternative, there would be short-term human health risks associated with the remedial construction and with transportation, both for remediation workers and for the public. All remediation workers involved with activities associated with handling sediments would need to comply with Occupational Safety and Health Administration (OSHA) health and safety regulations. However, risks remain for potential injury or fatality from safety hazards associated with working on the water and with heavy equipment, and those associated with transport of materials by truck to an offsite landfill.

Handling and transport of the dredged material would have some effect on the public, primarily as a result of impacts to traffic, businesses, and jobs. Transport to a landfill would generate substantial truck traffic through the community, and the concomitant impacts including exposure to dust, noise, and truck emissions, as well as the potential for truck-related accidents. These impacts are described further below:

- **Traffic.** Approximately 67,350 yd^3 of sediment would be disposed of at an offsite landfill under Alternative B1. Transport of sediments using trucks with a capacity of 15 yd^3 would result in approximately 8,980 truck trips (4,490 loaded and 4,490 returning empty) on heavily used city streets that transport thousands of community residents, workers, and Navy personnel.
- **Accidents.** Given a distance of 250 miles to the nearest regional disposal site with available capacity, the total round trip distance for truck traffic would be

2,245,000 miles. The accident risk for non-hazardous material shipments by truck is 7.3×10^{-7} per mile, the fatality rate per accident is 3.95 percent, and the non-fatal injury rate per accident is 86.5 percent (Battelle 2001). Consequently, the volume of truck traffic required for offsite landfill disposal is expected to result in two truck accidents with 1 to 2 injuries and an 8 percent chance of a fatality. Truck traffic to an upland disposal site would transit Sampson Street, which is used daily by thousands of civilian and Navy pedestrians to access Southwest Marine, Kelco, and Continental Maritime. The risk of accidental injury may therefore be greater than is indicated by Battelle (2001). Additional risks are associated with dredging and dewatering activities, so that the overall impact on human health of remediation to LAET criteria would be higher than the estimate based solely on transportation risks.

- **Noise.** With the number of trucks passing through the community every hour, there would be an ongoing noise impact over the course of the work.

- **Air Quality.** Diesel emissions from the trucks would have an effect on aesthetics, health, and quality of life. Health effects resulting from air quality impacts could result in some incremental health care costs that would be borne by the community. Approximately 386,800 g (852 lb) of particulate emissions would be released per month from the trucks (calculations based on 200 trucks per day (100 loaded and 100 returning empty), 250 miles each way to the nearest landfill with available space (500 miles total round trip), and idling time and emissions factors from CARB [2000] and U.S. EPA [1998b]). Diesel emissions from dredging equipment will add to this particulate load.

18.2.1.2 Long-Term Effectiveness and Permanence (Alternative B1)

Over the long term, benthic macroinvertebrate communities are expected to become re-established in areas where they were removed by dredging, and aquatic-dependent wildlife are expected to then be able to resume using the site for foraging.

Changes in habitat could result from an increase in bottom depths and changed substrate characteristics following dredging. Although benthic macroinvertebrate communities may be reestablished in 3–5 years, the type of fauna present is likely to be considerably different from current conditions and also to be different from reference conditions. Eelgrass is currently found primarily in areas with water depths less than 10 ft and may not be able to reestablish itself in the deeper water that would exist in the dredged areas. Alteration and loss of some of these resources may affect aquatic-dependent wildlife. Lost eelgrass beds would not be available as nursery areas for juvenile fish and other species, and the greater water depths and changed benthic communities may provide fewer feeding opportunities for epibenthic feeders such as diving birds. Reconstruction or restoration of eelgrass beds would be required but may not be successful.

With respect to modification of sediment chemical concentrations, the effectiveness of remediation to LAET criteria is expected to decline over the long term. This long-term decrease in effectiveness is a consequence of likely sediment recontamination. Although all industrial and surface water discharges from the shipyards are controlled, Chollas Creek and storm drains leading from city streets beyond the shipyard property are primary sources of recontamination. Over the long term, tidal currents and ship traffic are expected to resuspend and redistribute nearshore sediments, so that sediment chemistry concentrations in the shipyard leaseholds would gradually increase from the levels present immediately after dredging.

18.2.2 Technical Feasibility (Alternative B1)

In the following sections, Alternative B1 is evaluated to determine whether it meets all of the criteria of ARARs, implementability, and cost, which are described earlier in this section.

18.2.2.1 Compliance with ARARs (Alternative B1)

No long-term exceedances of water quality standards are anticipated; however, short-term localized exceedances are possible with this alternative during dredging activities. Measures would be taken during remediation to minimize water quality effects. Dredging activities would be conducted during the dredging window for San Diego Bay and federal dredge and fill standards would be followed (e.g., obtaining an appropriate Corps permit). Measures would be taken to prevent spills or runoff associated with dewatering dredged sediments. Compliance with ARARs associated with disposal in an offsite uplands landfill would be achieved. Workers who handle the contaminated dredged sediments would comply with all OSHA health and safety requirements. Alternative B1 would achieve compliance with ARARs.

18.2.2.2 Implementability (Alternative B1)

Approximately 74,850 yd^3 of sediment would be dredged under Alternative B1, the majority of it from within leasehold boundaries. Dredging would need to be scheduled around berth and dry dock use. The following berths and dry docks would be affected:

- The landward half of Southwest Marine's Pier 1
- The larger dry dock and vicinity of the associated pier at Southwest Marine
- Landward portions of Southwest Marine's Piers 3 and 4
- NASSCO's Berth IX and floating dry dock
- Entry and exit of vessels to and from Building Ways 3 and 4.

Interactions between access restrictions, schedule, and economic impact all have an interrelated effect on implementability, and thus must be considered together. Restrictions on access to berth and dry dock areas resulting from active shipyard operations and Navy security requirements would require that dredging be conducted on an intermittent basis, which would substantially extend the dredging schedule. Such a schedule, however, would make successful site remediation difficult, because of sediment resuspension, mixing, and deposition that would occur in areas dredged previously. Alternatively, if shipyard operations were interrupted to allow dredging to occur over a shorter period of time (to minimize sediment redistribution), there would be significant economic impacts on the shipyards and related businesses and employment. Because of the access and security issues for berths and dry docks, obtaining access for dredging would force berths and dry docks to be held open, resulting in delays in performance and breaches of contracts for shipbuilding and ship repair, layoffs of employees, and ripple effects in the local economy and local businesses. The cumulative effect of these considerations makes Alternative B1 extremely difficult and perhaps impossible to implement.

Another key obstacle involved with Alternative B1 is getting the dredged material dewatered and onto trucks for shipment to uplands disposal, given the limited space for such activities at the site. Although dewatering and transportation of sediment involves proven and readily available technologies, the limited space at the shipyards imposes substantial logistical constraints on the staging and loading of materials. Assuming that an acre of land could be made available at the shipyards, the overall processing and dewatering rate would be limited to an estimated 1,500 yd^3 a day, with a corresponding limitation in the allowable dredging rate. (Dredging 74,850 yd^3 at this rate would require approximately 50 workdays, without considering additional delays that would be caused by occupied berths and dry docks.) Alternatively, a stockpiling and staging area could be located offsite, or dewatering could be accomplished by addition of lime or similar dewatering agent to sediment directly within the barge. Use of an offsite stockpiling and treatment area will shift certain effects to other parts of the community. Sediment would have to be transported by barge to an offsite treatment area, increasing the time required to complete dredging, increasing cost, affecting vessel traffic in the bay, and limiting other uses of the offsite treatment area.

18.2.2.3 Cost (Alternative B1)

Estimated costs for this alternative were independently prepared by Anchor Environmental LLC and are presented in Table 18-2. Although Alternative B1 may be unimplementable, for the purposes of comparison, costs have been developed under the assumption that implementation could be completed with little interruption or delay. Based on this unrealistic assumption, Alternative B1 is estimated to cost approximately $14,800,000. Actual costs would be significantly higher because this estimate does not include costs for idle dredging equipment during standby periods, mitigation costs for habitat area covered by the CDF, or roadway modifications to support truck loading.

18.2.3 Economic Feasibility (Alternative B1)

Alternative B1 would have adverse economic impacts on the shipyards, the shipyards' customers, local businesses, and the community. These impacts as well as potential impacts on utilization of aquatic resources are described below. Incremental costs and benefits of this alternative are compared to other alternatives in Section 19.

18.2.3.1 Impacts on the Shipyards and Dependent Economic Activities (Alternative B1)

Without restrictions on dredging, a major negative effect on employment would be expected at the shipyards, because shipyard production would have to be curtailed or delayed during CDF construction and during dredging in operational areas of the yards. These job losses would have a ripple effect on other businesses and the economy of the area.

Both NASSCO and Southwest Marine perform strategically important ship maintenance, repair, and modernization work and are currently performing important multiyear contracts for both military and commercial customers. The ships under construction play vital roles in national defense and in transporting crude oil under improved environmental conditions. Delays or interruptions in the delivery of these ships, would have potentially broad consequences affecting important national goals.

For the Navy, NASSCO is under a long-term contract to deliver T-AKE Class ships, which deliver supplies to armed forces conducting national defense operations throughout the world. NASSCO is also building four 1.3 million barrel capacity commercial tankers for BP to transport crude oil from Valdez, Alaska, to oil refineries on the West Coast. These double-hull ships contain state-of-the-art environmental controls and will replace single-hulled tankers that must be phased out to meet the requirements of the Oil Pollution Act of 1990, enacted in response to the Exxon Valdez spill.

Both NASSCO and Southwest Marine conduct maintenance and repair activities on Navy and commercial vessels, collectively including all types of Navy vessels homeported in San Diego. This work is scheduled several years in advance, and shipyard berths and dry docks are generally fully utilized. NASSSCO and SWM are the only two shipyards in California that are capable of providing both dry docking and pier-side berthing for these contracts.

Interruptions and delay in ship construction activities not only would cause a breach of the schedule terms of those contracts, but would substantially drive up the costs of performing those contracts as scheduled work was disrupted and performed in later periods. Interruptions in ship repair activities would cause layoffs of shipyard employees, and would have similar potential disruptive effects on subcontractors and Navy AITs, who perform specialized onboard ship modernization activities. The shipyards could be exposed to millions of dollars of potential damages to both their customers and subcontractors. Interruptions in repair activities would have significant adverse consequences to shipyard employees, subcontractors, and Navy contractors.

Although some work could go to other shipyards, if larger contracts cannot be completed because of extensive remediation, this work would have to be done at facilities outside of California. The local tax base would also be affected, because taxable revenue from the shipyards and other local businesses would be reduced.

18.2.3.2 Impacts on Neighborhoods (Alternative B1)

Assuming continuous dredging were possible, transport of sediments to a landfill would result in truck traffic through the community of approximately 100 loaded trucks per day for 7 weeks, for a total of more than 8,980 truck trips (4,490 loaded and 4,490 returning empty). Because truck traffic through the community during off-hours is presumed to be unacceptable, an 8-hour workday is assumed for trucking. During the 8-hour workday, there would be about 26 trucks per hour (13 loaded and 13 returning empty) through the community. Figure 18-1 shows the likely trucking route from the shipyards to Interstate 5. This truck traffic could result in a variety of impacts on health, safety, and overall quality of life for the community, including:

- **Noise.** With the number of trucks passing through the community every hour, there would be an ongoing noise impact over the course of the work affecting both residences and local businesses.
- **Air Quality.** Diesel emissions from the trucks and dredging equipment would have an effect on aesthetics and quality of life, and they may negatively impact businesses as well. Health effects resulting from air quality impacts could result in some incremental health care costs that would be borne by the community. The health risk aspects of air quality were addressed in further detail in Section 18.3.1.1.
- **Service Life of Road Infrastructure.** Repetitive truck traffic may reduce the service life of road infrastructure by wearing out pavement. Ultimately, this could mean damaged roads that 1) may reduce the quality of the driving experience for residents, 2) may result in damage to vehicles, and 3) may result in a possible increase in the level of taxation and/or fees associated with road maintenance.
- **Accidents.** Accidents are likely to occur in the normal course of the transport process. The average cost of a truck accident for nonhazardous shipments is $340,000 in 1996 dollars (Battelle 2001), or about $431,000 in 2004 dollars (at a discount rate of 3 percent). For the one transportation accident expected to occur as a result of offsite landfill disposal (see discussion in Section 18.3.1.1), the economic cost is thus estimated to be $431,000.

18.2.3.3 Impacts on Utilization of Aquatic Resources (Alternative B1)

Minimal adverse effects are expected on sport or commercial angling and on shellfish harvesting/aquaculture. Commercial and sport fishing, shellfish harvesting/aquaculture, and recreational uses are all prohibited within the security boom at the shipyards and Alternative B1 does not include activities outside these areas.

18.3 Alternative B2—Remediation to LAET Criteria with Onsite Disposal

Alternative B2 involves dredging areas of sediment where exceedances of LAET-based criteria occur, and disposing of the dredged material onsite in a CDF. Exceedances based upon petroleum hydrocarbons only (not other site chemicals) would be addressed by natural recovery mechanisms. This alternative is similar to Alternative B1 except the dredged sediments would be placed in onsite CDFs instead of transported to an offsite landfill. Incremental costs and benefits of this alternative are compared to other alternatives in Section 19.

18.3.1 Effects on Beneficial Uses (Alternative B2)

For Alternative B2, effects on beneficial uses are evaluated by assessing the changes to baseline conditions that are expected to result from the implementation of these remedial measures.

Because there are currently no adverse effects on aquatic-dependent wildlife or human health at the site, the sediment removal would not result in any improvement of these beneficial uses. However, because existing sediment toxicity is believed to be caused by continuing offsite sources, and because current alterations of the benthic community are attributable, at least in part, to physical disturbance, the beneficial effects on aquatic life are likely to be temporary. A gradual return to approximately the baseline conditions is to be expected as sediment from neighboring areas is redistributed and contaminants from urban runoff in Chollas Creek and storm water discharges are continually introduced to the site.

In addition to the negative effects discussed in Section 18.2.1 for Alternative B1, the construction of the boundary-area CDF under Alternative B2 would further result in the elimination of approximately 2.5 acres of subtidal habitat within the leasehold. Mitigation of these lost subtidal areas would be required, but the lack of potential mitigation sites in the vicinity of the shipyards means that compensating habitat would most likely have to be obtained in other areas of San Diego Bay.

An advantage Alternative B2 has over Alternative B1 is that it avoids the relatively high risk to human health caused by trucking of sediment through the community.

18.3.1.1 Short-Term Effectiveness (Alternative B2)

The evaluation of short-term effectiveness considers both beneficial and adverse changes during and immediately after the period of active remediation. In the short term, Alternative B2 would result in the same destruction of benthic macroinvertebrate communities and eelgrass beds as Alternative B1. Alternative B2 would also have similar human health risks associated with dredging and sediment handling activities that Alternative B1 has, but would avoid those caused by trucking of sediment through the community.

18.3.1.2 Long-Term Effectiveness and Permanence (Alternative B2)

The long-term effectiveness of Alternative B2 is similar to that of Alternative B1 discussed in Section 18.2.1.2. Benthic macroinvertebrate communities that would be destroyed by dredging are expected to reestablish themselves, although, as a result of changes in water depth and substrate composition, the composition of these communities is likely to change. Aquatic-dependent wildlife that are dependent on benthic macroinvertebrates are expected to resume utilizing the site as the benthic communities recover. Also, redistribution of sediments from other areas and recontamination from uncontrolled offsite sources would cause sediment chemistry concentrations in the shipyard leaseholds to gradually increase from the levels present immediately after dredging.

All of the sediment that is dredged would be disposed of in engineered CDFs, where it would be permanently retained. This disposal option has good effectiveness and permanence. Monitoring would be conducted to verify long-term effectiveness and future protection of the environment.

18.3.2 Technical Feasibility (Alternative B2)

In the following sections, Alternative B2 is evaluated relative to implementability and cost criteria, which are described earlier in this section.

18.3.2.1 Compliance with ARARs (Alternative B2)

No long-term exceedances of water quality standards are anticipated; however, short-term localized exceedances are possible with this alternative during dredging activities. Measures would be taken during remediation to minimize water quality effects. Dredging activities would be conducted during the dredging window for San Diego Bay and federal dredge and fill standards would be followed (e.g., obtaining an appropriate Corps permit). Measures would be taken to prevent spills or runoff associated with dewatering dredged sediments. Compliance with ARARs associated with disposal in CDFs would be achieved. Workers who handle the contaminated dredged sediments would comply with all OSHA health and safety requirements. Alternative B2 would achieve compliance with ARARs.

18.3.2.2 Implementability (Alternative B2)

Alternative B2 is similar to Alternative B1 concerning implementability. Dredging areas and volumes are similar and there would be similar conflicts with site operations that make the successful completion of dredging operations extremely difficult and possibly unimplementable. In addition, Alternative B2 would also require finding a suitable habitat mitigation area to replace the area covered by the CDF, which could be quite difficult.

18.3.2.3 Cost (Alternative B2)

Estimated costs for this alternative were independently prepared by Anchor Environmental LLC and are presented in Table 18-3. Although Alternative B2 may be unimplementable, for the purposes of comparison, costs have been developed under the same assumption as Alternative B1 (i.e., that implementation could be completed with little interruption or delay). Based on this unrealistic assumption, Alternative B2 is estimated to cost a minimum of approximately $15,300,000. Actual costs would be significantly higher because this estimate does not include impacts on shipyard operations, costs for idle dredging equipment during standby periods, or mitigation costs for the habitat area covered by the CDF.

18.3.3 Economic Feasibility (Alternative B2)

Economic impacts on the shipyards for Alternative B2 are similar to those for Alternative B1 and were previously discussed in Section 18.2.3.1. Because all of the dredged material would be placed in an onsite CDF under Alternative B2, this alternative would have less impact on local businesses and the community than Alternative B1 because they would not be affected by truck traffic. Also, impacts on aquatic resources under Alternative B2 are similar to those expected under Alternative B1 and were previously discussed in Section 18.2.3.3.

18.4 Alternative C—Remediation to Final Reference Pool Chemistry

Alternative C involves dredging of all areas within and outside of the shipyard leaseholds, except where dredging would imperil existing shorelines and piers. Dredged sediment is anticipated to be disposed of at the LA-5 disposal site, in a nearshore CDF near the boundary between the shipyards, within the bulkhead extension (at the Southwest Marine shipyard), and at an offsite upland landfill. Alternative C is evaluated below.

18.4.1 Effects on Beneficial Uses (Alternative C)

For Alternative C, effects on beneficial uses are evaluated by assessing the changes to baseline conditions that are expected to occur resulting from the implementation of these remedial measures.

The nearly complete removal of sediments from the leasehold areas and between the leasehold boundaries to the shipping channel have both potentially positive and negative effects on beneficial uses. A potentially positive long-term effect on the aquatic life beneficial use may result from removal of all sediment from locations that currently have moderate toxicity or alterations of benthic macroinvertebrate communities. However, because existing sediment toxicity is believed to be caused by continuing offsite sources, and because current alterations of the benthic community are attributable, at least in part, to physical disturbance, any benefits from sediment removal are likely to be temporary. A gradual return to approximately the baseline conditions is to be expected as sediment from neighboring areas is redistributed and contaminants from urban runoff in Chollas Creek and storm water discharges are continually introduced to the site.

Because there are currently no adverse effects on aquatic-dependent wildlife or human health at the site, remediation to reference pool chemistry would not result in any improvement of these beneficial uses.

Potential negative effects of complete sediment removal are the following:

- Immediate destruction of all existing mature benthic macroinvertebrate communities. Long-term alterations in benthic communities may result from different physical characteristics of the sediment surface after dredging.
- Immediate and potentially permanent destruction of all eelgrass beds in both shipyards.
- The construction of the boundary-area CDF would result in the elimination of approximately 2.5 acres of subtidal habitat within the leasehold. Mitigation of these lost subtidal areas would be required, but the lack of potential mitigation sites in the vicinity of the shipyards means that compensating habitat would most likely have to be obtained in other areas of San Diego Bay, and the success of mitigation efforts is uncertain.
- Ongoing shipyard operations would also continue to physically disturb sediments in some of the leasehold areas and result in disruption of the benthic communities at these locations.

- Remediation activities would pose a relatively high risk to human health, primarily from transportation of the sediment through the community to an offsite uplands landfill. This risk is discussed in further detail in the following subsections.

18.4.1.1 Short-Term Effectiveness (Alternative C)

The evaluation of short-term effectiveness considers both beneficial and adverse changes during and immediately after the period of active remediation.

In the short term, this alternative would result in complete destruction of benthic macroinvertebrate communities and eelgrass beds. Epibenthic organisms (e.g., fish and lobsters) that feed on benthic macroinvertebrates or that use the eelgrass beds as nurseries would also be affected, because the site would not provide the resources they need. The destruction of benthic macroinvertebrate communities and likely absence of epibenthic fish would likely also cause short-term effects on some aquatic-dependent wildlife that feed at the site.

Dredging is expected to release some sediment to the water column. Water quality protection measures and monitoring would need to be implemented during remediation to minimize potential effects to the environment. Special procedures and equipment may need to be used to reduce the resuspension of sediments (i.e., slower production rates, removal of debris only if no portion is buried in the sediments). Silt curtains may be used around the dredging zone to contain suspended sediment.

Under this alternative, there would be short-term human health risks associated with the remedial construction and with transportation, both for remediation workers and for the public. All remediation workers involved with activities associated with handling sediments would need to comply with OSHA health and safety regulations. However, risks remain for potential injury or fatality from safety hazards associated with working on the water and with heavy equipment, and those associated with transport of materials by truck to an offsite landfill.

Handling and transport of the dredged material would have a significant effect on the public, primarily as a result of impacts to traffic, businesses, and jobs. Transport to a landfill would generate substantial truck traffic through the community, and the concomitant impacts including exposure to dust, noise, and truck emissions, as well as the potential for truck-related accidents. These impacts are similar to the traffic-related effects previously discussed for Alternative B1, but they are much larger because of the significantly greater sediment volume included under Alternative C.

- **Traffic.** Approximately 537,600 yd^3 of sediment would be disposed of at an offsite landfill under Alternative C, approximately 8 times the volume estimated under Alternative B1. Transport of sediments using trucks with a capacity of 15 yd^3 would result in more than 71,600 truck trips (35,800 loaded and 35,800 returning empty). Sediment processing rates would be similar to those expected under Alternative B1 (approximately 1,500 yd^3 per day); therefore, truck traffic rates would also be similar (i.e., approximately 26 trucks per hour (13 loaded and 13 returning empty) through the community). The duration of the traffic impacts would be significantly longer for Alternative C than for Alternative B1.

- **Accidents.** Given a distance of 250 miles to the nearest regional disposal site with available capacity, the total distance for truck traffic would be 17,920,000 miles. The accident risk for non-hazardous material shipments by truck is 7.3×10^{-7} per mile, the fatality rate per accident is 3.95 percent, and the non-fatal injury rate per accident is 86.5 percent (Battelle 2001). Consequently, the volume of truck traffic required for offsite landfill disposal is expected to result in 13 truck accidents. The corresponding probability of a fatality is approximately 51 percent, and an additional eleven or so non-fatal injuries are expected. If some of the sediment currently assumed acceptable for open-water disposal should instead need to be disposed of at an offsite landfill, human health risks would increase further. Because of the heavy usage of Sampson Street by employees of Southwest Marine, Kelco, and Continental Maritime, and by Navy personnel, risks from truck traffic may be even higher. Additional risks are associated with dredging and dewatering activities, so that the overall impact on human health of remediation to reference pool chemistry would be higher than the estimate based solely on transportation risks.

- **Noise.** With the number of trucks passing through the community every hour, there would be an ongoing noise impact over the course of the work.

- **Air Quality.** Diesel emissions from the trucks would have an effect on aesthetics, health, and quality of life. Health effects resulting from air quality impacts could result in some incremental health care costs that would be borne by the community. Approximately 386,800 g (852 lb) of particulate emissions would be released per month from the trucks (calculations based on 200 trucks per day (100 loaded and 100 returning unloaded), 250 miles each way to the nearest landfill with available space (500 miles total per trip), and idling time and emissions factors from CARB [2000] and U.S. EPA [1998b]). Diesel emissions from dredging equipment will add to this particulate load.

18.4.1.2 Long-Term Effectiveness and Permanence (Alternative C)

Over the long term, benthic macroinvertebrate communities are expected to become re-established in areas where they were removed by dredging, and aquatic-dependent wildlife are expected to then be able to resume using the site for foraging. Changes in habitat are likely to result from an increase in bottom depths and changed substrate characteristics following dredging. Although benthic macroinvertebrate communities may be reestablished in 3–5 years, the type of fauna present is likely to be considerably different from current conditions and also to be different from reference conditions. Eelgrass is currently found primarily in areas with water depths less than 10 ft, and may not be able to reestablish itself in some areas of deeper water that would exist after dredging. Alteration and loss of benthic communities and eelgrass beds would affect aquatic-dependent wildlife. Lost eelgrass beds would not be available as nursery areas for juvenile fish and other species, and the greater water depths and changed benthic communities may provide fewer feeding opportunities for epibenthic feeders such as diving birds. Reconstruction or restoration of eelgrass beds would be required, a process of uncertain success.

With respect to modification of sediment chemical concentrations, the effectiveness of remediation to reference pool chemistry is expected to decline over the long term. This long-term decrease in effectiveness is a consequence of likely sediment recontamination. Although all industrial and surface water discharges from the shipyards are controlled, Chollas Creek and storm drains leading from city streets beyond the shipyard property are primary sources of recontamination. In addition, because the final reference pool chemical concentrations are derived from the cleanest stations in San Diego Bay, they are not likely to be representative of nearshore conditions elsewhere along the eastern shore of San Diego Bay. Over the long term, tidal currents and ship traffic are expected to resuspend and redistribute nearshore sediments, so that sediment chemistry concentrations in the shipyard leaseholds would gradually increase from the levels present immediately after dredging.

18.4.2 Technical Feasibility (Alternative C)

In the following sections, Alternative C is evaluated relative to ARARs, implementability, and cost criteria, which are described earlier in this section.

18.4.2.1 Compliance with ARARs (Alternative C)

No long-term exceedances of water quality standards are anticipated; however, short-term localized exceedances are possible with this alternative during dredging activities. Measures would be taken during remediation to minimize water quality effects. Dredging activities would be conducted during the dredging window for San Diego Bay and federal dredge and fill standards would be followed (e.g., obtaining an appropriate Corps permit). Measures would be taken to prevent spills or runoff associated with dewatering dredged sediments. Compliance with ARARs associated with uplands disposal in an offsite landfill would be achieved. Workers who handle the contaminated dredged sediments would comply with all OSHA health and safety requirements. Alternative C can be implemented in a manner that would achieve compliance with ARARs. However, because of the volume of dredged sediment, permitting issues may substantially affect the initiation of remedial activities.

18.4.2.2 Implementability (Alternative C)

Interactions between access restrictions, schedule, and economic impact all have an interrelated effect on implementability, and thus must be considered together. Because of restrictions on access to berth and dry dock areas resulting from active shipyard operations, as well as dredging window restrictions, dredging would occur over an extended period of 5 years or more. However, a schedule of that length would be technically unimplementable, because during such an extended dredging schedule sediment would be resuspended, mixed, and redeposited in areas dredged previously. Alternatively, if dredging were conducted with fewer interruptions over a shorter period of time (to minimize sediment redistribution), there would be major economic impacts on the shipyards and related businesses and employment. Because of the access and security constraints on berths and dry docks, obtaining access for dredging would force berths and dry docks to be held open, resulting in delays in performance and breaches of contracts for shipbuilding and ship repair, layoffs of employees, and ripple effects in the local economy and local businesses. The cumulative effect of these considerations makes Alternative C technically unimplementable. These issues are further discussed in the following paragraphs.

Dredging is a proven technology, but because of constraints on overall production rates, an excessively long period of time would be required to complete this alternative. The length of time required is the result of three factors: 1) the large volume of sediment dredged under this alternative, 2) rate limitations imposed by upland staging and dewatering needs, and 3) restrictions on dredging activities in actively used berths and dry docks. The overall volume of dredging required (nearly 1,200,000 yd^3), would have to occur over an unreasonably extended period of time, and in so doing would have dramatic negative repercussions not only on shipyard activities, but also on vessel traffic and shipping activities by others in the vicinity, as well as on the surrounding community at large. In addition, Alternative C would also require finding a suitable habitat mitigation area to replace the area covered by the CDF, which could be quite difficult.

Alternative C requires dredging approximately 630,000 yd^3 from the area outside leasehold boundaries, and approximately 552,100 yd^3 from within leasehold boundaries (accounting for some sediment covered in-place below the constructed CDF). Outside of leasehold boundaries, where passing vessels are expected to be relatively infrequent, a reasonable expectation for dredging rate is about 5,000 yd^3 dredged per day. In the accessible areas within the leaseholds, the rate of full production dredging could be expected to be approximately half the production rate outside of the shipyards, resulting from a significant amount of obstructions and delays caused by passing vessels and movements of large ships into and from the shipyards. The need for uplands disposal of a large volume of sediment would further reduce dredging rates, because the area available for staging, stockpiling, dewatering, and loading is limited.

Of even greater impact on overall operations is the fact that within the shipyard leaseholds, dredging crews would experience large-scale delays, because occupied berths and dry docks prevent dredging for weeks or months while the dredging equipment and crews await access to those spaces. Based on existing contracts and projected facility use schedules, the shipyards' dry docks and berths are fully booked, with only insignificant amounts of unscheduled time.

The areas occupied by vessels or dry docks would preclude dredging work not only within the footprint of the vessel, but also would require an additional distance of 25 to 50 ft to provide sufficient clearance for repair and shipbuilding operations. However, force protection measures are required for Navy vessels and prohibit non-mission-essential vessels from approaching Navy ships. A security boom prevents unauthorized vessels from approaching closer than 300 ft. Because the presence of vessels at berths or in dry dock would prevent a dredging rig and barge from entering the area of the occupied berth and the surrounding security perimeter, in most cases this would also prevent access to areas beyond ship berths (closer to shore).

Projected shipyard schedules indicate that the open time between vessels at dry docks and berths is typically limited to only 1 to 2 weeks. This period is not likely to provide enough time for a dredging rig to move into position and complete the dredging of the area. All of the active berth and dry dock areas would need to be dredged under this alternative, and scheduled shipyard activities limit access to each of them, imposing serious dredging delays. For example, the NASSCO dry dock is fully booked for the foreseeable future (e.g., at least 5 years). Dredging of these areas would have to be performed during a period of berth inactivity or scheduled dry dock maintenance, which may require waiting months or years. Limitations on dredging during the least tern nesting season would further extend the overall remedial action time frame. As a result of these factors, more than 5 years would be required to complete the work.

The considerations discussed above indicate that at least 5 years would be needed to complete dredging operations under Alternative C, accounting for rate limitations imposed by upland disposal activities, dredging windows, and even greater potential conflicts with shipyard operations. This itself poses complications on the overall remedial action, because dredging completed during multiple construction seasons would likely compromise prior dredging work through mixing and sediment redeposition resulting from tidal currents, ship traffic, and other disturbances. The result is that areas could end up having to be passed over multiple times, which, given the difficulty of accessing berthing and dry dock areas for dredging, makes implementation of this alternative likely to be ineffective.

The overall effect of these factors is that Alternative C is technically unimplementable and therefore infeasible.

18.4.2.3 Cost (Alternative C)

Estimated costs for this alternative were independently prepared by Anchor Environmental LLC and are presented in Table 18-4. Although Alternative C is considered to be technically unimplementable, for the purposes of comparison, costs have been developed under the same assumption as Alternatives B1 and B2 (i.e., that implementation could be completed with little interruption or delay). Based on this assumption, Alternative C is estimated to cost a minimum of approximately $121,900,000. Actual costs would be significantly higher because this estimate does not include impacts on shipyard operations, costs for idle dredging equipment during standby periods, mitigation costs for habitat area covered by the CDF, or roadway modifications to support truck loading. Costs would also be higher if sediment between the leaseholds and the ship channel cannot be disposed of at LA-5.

18.4.3 Economic Feasibility (Alternative C)

Alternative C would have substantial economic impacts on the shipyards, the shipyards' customers, local businesses, and the community. These impacts as well as potential impacts on utilization of aquatic resources are described below. The incremental benefit of protecting beneficial uses compared to the incremental loss of that benefit is summarized in Section 19.

18.4.3.1 Impacts on the Shipyards and Dependent Economic Activities (Alternative C)

Economic impacts on the shipyards caused by dredging and construction activities associated with Alternatives B1 and B2 were previously discussed in Section 18.2.3.1. The impacts caused by the implementation of Alternative C will be similar but greatly magnified because of the significantly larger areas and sediment volumes involved. The implementation of Alternative C would have substantial negative economic impacts on the shipyards, the shipyard customers, local businesses, the local employment rate, and the local tax base.

18.4.3.2 Impacts on Neighborhoods (Alternative C)

Transport of sediments to a landfill would result in truck traffic through the community of approximately 200 trucks per day (100 loaded and 100 returning empty) for 45 weeks, for a total of more than 71,600 truck trips (35,800 loaded and 35,800 returning empty). Because truck traffic through the community during off-hours is presumed to be unacceptable, an 8-hour workday is assumed for trucking. During the 8-hour workday, there would be about 26 trucks per hour (13 loaded and 13 returning empty) through the community. This truck traffic could result in a variety of impacts on health, safety, and overall quality of life for the community, including:

- **Noise.** With the number of trucks passing through the community every hour, there would be an ongoing noise impact over the course of the work affecting both residences and local businesses.

- **Air Quality.** Diesel emissions from the trucks would have an effect on aesthetics and quality of life, and they may negatively impact businesses as well. Health effects resulting from air quality impacts could result in some incremental health care costs that would be borne by the community. The health risk aspects of air quality were addressed in further detail in Section 18.2.1.1.

- **Service Life of Road Infrastructure.** Repetitive truck traffic may reduce the service life of road infrastructure by wearing out pavement. Ultimately, this could mean damaged roads that 1) may reduce the quality of the driving experience for residents, 2) may result in damage to vehicles, and 3) may result in a possible increase in the level of taxation and/or fees associated with road maintenance.

- **Accidents.** Accidents are likely to occur in the normal course of the transport process. The average cost of a truck accident for nonhazardous shipments is $340,000 in 1996 dollars (Battelle 2001), or about $431,000 in 2004 dollars (at a discount rate of 3 percent). For the eight transportation accidents expected to occur as a result of offsite landfill disposal (see discussion in Section 18.2.1.1), the total economic cost is estimated to be $3.4 million.

18.4.3.3 Impacts on Aquatic Resources (Alternative C)

Few or no adverse effects are expected on sport or commercial angling and on shellfish harvesting/aquaculture. Commercial and sport fishing, shellfish harvesting/aquaculture, and recreational uses are all prohibited within the security boom at the shipyards. Outside of these areas, these uses would be affected in the short term during the course of the dredging and also during the recovery of the benthic community and higher trophic levels following dredging.

19 Comparison and Rankings of Alternatives

In this section, the effects on beneficial uses and the technical and economic feasibility of each alternative are compared with one another. The bases for these comparisons are presented in the detailed evaluations of each alternative in Section 18. The remedial alternatives are then ranked relative to their performance within each evaluation criterion and, where possible, a ranking score is given on the basis of their degree of positive or negative effect relative to a neutral baseline condition. The scale of the scoring ranges from positive 5 (major improvement) to negative 5 (major adverse effect) with 0 generally representing baseline conditions. The rankings and their associated scores are then summarized to present the overall feasibility of different remedial alternatives and to identify those alternatives that have the highest environmental benefits relative to their technical and economic impacts.

19.1 Beneficial Use Effects Comparison

As discussed in Section 18.1.1, the monitored natural recovery alternative (Alternative A) represents baseline conditions concerning effects on beneficial uses. The assessment of current site conditions performed in Part 1 of this report found that risks to human health and to aquatic-dependent wildlife at the shipyards are well within acceptable levels. Sediment toxicity and adverse effects on benthic communities that are observed at some locations are not statistically or causally related to concentrations of metals, butyltins, PCBs, or PAH. Adverse biological effects are statistically associated with pesticides from offsite sources. Also, reductions in benthic macroinvertebrate abundance are associated with physical disturbance of sediments in areas of active shipyard operations.

The positive and negative effects of the remedial alternatives can be combined with assessments of current biological conditions at the shipyards to derive quantitative estimates of improvements resulting from the remedial alternatives relative to current conditions. Current conditions regarding potential impairments of aquatic life beneficial uses are described in Section 9.2 and represented in Table 9-7 as a set of categorical descriptions of the likelihood of impairment of the aquatic life beneficial use. The greatest differences between benthic communities at shipyard and reference sites—and therefore the greatest potential impairment—is approximately a 50 percent reduction in the total number of organisms at some locations. By associating a quantitative probability of impairment with the categories shown in Table 9-7, and using the 50 percent reduction in abundance as an indication of the greatest level of impairment, an overall fractional impairment can be calculated for current conditions and for remedial scenarios.

Table 19-1 shows the calculation of overall fractional impairment for current conditions and remedial scenarios. Probabilities of 0.95, 0.75, 0.25, and 0.05, respectively, have been assigned to the potential impairment categories of “highly likely,” “likely,” “possible,” and “unlikely.” The product of these probabilities and the maximum observed impairment (50 percent reduction in abundance) produces an effective impairment factor for each of the categories. Only a fraction of stations falls into each impairment category, and multiplication of that fraction by the corresponding impairment factor produces an absolute estimate of fractional impairment for each category. Summing these impairment estimates across categories produces an estimate of the overall fractional impairment of aquatic life beneficial uses. As shown in Table 19-1, this procedure produces an overall estimate of 20 percent impairment of aquatic life beneficial uses under current (baseline) conditions.

These conditions are expected to continue indefinitely under Alternative A, and are expected to be reestablished following remediation under any of the other alternatives due to ongoing offsite sources and physical disturbance from shipyard activities. Similar analyses to quantify baseline beneficial effects for aquatic dependent wildlife and for human health are not performed here because these are within acceptable levels.

19.1.1 Aquatic Life Beneficial Uses

Because observed effects on benthic macroinvertebrate communities are likely caused either by continuing offsite chemical sources or by physical disturbance attributable to shipyard operations, there are no significant differences between any of the remedial alternatives on the long-term time frame for complete recovery of the benthic communities. However, there are significant differences between alternatives in their short-term effects. Negative short-term effects on aquatic resources will be most pronounced under Alternative C because of the nearly complete removal of eelgrass beds and destruction of the existing benthic communities. Much smaller negative effects are expected under Alternatives B1 and B2 because the dredging areas are not as extensive (approximately 8.5 acres for Alternatives B1 and B2 versus 142 acres for Alternative C) and do not extend outside of the leasehold boundaries. Rankings of the alternatives with respect to short-term effects on aquatic life (including eelgrass) therefore range from 0 (representing no change from baseline conditions) for Alternative A; –2 (representing a minor to moderate negative effect) for Alternatives B1 and B2; and –5 (representing severe effect) for Alternative C.

After active remedial measures are completed under Alternatives B1, B2, and C, recovery of the aquatic resources is expected to occur over 3 to 5 years, but is not expected to result in significant positive improvement in aquatic resources over baseline conditions.

The same approach described above to estimate baseline impairment can be used to estimate overall fractional impairment of aquatic life beneficial uses following implementation of any of the remedial alternatives. Because shipyard chemicals are not the cause of the impairments that are currently observed, and because other sources of contaminants are currently uncontrolled and physical disturbance is expected to continue indefinitely, dredging is not expected to completely eliminate beneficial use impairments. Following dredging, stations where impairment of the aquatic life beneficial use is currently "likely" or "highly likely" are presumed to still have a possibility of impairments of aquatic life beneficial uses. Stations where physical disturbance is currently present (two of the stations with highly likely impairments) are presumed to still have likely impairments of beneficial uses following dredging. These interpretations represent long-term changes, and ignore the short-term destruction of benthic communities that results from dredging. Table 19-1 shows the results of these calculations.

Following implementation of Alternatives B1 or B2, the aquatic life beneficial use is estimated to be 19 percent impaired. Following implementation of Alternative C, the aquatic life beneficial use is estimated to be 12 percent impaired. If locations that are currently "highly likely" to have effects are considered "likely" to continue to have effects after dredging (rather than assuming that effects are only "possible"), the aquatic life beneficial use impairment is estimated to be 17 percent rather than 12 percent following dredging.

In summary, baseline conditions represented by Alternative A are estimated to have a 20 percent impairment of aquatic life beneficial uses. After implementation of remedial measures aquatic life beneficial uses are estimated to continue to be 19 percent impaired under either Alternatives B1 or B2, and between 12 and 17 percent impaired under Alternative C.

Ranking of the alternatives with respect to long-term effects on aquatic life therefore produces scores of 0 for Alternative A, +1 for Alternatives B1 and B2 (representing minor improvement), and +2 for Alternative C (representing minor to moderate improvement).

19.1.2 Aquatic-Dependent Wildlife Beneficial Uses

Under Alternative C, the destruction of benthic macroinvertebrate communities and eelgrass beds will result in short-term effects on the local epibenthic organisms and on some aquatic dependent wildlife that feed at the site.

Also, the physical alteration of substrate composition and permanent changes in water depth caused by the implementation of Alternative C may result in permanent changes in the composition of benthic communities and may result in the permanent loss of some eelgrass beds. As discussed in Section 18.4.1, these changes in the benthic communities and the loss of eelgrass beds are likely to affect aquatic-dependent wildlife. Similar but smaller negative short term effects are expected under Alternatives B1 and B2 because the dredging areas are not as extensive. Long-term effects for Alternatives B1 and B2 are expected to more closely resemble baseline conditions.

Ranking of the alternatives with respect to short-term effects on aquatic wildlife beneficial uses therefore produces scores of 0 for Alternative A, –1 for Alternatives B1 and B2, and −2 for Alternative C.

Ranking of the alternatives with respect to long-term effects on aquatic wildlife beneficial uses produces scores of 0 for Alternatives A, B1, and B2 and −1 for Alternative C.

19.1.3 Human Health Beneficial Uses

Although current conditions at the site are protective of human health, there are significant differences between alternatives in their short-term effects on human health. During implementation of Alternatives B1 and B2 (remediation to LAET criteria), there will be an increase in human (worker) health risk over baseline conditions due to sediment dredging and processing activities. For Alternative B1 (upland disposal), increased truck traffic is estimated to result in one accident. The corresponding probability of a non-fatal injury occurring in that accident is 86 percent and for a fatality is approximately 4 percent.

The implementation of Alternative C (remediation to final reference pool chemistry) results in an additional increase (beyond that expected under Alternatives B1 and B2) in human worker health risk during active sediment dredging and processing activities. There would also be a significant increase in risk to workers and the general public from truck traffic; specifically, there are eight vehicle accidents estimated with a corresponding probability of seven non-fatal injuries and a 32 percent chance of a fatality.

Ranking of the alternatives in order of preference with respect to short-term effects on human health beneficial uses therefore produces scores of 0 for Alternative A, –1 for Alternative B2, −2 for Alternative B1, and −5 for Alternative C. All alternatives score 0 with respect to long-term effects on human health beneficial uses.

19.1.4 Summary of Beneficial Use Effects Rankings

A summary of the ranking scores for each of the alternatives under the beneficial use effects evaluation criteria is presented in the table below.

Comparative summary of beneficial use effects

	Alternative A	Alternative B1	Alternative B2	Alternative C
Short-term Effects				
Aquatic life	0	–2	–2	–5
Aquatic dependent wildlife	0	–1	–1	–2
Human health	0	–2	–1	–5
Long-term Effects				
Aquatic life	0	+1	+1	+2
Aquatic dependent wildlife	0	0	0	–1
Human health	0	0	0	0

For aquatic life and aquatic-dependent wildlife beneficial uses, the overall effect is regarded as equivalent to the long-term effect. However, for human health effects, the overall effect is regarded as equivalent to the (more serious) short-term impact, because recovery from human health impacts is not considered to take place the same way as does recovery of the benthic community. Table 19-2 integrates the expected impairments of all beneficial uses and shows the expected overall impairment. In this table, beneficial uses are represented on a percentage scale, where 100 percent represents no impairment. The estimates shown in this table represent overall effects. The relative impairments of aquatic life criteria in this table are carried over directly from Table 19-1. Other beneficial uses are generally not impaired under either current conditions or implementation of the remedial alternatives, with the exception of human health under Alternative C. An impairment of human health under Alternative C is shown, representing the increased risks of injury and death associated with dredging and disposal alternatives. As described in Section 19.1.3, this risk may be substantial, and the level of impairment shown in Table 19-2 may under-represent this risk. This disparity in the value of different beneficial uses is explicitly represented by a factor for relative value that is included in Table 19-2.

The improvement in overall beneficial uses that can be achieved by implementation of any remedial alternative is very low, on the order of a 1 percent improvement. As shown in Table 19-2, overall beneficial uses are currently at approximately 95 percent of their ideal value, and even the most drastic remedial alternative will improve this only to about 96 percent of the ideal value. Although different values could be chosen for the probability assignments in Table 19-1 and the relative values in Table 19-2, the overall conclusion is quite consistent regardless of changes in these values: implementation of any of the remedial alternatives will lead to negligible or only minor improvements in beneficial uses, even without consideration of short-term adverse effects.

19.2 Technical Feasibility Comparison

As discussed in Section 18, the evaluation criteria used to assess the technical feasibility of the remedial alternatives are 1) compliance with ARARs; 2) implementability; and 3) cost. The general findings are that all of the remedial alternatives can be implemented in a manner that will comply with ARARs, and they all rely on proven technologies and use equipment and materials that are readily available. However, there are significant logistical obstacles that make the implementation of dredging operations under Alternatives B1 and B2 nearly impossible, and the far more extensive dredging required under Alternative C is considered to be technically unimplementable. If the issues with implementability are ignored, costs for the alternatives increase dramatically in relation to the complexity of the remedial actions and the volumes of sediment dredged.

19.2.1 Compliance With ARARs

As discussed in the ARARs sections in Section 18, there are chemical-specific, action-specific, and location-specific ARARs of concern associated with the implementation each of the alternatives, but all of them can reasonably be expected to achieve compliance through proper planning and implementation. There are no distinguishing positive or negative aspects associated with this evaluation criterion to differentiate between alternatives and all are therefore given a ranking score of 0.

19.2.2 Implementability

Alternative A is the only alternative that is readily implementable and can be effectively accomplished.

Alternatives B1 and B2 involve remediation of approximately 75,850 yd^3 of sediment within areas of active shipyard operations. Dredging would need to be scheduled around berth and dry dock use, which will likely result in interruptions of dredging operations for weeks or months at a time awaiting access to areas to be dredged. Additional conflicts are likely under Alternative B2 associated with the construction of a CDF in the vicinity of the property boundary between NASSCO and Southwest Marine. Alternative B1 has the added limitation of requiring a minimum of 1 to 2 acres of space for sediment dewatering, stockpiling, and loading for truck transport to an offsite landfill. The lack of any available space at the shipyards or in the local area presents significant logistical problems as does the estimated 80 round-trip trucks per day required for transport of sediments to the landfill. These site constraints and logistical issues make the implementation of either Alternative B1 or B2 extremely difficult without significant effect on costs, the effectiveness of the remedial measures, or on economic feasibility.

Alternative C has logistical challenges similar to those of Alternatives B1 and B2, but because sediment volumes are greater by a factor of more than 10, the conflicts with shipyard operations and the adverse traffic effects on the surrounding community are greatly magnified. There will be increased logistical problems in active areas of the shipyards that were not being dredged under Alternatives B1 and B2, and there will also be additional logistical issues with San Diego Bay shipping activities associated with dredging operations outside of the leasehold boundaries. Haulage of the estimated 537,600 yd^3 of sediment to be disposed at an offsite landfill will result in extended periods over which increased traffic of approximately 80 round-trip trucks per day will occur.

Completion of all dredging under Alternatives B1, B2, and C is expected to require several years based upon current contracts and planned utilization of berth and dry dock areas. Completion of Alternative C is expected to require a minimum of 5 years. Permitting issues may also delay the start of dredging for several years. During the dredging period, sediment at the shipyards will consist of a patchwork of dredged and undredged areas. Tidal currents and ship movements will redistribute sediment from undredged areas to dredged areas. Consequently, repeated redredging would be required, further increasing the time required and the likelihood of additional sediment redistribution.

Alternatives B1 and B2 are considered to be nearly unimplementable, but because of the restricted extent of the dredging areas and volumes, there is a possibility that an implementation plan could be developed to allow remedial activities to be completed within a reasonable time frame. The magnitude of the logistical issues and conflicts associated with Alternative C make it unimplementable. Ranking of the alternatives with respect to implementability therefore ranges from 0 for Alternative A (representing baseline conditions) to –5 for Alternatives B1 and B2. Alternative C is considered unimplementable and, therefore, technically infeasible, and is not scored under this evaluation criterion.

19.2.3 Cost

Alternative A (monitored natural recovery) does not involve active remedial measures, is the least disruptive of shipyard and local area activities and is therefore the least costly (estimated costs of $900,000) to implement.

Because of the implementability issues with Alternatives B1, B2, and C, costs comparison for these alternatives cannot be estimated without making simplifying and unrealistic assumptions. Nevertheless, for comparative purposes, costs were developed for these alternatives by ignoring the conflicts with shipyard operations, and by assuming that acceptable space for materials staging and sediment drying, stockpiling, and loading activities could be found. Based on these assumptions, implementation costs are estimated at $14.8 million for Alternative B1, $15.3 million for Alternative B2, and $121.9 million for Alternative C.

Scores are not developed for the evaluation of costs because the cost estimates provide a more appropriate comparative ranking of the alternatives.

19.2.4 Summary of Technical Feasibility Rankings

A summary of the ranking scores for each of the alternatives under the technical feasibility evaluation criteria is presented in the table below. Alternative C is not scored because, as described in Section 18.4, it is both technically and economically infeasible.

Comparative summary of technical feasibility

	Alternative A	Alternative B1	Alternative B2	Alternative C
Compliance with ARARs	0	0	0	0
Implementability	0	–5	–5	Not Implementable
Cost	$0.9 million	$14.8 million[a]	$15.3 million	$121.9 million[a]

[a] Estimated costs for Alternatives B1, B2, and C are provided for comparative purposes only. These estimates are based on the unrealistic assumptions that minimal conflicts occur between shipyard operations and the remedial dredging and construction activities, and that acceptable space for materials staging and sediment drying, stockpiling, and loading activities is available.

19.3 Economic Feasibility Comparison

As discussed in Section 18, the evaluation criteria used to assess the economic feasibility of the remedial alternatives are 1) effects on shipyard business and associated economic activities; 2) effects on local businesses and neighborhood quality of life; and 3) effects on recreational, commercial, or industrial uses of aquatic resources.

Monitored natural recovery (Alternative A) represents baseline conditions concerning effects on economic feasibility. Its implementation will create neither a positive nor a negative effect on area jobs, tax base, or commercial, recreational, or industrial use of aquatic resources. Economic effects (both positive and negative) of the other remedial alternatives are directly related to the size of the active remediation activities and to the period of time over which those activities are conducted. The economic feasibility of the active remediation alternatives are also interrelated with their technical feasibility. For Alternatives B1 and B2, which were found to have significant implementability concerns, and for Alternative C, which was found to be unimplementable, the economic feasibility can be evaluated only by making assumptions that are unlikely to occur. For comparative purposes, the economic feasibility evaluation presented in Section 18 ignored cost and schedule implications and assumed that effective remediation could be achieved under Alternatives B1, B2, and C without significant conflict with shipyard activities.

19.3.1 Financial Effects on Shipyards and Associated Economic Activities

Substantial operational and economic conflicts are associated with dredging alternatives. Berth space at the shipyards is scheduled 3 to 5 years in advance of the work to be performed, and access to berth areas will limit or delay dredging activity. The locations and extent of dredging operations under Alternatives B1, B2, and C make conflicts with shipyard operations unavoidable, and would result in layoffs and harmful effects on the San Diego economy. The magnitude of these negative effects is directly related to the size and duration of the onsite activities.

The ranking score for Alternative A with respect to its financial effects on the shipyards and dependent economic activities is 0 (representing baseline). Even under the unrealistic assumptions that cost and schedule implications can be ignored, the ranking scores for Alternatives B1 and B2 are both –3 (representing moderate adverse effects), and the ranking score for Alternative C is –5. If disruption of shipyard operations were considered, Alternatives B1 and B2 would be ranked −5.

19.3.2 Quality-of-Life Effects on Neighborhoods

For those remedial alternatives that include uplands landfill disposal of sediments (Alternatives B1 and C), there will be negative financial, noise, safety, and quality-of-life effects on local businesses and the public caused by the significant increase in truck traffic on local roads. These effects will be especially pronounced under Alternative C because of the extended period (several years) over which sediment haulage will be required. The total economic cost estimated for transportation accidents alone is estimated at approximately $3.4 million for Alternative C. By comparison, the estimated economic cost for transportation accidents under Alternative B1 is approximately $0.4 million and these costs are avoided under Alternatives A and B2.

Ranking scores for the alternatives with respect to quality-of-life issues are 0 for Alternative A, −1 for Alternative B2, −2 for Alternative B1, and −5 for Alternative C.

19.3.3 Effects on Recreational and Commercial Uses of Aquatic Resources

Alternative C is the only remedial alternative that is expected to have an effect on sport or commercial angling, shellfish harvesting, or recreational uses. Remedial activities associated with all other alternatives occur only within the leasehold boundaries where these uses are all prohibited. The dredging and barging activities performed outside the leasehold boundaries under Alternative C will interrupt these activities but is not expected to have a significant effect because of the short duration of active remedial operations in this area (estimated at approximately 5–6 months) and the ability of these users to avoid these remediation operations.

Ranking scores for the alternatives with respect to effects on recreational and commercial uses of aquatic resources are 0 for Alternatives A, B1, and B2 and −1 for Alternative C.

19.3.4 Summary of Economic Feasibility Rankings

A summary of the ranking scores for each of the alternatives under the economic feasibility evaluation criteria is presented in the table below.

Comparative summary of economic feasibility

	Alternative A	Alternative B1	Alternative B2	Alternative C
Shipyards and shipyard customers	0	–3[a]	–3[a]	–5[a]
Local quality-of-life effects on businesses and residents	0	–2	–1	–5
Recreational and commercial users of aquatic resources	0	0	0	–1

[a] Estimated economic effects on shipyard and shipyard customers s for Alternatives B1, B2, and C are provided for comparative purposes only. These evaluations are based on the unrealistic assumptions that cost and schedule implications can be ignored in favor of minimizing conflicts with shipyard operations.

19.4 Feasibility Study Summary

The results of the feasibility study show that Alternative A, monitored natural recovery, is the only alternative that provides acceptable effects on beneficial uses and is technically and economically feasible. Overall, aquatic life, aquatic-dependent wildlife, and human health beneficial uses are at approximately 95 percent of ideal conditions, and active remedial alternatives will result in improvements that are minimal—on the order of only a percent or so. Thus, Alternatives B1 (offsite disposal) and B2 (onsite CDF disposal), which involve removal of sediments to the site-specific LAET criteria, provide little or no incremental benefit over baseline conditions but impose significant impacts on shipyard operations and on the local community, and do so at a high cost. Alternative C, remediation to final reference pool chemical conditions, similarly provides little long-term benefit and imposes even more severe impacts on shipyard operations and on the local community; this alternative is consequently technically and economically infeasible to implement. Because there are uncontrolled contaminant sources nearby (Chollas Creek and municipal storm drains), and because physical sediment disturbance associated with shipyard operations will continue indefinitely, sediment conditions are likely to return to current conditions even if extensive dredging were to be conducted. Monitored natural recovery is therefore the most technically and economically feasible approach to addressing current sediment conditions at the shipyards.

Part 3

References

20 References

Allen, L. 1996. What do we know about spotted sand bass? Western Outdoor News.

Allen, H.E., G. Fu, and B. Deng. 1993. Analysis of acid-volatile sulfide (AVS) and simultaneously extracted metals (SEM) for the estimation of potential toxicity in aquatic sediments. Environ. Toxicol. Chem. 12:1441–1453.

Allen, L.G., T.E. Hovey, M.S. Love, and J.T. Smith. 1995. The life history of the spotted sand bass (*Paralabrax maculatofasciatus*) within the Southern California Bight. CalCOFI Reports 36:193–203.

Aller, R.C. 1982. The effects of macrobenthos on chemical properties of marine sediment and overlying water. *In:* Animal-Sediment Relations, P.L. McCall and J.J.S. Tevesz (eds.). Plenum Publishing, New York. pp. 53-102.

Alo, T. 2002a. Personal communication (e-mail to M. Chee, NASSCO, and S. Halvax, Southwest Marine, on July 29, 2002, regarding additional sampling of Bight ’98 stations). Regional Water Quality Control Board, San Diego, CA.

Alo, T. 2002b. Personal communication (e-mail to D. Nielsen, Exponent, Bellevue, WA, on August 28, 2002, regarding fish bile analyses). Regional Water Quality Control Board, San Diego, CA.

Alo, T. 2002c. Personal communication (e-mail to Shaun Halvax, Southwest Marine, and Mike Chee, NASSCO, on October 25, 2002, regarding criteria to be used for PAH analyses in pore water). Regional Water Quality Control Board, San Diego, CA.

Alo, T. 2002d. Personal communication (e-mail to Mike Chee, NASSCO, and Shaun Halvax, Southwest Marine, on October 27, 2002, regarding analysis of total organic carbon in pore water). Regional Water Quality Control Board, San Diego, CA.

Alo, T., C. Carlisle, and D. Barker. 2002. Personal communication (telephone call to D. Nielsen, Exponent, Bellevue, WA, on September 19, 2002, regarding Phase 2 fish sampling and ecological receptors). Regional Water Quality Control Board, San Diego, CA.

Ambrose, A.M., P.S. Larson, J.F. Borzelleca, and G.R. Hennigar, Jr. 1976. Long-term toxicologic assessment of nickel in rats and dogs. J. Food Sci. Technol. 13:181–187.

Anderson, B.S., J.W. Hunt, M. Hester, and B.M. Phillips. 1996. Assessment of sediment toxicity at the sediment-water interface. pp. 609–624. In: Techniques in Aquatic Toxicology. G.K. Ostrander (ed). CRC Lewis Publishers, Boca Raton, FL.

Anderson, B.S., J.W. Hunt, B.M. Phillips, R. Fairey, H.M. Puckett, M. Stephenson, K. Taberski, J. Newman, and R.S. Tjeerdema. 2001. Influence of sample manipulation on contaminant flux and toxicity at the sediment–water interface. Mar. Environ. Res. 51:191–211.

Ankley, G.T., D.M. DiToro, D.J. Hansen, and W.J. Berry. 1996. Technical basis and proposal for deriving sediment quality criteria for metals. Environ. Toxicol. Chem. 15(12):2056–2066.

ASTM. 1998. Standard guide for collecting static acute toxicity tests starting with embryos of four species of saltwater bivalve molluscs. ASTM E724-98. American Society for Testing and Materials, West Conshohocken, PA.

ASTM. 1999. Standard guide for conducting 10-day static sediment toxicity tests with marine and estuarine amphipods. Method E1367-99. American Society for Testing and Materials, West Conshohocken, PA.

ASTM. 2000. Standard guide for determination of the bioaccumulation of sediment-associated contaminants by benthic invertebrates. Method E1688-00a. American Society for Testing and Materials, West Conshohocken, PA.

Atwood, J.L., and P.R. Kelly. 1984. Fish dropped on breeding colonies as indicators of least tern food habits. Wilson Bull. 96(1):34–47.

Atwood, J.L., and B.W. Massey. 1988. Site fidelity of least terns in California. Condor 90:389–394.

Atwood, J.L., and D.E. Minsky. 1983. Least tern foraging ecology at three major California breeding colonies. Western Birds 14:57–72.

Au, D.W.T., and R.S.S. Wu. 2001. A field study on EROD activity and quantitative hepatocytological changes in an immature demersal fish. Environ. Pollut. 115(1):23–32.

Au, D.W.T., R.S.S. Wu, B.S. Zhou, and P.K.S. Lam. 1999. Relationship between ultrastructural changes and EROD activities in liver of fish exposed to benzo[a]pyrene. Environ. Pollut. 104(2):235–247.

Aulerich, R.J., and R.K. Ringer. 1977. Current status of PCB toxicity to mink, and effect on their reproduction. Arch. Environ. Contam. Toxicol. 6:279–292.

Aulerich, R., R.K. Ringer, M.R. Bleavins, and A. Napolitano. 1982. Effects of supplemental dietary copper on growth, reproductive performance and kit survival of standard dark mink and the acute toxicity of copper to mink. J. Animal Sci. 55:337–343.

Azar, A., H.J. Trochimowicz, and M.E. Maxfield. 1973. Review of lead studies in animals carried out at Haskell Laboratory—two-year feeding study and response to hemorrhage study. pp. 199–210. In: Environmental Health Aspects of Lead. D. Barth, A. Berlin, R. Engel, et al. (eds). Commission of the European Communities, Luxembourg.

Baker, J.E., P.D. Capel, and S.J. Eisenreich. 1986. Influence of colloids on sediment-water partition coefficients of polychlorobiphenyl congeners in natural waters. Environ. Sci. Technol. 20:1136–1143.

Barker, D. 2003. Personal communication (letter to M. Chee, S. Halvax, B. Chadwick, and S. Bay, dated June 9, 2003, regarding Regional Board final position on a reference pool for the NASSCO, Southwest Marine, mouth of Chollas Creek, and 7th Street Channel sediment investigations). San Diego Regional Water Quality Control Board, San Diego, CA.

Barnard, J.L., and D.J. Reish. 1959. Ecology of Amphipoda and Polychaeta of Newport Bay, California. Allan Hancock Foundation Publications Occasional Paper, No. 21. University of Southern California Press.

Battelle. 2001. Comparative Risks of Hazardous Materials and Non-Hazardous Materials Truck Shipment Accidents/Incidents. Final Report. Prepared for Federal Motor Carrier Safety Administration. Battelle, Columbus, Ohio. March, 2001. 190 pp.

Bay, S. 2001a. Personal communication (e-mail to Dreas Nielsen, Exponent, Bellevue, WA, dated June 13, 2001, regarding San Diego reference sites). Southern California Coastal Water Research Project.

Bay, S. 2001b. Summary of reference site selection for San Diego Bay sediment quality studies. Southern California Coastal Water Research Project.

Bay, S., and J. Brown. 2003. Distance-from-shore approach to identify Bight '98 reference sites in San Diego Bay. Prepared for the San Diego Regional Water Quality Control Board. Southern California Coastal Water Research Project. January 8, 2003. 9 pp.

Beyer, W.N., E.E. Connor, and S. Gerould. 1994. Estimates of soil ingestion by wildlife. J. Wildl. Manage. 58(2):375–382.

Biagianti-Risbourg, S., and J. Bastide. 1995. Hepatic perturbations induced by a herbicide (atrazine) in juvenile grey mullet *Liza ramada*, (Mugilidae, Teleostei): An ultrastructural study. Aquat. Toxicol. 31(3):217–229.

Bjorndal, K.A. 1980. Nutrition and grazing behavior of the green turtle *Chelonia mydas*. Mar. Biol. 56:147–154.

Bjorndal, K.A. 1985. Nutritional ecology of sea turtles. Copeia (3):736–751.

Bleavins, M.R., and R.J. Aulerich. 1981. Feed consumption and food passage time in mink (*Mustela vison*) and European ferrets (*Mustela putorius furo*). Lab. Anim. Sci. 31(3):268–269.

Boesch, D.F. 1977. Application of numerical classification in ecological investigations of water pollution. Special Scientific Report No. 77. Virginia Institute of Marine Science, Gloucester Point, VA.

Briggs, K.T., D.B. Lewis, W.B. Tyler, and G.L. Hunt, Jr. 1981. Brown pelicans in Southern California: Habitat use and environmental fluctuations. Condor 83(1):1–15.

Brix, H., J.E. Lyngby, and H-H. Schierup. 1983. Eelgrass (*Zostera marina* L.) as an indicator organism of trace metals in the Limfjord, Denmark. Mar. Environ. Res. 8:165–181.

Brodberg, R. 2002. Personal communication (e-mail to T. Alo, Regional Water Quality Control Board, San Diego, CA, dated August 10, 2002, regarding Phase 2 sampling). California Environmental Protection Agency, Office of Environmental Health Hazard Assessment, Pesticide and Environmental Toxicity Section, Sacramento, CA.

Brody, N. 2003. Personal communication (e-mail to D. Nielsen, Exponent, Bellevue, WA, on August 22, 2003, regarding NASSCO historical operations). LECG, LLC, Los Angeles, CA.

Cain, B.W., and E.A. Pafford. 1981. Effects of dietary nickel on survival and growth of mallard ducklings. Arch. Environ. Contam. Toxicol. 10:737–745.

Calder, W.A., and E.J. Braun. 1983. Scaling of osmotic regulation in mammals and birds. Am. J. Physiol. 244:R601–R606.

CARB. 2000. California Air Resources Board, EMFAC 2000, On-Road Motor Vehicle Emissions Inventory. California Environmental Protection Agency.

Carpenter, R. 2002. Personal communication (telephone conversation with M. Whelan, Anchor Environmental, LLC, on October 10, 2002, regarding sediment composition and dredging feasibility at the NASSCO and Southwest Marine shipyards). R.E. Staite Company.

Carr, R.S., and D.C. Chapman. 1992. Comparison of solid-phase and pore-water approaches for assessing the quality of marine and estuarine sediments. Chem. Ecol. 7:19–30.

Carr, R.S., and D.C. Chapman. 1995. Comparison of methods for conducting marine and estuarine sediment porewater toxicity tests—extraction, storage, and handling techniques. Arch. Environ. Contam. Toxicol. 28:69–77.

CDFG. 2001a. The status of rare, threatened, and endangered animals and plants in California, California least tern. www.dfg.ca.gov/hcpb/species/jsp/ssc_result.jsp?specy=birds&query= Sterna%20anitllarum%20browni. Accessed on October 22, 2001. California Department of Fish and Game, Habitat Conservation Planning Branch, Sacramento, CA.

CDFG. 2001b. The status of rare, threatened, and endangered animals and plants in California, California brown pelican. www.dfg.ca.gov/hcpb/species/jsp/ssc_result.jsp?specy=birds&query= Pelecanus%20occidentalis Accessed on October 22, 2001. California Department of Fish and Game, Habitat Conservation Planning Branch, Sacramento, CA.

Chee, M. 2003. Personal communication (e-mail to D. Nielsen, Exponent, Bellevue, WA, dated September 3, 2003, regarding diesel spills in San Diego Bay). NASSCO, San Diego, CA.

Chew, C. 1996. Toxicity and exposure concerns related to arsenic in seafood: An arsenic literature review for risk assessments. Prepared for USEPA Region 10 ESAT, Seattle Washington as Technical Instruction Document 10-9601-815 under Work Unit document 4038.

Cogswell, H.L. 1977. Water birds of California. University of California Press, Berkeley, CA.

Cornwell, J.C., and J.W. Morse. 1987. The characterization of iron sulfide minerals in anoxic marine sediments. Mar. Chem. 22:193–206.

Crooks, J. 1996. The population ecology of an exotic mussel *Musculista senhousia*, in a southern California bay. Estuaries 19(1):42–50.

Dahlgren, R.B., R.L. Linder, and C.W. Carlson. 1972. Polychlorinated biphenyls: Their effects on penned pheasants. Environ. Health Perspect. 1:89–101.

Davis, A., R. Barale, G. Brun, et al. 1987. Evaluation of the genetic and embryotoxic effects of bis(tri-*n*-butyltin)oxide (TBTO), a broad-spectrum pesticide, in multiple *in vivo* and *in vitro* short-term tests. Mutat. Res. 188:69–95.

DiToro, D.M., J.D. Mahoney, D.J. Hansen, K.J. Scott, A.R. Carlson, and G.T. Ankley. 1992. Acid volatile sulfide predicts the acute toxicity of cadmium and nickel in sediments. Environ. Sci. Technol. 26(1):96–101.

Donohue, J.M., and C.O. Abernathy. 1999. Exposure to inorganic arsenic from fish and shellfish. In: Arsenic Exposure and Health Effects. W.R. Chappell, C.O. Abernathy, and R.L. Calderon (eds). Elsevier Sciences, BV.

Dourson, M. 1993. Modifying uncertainty factors for noncancer endpoints. Advanced Topics in Risk Assessment, Society of Toxicology 1993 Annual Meeting, New Orleans, LA.

Drifmeyer, J.E., G.W. Thayer, F.A. Cross, and J.C. Zieman. 1980. Cycling of Mn, Fe, Cu, and Zn by eelgrass, *Zostera marina* L. Am. J. Botany 67:1809–1096.

Dunning, Jr., J.B. (ed). 1993. CRC handbook of avian body masses. CRC Press, Boca Raton, FL.

Dutton, D.M., and P.H. Dutton. 1997. Accelerated growth in San Diego Bay green turtles. pp. 175–176. In: Proc. Seventeenth Annual Sea Turtle Symposium. S.P. Epperly and J. Braun (eds.) NOAA Tech. Memo. NMFS-SEFSC-415. U.S. Department of Commerce.

Dutton, P., and D. McDonald. 1992. Tagging study of sea turtles in San Diego Bay, 1990–1991. pp. 35–38. In: Proc. Eleventh Annual Workshop on Sea Turtle Biology and Conservation. M. Salmon and J. Wyneken (eds). NOAA Technical Memorandum NMFS-SEFSC-302.

Ecology. 1995. Progress re-evaluating Puget Sound apparent effects thresholds (AETs). Draft Report. Prepared for the Puget Sound Dredged Disposal Analysis (PSDDA) Agencies. Washington Department of Ecology, Olympia, WA. March 1995.

Edmonds, J.S., and K.A. Francesconi. 1993. Arsenic in seafoods: Human health aspects and regulations. Mar. Pollut. Bull. 26(12):665–674.

Eisler, R. 1994. A review of arsenic hazards to plants and animals with emphasis on fishery and wildlife resources. pp. 185–259. In: Advances in Environmental Science and Technology, Volume 27: Arsenic in the Environment, Part II: Human Health and Ecosystem effects. J.O. Nriagu (ed). John Wiley and Sons, Inc., New York, NY.

ENV America. 2002. Workplan for site assessment, former wastewater ponds, Silver Gate power plant, 1348 Sampson Street, San Diego, California. Prepared for San Diego Gas & Electric Company, Los Angeles, CA. August 21, 2002. ENV America, Inc., San Diego, CA. 25 pp.

Exponent. 2001. Work plan for the NASSCO and Southwest Marine detailed sediment investigation. Prepared for NASSCO and Southwest Marine, San Diego, CA. Exponent, Bellevue, WA.

Exponent. 2002. Field sampling plan addendum for the NASSCO and Southwest Marine detailed sediment investigation. Prepared for NASSCO and Southwest Marine, San Diego, CA. Exponent, Bellevue, WA. September 2002.

Fahraeus-Van Ree, G.E., and D.R. Spurrell. 2000. Effect of toxaphene on isolated hepatocytes of the yellowtail flounder, *Pleuronectes ferrugineus storer*. Ecotoxicol Environ. Saf. 46(3):238–297.

Fairey, R., C. Bretz, S. Lamerdin, J. Hunt, B. Anderson, S. Tudor, C.J. Wilson, F. LaCaro, M. Stephenson, M. Puckett, and E.R. Long. 1996. Chemistry, toxicity and benthic community conditions in sediments of the San Diego Bay region. Final Report. California State Water Resource Control Board, Sacramento, CA.

Ferraro, S.P., and F.A. Cole. 1997. Effects of DDT-sediment contamination on macrofaunal community structure and composition in San Francisco Bay. Mar. Biol. 130:323–334.

Francois, R., F.T. Shrot, and J.H Weber. 1989. Accumulation and persistence of tributyltin in eelgrass (*Zostera marina* L.). Environ. Sci. Technol. 23:191–196.

FWIE. 2001. Endangered species information exchange web page for brown pelican. http://fwie.fw.vt.edu/WWW/esis/lists/e101047.htm. Accessed on October 29, 2001. Fish and Wildlife Information Exchange, Conservation Management Institute, Virginia Polytechnic Institute and State University, Blacksburg, VA.

FWS. 1964. Pesticide-wildlife studies, 1963: a review of Fish and Wildlife Service investigations during the calendar year. FWS Circular 199.

FWS. 1998. Draft environmental assessment and land protection plan, proposed South San Diego Bay Unit, San Diego National Wildlife Refuge. U.S. Fish and Wildlife Service, Portland, OR.

FWS. 2001a. Endangered species account for California brown pelican. http://www.sacramento.fws.gov/es/animal_spp_acct/ca_brown_pelican.htm. Accessed on October 19, 2001. U.S. Fish and Wildlife Service, Sacramento Field Office, Endangered Species Division, Sacramento, CA.

FWS. 2001b. AFWO listed species web page for California brown pelican. www.ccfwo.r1.fws.gov/EndangeredSpecies/brownpelican.html. Accessed on October 19, 2001. U.S. Fish and Wildlife Service, Arcata Fish and Wildlife Office, Listed Species Branch, Arcata, CA.

GLSFATF. 1993. Proposed protocol for a uniform sport fish consumption advisory. Draft. Great Lakes Sport Fish Advisory Task Force. (not seen, as cited in U.S. EPA 1993c)

Granholm, S. 2001a. California Wildlife Habitat Relationship System web page for brown pelican. www.dfg.ca.gov/whdab/B043.html. Accessed on October 5, 2001. California Department of Fish and Game, California Interagency Wildlife Task Group, Sacramento, CA.

Granholm, S. 2001b. California Wildlife Habitat Relationship System web page for surf scoter. www.dfg.ca.gov/whdab/B099.html. Accessed on October 5, 2001. California Department of Fish and Game, California Interagency Wildlife Task Group, Sacramento, CA.

Gross, W.G., and V.G. Heller. 1946. Chromates in animal nutrition. J. Ind. Hyg. Toxicol. 28(2):52–56.

Hartman, O. 1969. Atlas of the sedentariate polychaetous annelids from California. Allan Hancock Foundation, University of Southern California Los Angeles, California.

Hartman, G., and J. Goldston. 1994. Analysis of enclosed clamshell bucket for remedial dredging. 28th International Navigation Congress, Seville, Spain. PIANC, Brussels, Belgium.

Haseltine, S.D., L. Sileo, D.J. Hoffman, and B.D. Mulhern. 1985. Effects of chromium on reproduction and growth in black ducks. Unpublished data. (not seen, as cited in Sample et al. 1996)

Heinz, G. 1974. Effects of low dietary levels of methylmercury on mallard reproduction. Bull. Environ. Contam. Toxicol. 11(4):386–392.

Heinz, G. 1976a. Methylmercury: Second-year feeding effects on mallard reproduction and duckling behavior. J. Wildl. Manage. 40:82–90.

Heinz, G. 1976b. Methylmercury: Second-generation reproductive and behavioral effects on mallard ducks. J. Wildl. Manage. 40(4):710–715.

Heinz, G.H. 1979. Methylmercury: Reproductive and behavioral effects on three generations of mallard ducks. J. Wildl. Manage. 43(2):394–401.

Heinz, G.H., D.J. Hoffman, A.J. Krynitsky, and D.M.G. Weller. 1987. Reproduction in mallards fed selenium. Environ. Toxicol. Chem. 6:423–433.

Heinz, G.H., D.J. Hoffman, and L.G. Gold. 1989. Impaired reproduction of mallards fed an organic form of selenium. J. Wildl. Manage. 53(2):418–426.

Help, C., R.M. Warwick, M.R. Carr, P.M.J. Herman, R. Huys, N. Smol, and K. Van Holsbeke. 1988. Analysis of community attributes of the benthic meiofauna of Frierfjord/Langesundfjord. Mar. Ecol. Progr. Ser. 46:171–180.

Hoffman, B. 2001. Personal communication (telephone conversation with L. Maier, Exponent, Bellevue, WA, on December 14, 2001, regarding dredge windows in San Diego Bay). National Marine Fisheries Service, Southern California Office, Long Beach, CA.

Hough, J.L., M.B. Baird, G.T. Sfeir, C.S. Pacini, D. Darrow, and C. Wheelock. 1993. Benzo[a]pyrene enhances atherosclerosis in white carneau and show racer pigeons. Arterioscler Thromb. 13:1721–1727.

Hruby, T. 1987. Using similarity measures in benthic impact assessments. Environ. Monitor. Assess. 8:163–180.

Huckabee, J.W., J.W. Elwood, and S.G. Hildebrand. 1979. Accumulation of mercury in freshwater biota. pp. 277–302. In: The Biogeochemistry of Mercury in the Environment. J.O. Nriagu (ed). Elsevier, New York.

Hunt, J.W., B.S. Anderson, B.M. Phillips, J. Newman, R.S. Tjeerdema, R. Fairey, H.M. Puckett, M. Stephenson, R.W. Smith, C.J. Wilson, and K.M. Taberski. 2001. Evaluation and use of sediment toxicity reference sites for statistical comparisons in regional assessments. Environ. Toxicol. Chem. 20(6):1266–1275.

IEHR. 1991. Reassessment of liver findings in five PCB studies in rats. July 1, 1991. Institute for Evaluating Health Risks, Washington, DC.

Kabata-Pendias, A., and H. Pendias. 1992. Trace elements in soil and plants. Second Edition. CRC Press, Boca Raton, FL.

Kaufman, K., and R.T. Peterson. 2001. Peterson Online Bird Identification web page for brown pelican. www.petersononline.com/birds/month/brpe. Accessed October 19, 2001.

Keane, K. 2000. California least tern breeding survey, 1998 season. California Department of Fish and Game, Habitat Conservation and Planning Branch Report, 2000–2001. Sacramento, CA.

Kenaga, E.E. 1973. Factors to be considered in the evaluation of the toxicity of pesticides to birds in their environment. pp. 166–181. In: Environmental Quality and Safety, Global Aspects of Chemistry, Toxicology and Technology as Applied to the Environment. Volume II. Academic Press, Inc., New York.

Kenney, M. 2001. Personal communication (telephone conversation with L. Maier, Exponent, Bellevue, WA, on December 12, 2001, regarding dredge windows in San Diego Bay). U.S. Fish and Wildlife Service, Carlsbad, CA.

Keyes, M.C. 1968. The nutrition of pinnipeds. pp. 359–395. In: The Behavior and Physiology of Pinnipeds. R.J. Harrison (ed). Appleton–Century–Crofts, New York, NY.

Keyser Marston. In prep. NASSCO's economic benefits analysis. Draft Report. Prepared for NASSCO, San Diego, California. Keyser Marston Associates, Inc., San Diego, CA.

Khan, R.A., D.E. Barker, R. Hooper, E.M. Lee, K. Ryan, and K. Nag. 1994. Histopathology in winter flounder (*Pleuronectes americanus*) living adjacent to a pulp and paper mill. Arch. Environ. Contam. Toxicol. 26(1):95–102.

Krahn, M.M., M.S. Myers, D.G. Burrows, and D.C. Malins. 1984. Determination of metabolites of xenobiotics in the bile of fish from polluted waterways. Xenobiotica 14(8):633–646.

Kucera, T. 2001. California Wildlife Habitat Relationship System web page for western grebe. www.dfg.ca.gov/whdab/B010.html. Accessed on November 20, 2001. California Department of Fish and Game, California Interagency Wildlife Task Group, Sacramento, CA.

Landrum, P.F., and J.A. Robbins. 1990. Bioavailability of sediment-associated contaminants to benthic invertebrates. pp. 237–263. In: Sediments: Chemistry and Toxicity of In-Place Pollutants. R. Baudo, J. Giesy, and H. Muntau (eds). Lewis Publishers, Ann Arbor, MI.

Laubitz, D.R. 1970. Studies on the Caprellidae (Crustacea, Amphipoda) of the American North Pacific. National Museum of Natural Sciences, Publication in Biological Oceanography, No. 1. National Museum of Canada.

Lawrence, G.E. 1950. The diving and feeding activity of the western grebe on the breeding grounds. Condor 52(1):3–16.

Lee, R.F., and D.S. Page. 1997. Petroleum hydrocarbons and their effects in subtidal regions after major oil spills. Mar. Poll. Bull. 34(11):928–940.

Legendre, P., and L. Legendre. 1998. Numerical ecology. Second edition. Elsevier, Amsterdam.

Lillie, R.J., H.C. Cecil, J. Bitman, and G.F. Fries. 1974. Differences in response of caged white leghorn layers to various polychlorinated biphenyls (PCBs) in the diet. Poult. Sci. 53:726–732.

Long, E.R., and C.J. Wilson. 1997. On the identification of toxic hot spots using measures of the sediment quality triad. Mar. Pollut. Bull. 34(6):373–374.

Long, E.R., D.D. MacDonald, S.L. Smith, and F.D. Calder. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. Environ. Manage. 19:81–97.

Long, E.R., C.B. Hong, and C.G. Severn. 2001. Relationships between acute sediment toxicity in laboratory tests and abundance and diversity of benthic infauna in marine sediments: a review. Environ. Toxicol. Chem. 20(1):46–60.

Lowe, S., and B. Thompson. 1999. Identifying benthic indicators for San Francisco Bay. In: Chapter 4, Sediment Monitoring. 1997 Annual Report: San Francisco Estuary Regional Monitoring Program for Trace Substances. San Francisco Estuary Institute, Oakland, CA.

Lyngby, J.E., and H. Brix. 1984. The uptake of heavy metals in eelgrass *Zostera marina* and their effect on growth. Ecol. Bull. 36:81–89.

Lyngby, J.E., and H. Brix. 1987. Monitoring of heavy metal contamination in the Limfjord, Denmark, using biological indicators and sediment. Sci. Total Environ. 64:239–252.

Lyngby, J.E., and H. Brix. 1989. Heavy metals in eelgrass (*Zostera marina* L.) during growth and decomposition. Hydrobiologia 176/177:189–196.

Mackenzie, K.M., and D.M. Angevine. 1981. Infertility in mice exposed *in utero* to benzo[a]pyrene. Biol. Reprod. 24:183–191.

Mackenzie, R.D., R.U. Byerrum, C.F. Decker, C.A. Hoppert, and R.F. Langham. 1958. Chronic toxicity studies. II: hexavalent and trivalent chromium administered in drinking water to rats. Am. Med. Assoc. Arch. Ind. Health 18:232–234.

Malins, D.C., B.B. McCain, D.W. Brown, M.S. Myers, M.M. Krahn, and S.L. Chan. 1987. Toxic chemicals, including aromatic and chlorinated hydrocarbons and their derivatives, and liver lesions in white croaker (*Genyonemus lineatus*) for the vicinity of Los Angeles. Environ. Sci. Technol. 21:765–770.

Marty, G.D. 2003. Necropsy and histopathology of spotted sea bass sampled from San Diego Harbor in September 2002. Final Report. Fish Pathology Services, Davis, CA.

McCain, B.S., S-L. Chan, M.M. Krahn, D.W. Brown, M.S. Myers, J.T. Landahl, S. Pierce, R.C. Clark, Jr., and U. Varanasi. 1992. Chemical contamination and associate fish diseases in San Diego Bay. Environ. Sci. Technol. 269(4):725–733.

McCarthy, K.J., A.D. Uhler, and S.A. Stout. 1998. Weathering affects petroleum ID. Soil and Groundwater Cleanup August/September 1998.

McDonald, T.G., S. Aldrett, J.S. Bonner, R.L. Autenrieth, and P.J. Morris. 1997. Differential biodegradation of polynuclear aromatic hydrocarbons. pp. 213–218. In: *In Situ* and On-Site Bioremediation: Volume 2. Battelle Press, Columbus, OH.

McLane, M.A.R., and D.L. Hughes. 1980. Reproductive success of screech owls fed Aroclor® 1248. Environ. Contam. Toxicol. 9:661–665.

Mehring, Jr., A.L., J.H. Brumbaugh, A.J. Sutherland, and H. Titus. 1960. The tolerance of growing chickens for dietary copper. Poult. Sci. 39:713.

Minchin, P.R. 1987. An evaluation of the relative robustness of techniques for ecological ordination. Vegetation 69:89–107.

Morrier, A., L. Lesage, A. Reed, and J-P.L. Savard. 1997. Étude sur l'écologie de la Macreuse á front blanc au lac Malbaie, Réserve des Laurentides—1994–1995. Tech. Report Series No. 301. Can. Wildl. Serv., QC. (not seen, as cited in Savard et al. 1998)

Murchelano, R.A., and R.E. Wolke. 1985. Epizootic carcinoma in the winter flounder, *Pseudopleuronectes americanus*. Science 228:587–589.

Myers, M.S., L.D. Rhodes, and B.B. McCain. 1987. Pathologic anatomy and patterns of occurrence of hepatic neoplasms, putative preneoplastic lesions, and other idiopathic hepatic conditions in English sole (*Parophrys vetulus*) from Puget Sound, Washington. J. Natl. Cancer Inst. 78:333–363.

Myers, M.S., C. M. Stehr, O.P Olson, L.L. Johnson, B.B. McCain, S.L. Chan, and U. Varanasi. 1994. Relationships between toxicopahtic lesions and exposure to chemical contaminants in English sole (*Parophrys vetulus*), starry flounder (*Platichthys stellatus*), and white croaker (*Genyonemus lineatus*) from selected marine environments on the Pacific Coast, USA. Environ. Health Perspect. 102:200–215.

Nagy, K.A. 1987. Field metabolic rate and food requirement scaling in mammals and birds. Ecol. Monogr. 57(2):111–128.

Nagy, K.A., I.A. Girard, and T.K. Brown. 1999. Energetics of free-ranging mammals, reptiles, and birds. Ann. Rev. Nutr. 19:247–277.

Navy and SDUPD. 2000. San Diego Bay Integrated Natural Resources Management Plan. Prepared by Tierra Data Systems, Escondido, CA. U.S. Department of the Navy, Southwest Division and San Diego Unified Port District, San Diego, CA.

NGS. 1987. National Geographic field guide to the birds of North America. Second Edition. National Geographic Society.

NMFS. 2001. Office of Protected Resources web page for green sea turtle (*Chelonia mydas*). www.nmfs.noaa.gov/prot_res/species/turtles/green.html. Accessed on November 19, 2001. Last updated April 13, 2001. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD.

NMFS and FWS. 1998. Recovery plan for U.S. Pacific populations of the East Pacific green turtle (*Chelonia mydas*). National Marine Fisheries Service, Silver Spring, MD, and U.S. Fish and Wildlife Service.

NOAA. 2001. Fact sheet: Small diesel spills (500–5000 gallons). National Oceanic and Atmospheric Administration, Hazardous Materials Response and Assessment Division, Seattle, WA. 2 pp.

Norris, R.H., and A. George. 1993. Analysis and interpretation of benthic macroinvertebrate surveys. Chapter 7. In: Freshwater Biomonitoring and Benthic Macroinvertebrates. D.M. Rosenberg and V.H. Resh (eds). Chapman & Hall, New York, NY.

Nowak, B.F., and M.J. Kingsford. 2003. Exposure to Thiodan® results in lipofuscin accumulation in hepatocytes of the freshwater catfish *Tandanus tandanus*. Dis. Aquat. Org. 56:135–143.

O'Connor, T.P., K.D. Daskalakis, J.L. Hyland, J.F. Paul, and J.K. Summer 1998. Comparisons of sediment toxicity with predictions based on chemical guidelines. Environ. Toxicol. Chem. 17:468-471.

OCSD. 2003. Annual Report 2002: Ocean monitoring. Orange County Sanitation District, Fountain Valley, CA.

OEHHA. 1999. Prevalence of selected target chemical contaminants in sport fish from two California lakes: Public health designed screening study. California Environmental Protection Agency, Office of Environmental Health Hazard Assessment, Sacramento, CA.

OEHHA. 2001. Chemicals in fish: Consumption of fish and shellfish in California and the United States. Final Report. California Environmental Protection Agency, Office of Environmental Health Hazard Assessment, Sacramento, CA.

Olsgard, F. 1999. Effects of copper contamination on recolonisation of subtidal marine soft sediments—An experimental field study. Mar. Pollut. Bull. 38:448–462.

Page, D.S., E.S. Gilfillan, P.D. Boehm, W.A. Stubblefield, K.R. Parker, and A.W. Maki. 2001. Oil weathering and sediment toxicity in shorelines affected by the Exxon Valdez oil spill in Prince William Sound, Alaska. pp. 551–557. In: Proc. of the 2001 Oil Spill Conference. First Edition. American Petroleum Institute, Washington, DC.

Palermo, M.R., and D. Hayes. 1992. Environmental effects of dredging. In: Handbook of Coastal and Ocean Engineering. Volume 3. J.B. Herbich (ed). Gulf Publishing Company, Houston, TX.

Palmer, R.S. (ed). 1962. Handbook of North American birds. Vol. 1. Yale University Press, New Haven, CT.

Pattee, O.H. 1984. Eggshell thickness and reproduction in American kestrels exposed to chronic dietary lead. Arch. Environ. Contam. Toxicol. 13:29–34.

Pattee, O.H., S.N. Wiemeyer, and D.M. Swineford. 1988. Effects of dietary fluoride on reproduction in eastern screech-owls. Arch. Environ. Contam. Toxicol. 17:213–218.

Pearson, T.H., and R. Rosenberg. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanogr. Mar. Biol. Annu. Rev. 16:229–311.

Peng, J., E.Y. Zeng, T-L. Ku, and S. Luo. 2003. Significance of sediment resuspension and tidal exchange to reduction of polychlorinated biphenyl mass in San Diego Bay, California. pp. 101-109. In: Southern California Coastal Water Research Project Biennial Report 2001-2002. First edition. Southern California Coastal Water Research Project Authority

Peterson, R.S., and G.A. Bartholomew. 1967. The natural history and behavior of the California sea lion. Am. Soc. Mammal. Special Publ. No. 1, Lawrence, KS. 79 pp.

Prange, J.A., and W.C. Dennison. 2000. Physiological responses of five seagrass species to trace metals. Mar. Pollut. Bull. 41:327–336.

PTI. 1988. Sediment quality values refinement: Volume 1. 1988 Update and evaluation of Puget Sound AET. Prepared for the U.S. Environmental Protection Agency, Seattle, WA. PTI Environmental Services, Bellevue, WA.

PTI. 1992. Draft Campbell Shipyards remedial action alternatives analysis report. Prepared for MARCO Seattle, Seattle, WA. PTI Environmental Services, Bellevue, WA. February 1992.

Ratti, J.T. 1981. Identification and distribution of Clark's grebe. West. Birds 12:41–46.

Rhoads, D.C. 1974. Organism-sediment relations on the muddy seafloor. Oceanogr. Mar. Biol. Ann. Rev. 12:263–300.

Rhoads, D.C., and L.F. Boyer. 1982. Chapter 1. The effects of marine benthos on physical properties of sediments, a successional perspective. In: Animal-sediment relations. P.L. McCall and M.J.S. Tevesz (eds). Plenum Publishing, New York.

Rhoads, D.C., and J.D. Germano. 1982. Characterization of benthic processes using sediment profile imaging: An efficient method of remote ecological monitoring of the seafloor (REMOTS® System). Mar. Ecol. Prog. Ser. 8:115–128.

Rhoads, D.C., P.L. McCall, and J.Y. Yingst. 1978. Disturbance and production on the estuarine seafloor. American Scientist. Sept-Oct. 577-586.

Riedman, M.L. 2001. California Wildlife Habitat Relationships System web page for California sea lion. www.dfg.ca.gov/whdab/M170.html. Accessed on October 19, 2001. California Department of Fish and Game, California Interagency Wildlife Task Group, Sacramento, CA.

Robertus, J.H. 2001a. Personal communication (letter to M. Chee, NASSCO, and S. Halvax, Southwest Marine, dated June 1, 2001, regarding assessment and remediation of contaminated sediments in San Diego Bay at NASSCO and Southwest Marine Shipyards). California Regional Water Quality Control Board, San Diego, CA.

Robertus, J.H. 2001b. Personal communication (letter to M. Chee, NASSCO, and S. Halvax, Southwest Marine, dated July 16, 2002, regarding 12/24/01 RWQCB 13267 letter in response to Tech Memo 2). California Regional Water Quality Control Board, San Diego, CA.

Robertus, J.H. 2002a. Personal communication (letter to S. Halvax, Southwest Marine, and M. Chee, NASSCO, dated March 6, 2002, regarding background reference conditions for assessment and remediation of contaminated sediments at NASSCO and Southwest Marine shipyards). San Diego Regional Water Quality Control Board, San Diego, CA.

Robertus, J.H. 2002b. Personal communication (letter to M. Chee, NASSCO, and S. Halvax, Southwest Marine, dated July 16, 2002, regarding assessment of bioaccumulation and risk to fish health from sediment contaminants at NASSCO and Southwest Marine shipyards). San Diego Regional Water Quality Control Board, San Diego, CA.

Robertus, J.H. 2002c. Personal communication (letter to S. Halvax, Southwest Marine, and M. Chee, NASSCO, dated October 18, 2002, regarding analysis of polynuclear aromatic hydrocarbons [PAHs] in pore water at NASSCO and Southwest Marine Shipyards). San Diego Regional Water Quality Control Board, San Diego, CA.

Rosenberg, R. 2001. Marine benthic faunal successional stages and related sedimentary activity. Scientia Marina 65:107–119.

Rosenfeld, I., and O.A. Beath. 1954. Effect of selenium on reproduction in rats. Proc. Soc. Exper. Biol. Med. 87(2):295–297.

RWQCB. 1994. 1994 Water quality control plan for San Diego Basin. California Regional Water Quality Control Board, Region 9, San Diego, CA.

RWQCB. 1997. Proposed regional toxic hot spot cleanup plan. Regional Water Quality Control Board, San Diego Region.

RWQCB. 2000. Final problem statement for the total maximum daily load for toxic substances in Newport Bay and San Diego Creek. California Regional Water Quality Control Board, Santa Ana Region.

RWQCB. 2001. Guidelines for assessment and remediation of contaminated sediments in San Diego Bay at NASSCO and Southwest Marine shipyards. California Regional Water Quality Control Board, San Diego Region, San Diego, CA.

Sample, B.E., D.M. Opresko, and G.W. Suter. 1996. Toxicological benchmarks for wildlife: 1996 revision. ES/ER/TM-86/RS. Prepared for the U.S. Department of Energy, Office of Environmental Management. Oak Ridge National Laboratory, Risk Assessment Program, Health Sciences Research Division, Oak Ridge, TN.

Sauer, T.C., and A.D. Uhler. 1994. Pollutant source identification and allocation: Advances in hydrocarbon fingerprinting. Remediation (Winter):25–50. John Wiley & Sons, Inc.

Savard, J.-P., D. Bordage, and A. Reed. 1998. Surf scoter (*Melanitta perspicillata*). In: The Birds of North America, No. 363. A. Poole and F. Gill (eds). The Birds of North America, Inc., Philadelphia, PA.

SCAMIT. 2001. A taxonomic listing of soft bottom macro- and megainvertebrates from infaunal and epibenthic monitoring programs in the Southern California Bight. Edition 4. The Southern California Association of Marine Invertebrate Taxonomists, San Pedro, CA.

SCCWRP and MBC. 1994. Santa Monica Bay Seafood Consumption Study: Final Report. Southern California Coastal Water Research Project and MBC Applied Environmental Sciences. Westminster and Costa Mesa, CA.

Schiff, K.C., S.M. Bay, and D.W. Diehl. 2003. Stormwater toxicity in Chollas Creek and San Diego Bay, California. In: Southern California Coastal Water Research Project Biennial Report 2001-2002. Southern California Coastal Water Research Project Authority. May 2003.

Schlatterer, B.T., M.M. Coenen, E. Ebert, R. Grau, V. Hilbig, and R. Munk. 1993. Effects of bis(tri-*n*-butyltin)oxide in Japanese quail exposed during egg laying period: An interlaboratory comparison study. Arch. Environ. Contam. Toxicol. 24:440–448.

Schlicker, S.A., and D.H. Cox. 1968. Maternal dietary zinc, and development and zinc, iron, and copper content of the rat fetus. J. Nutr. 95:287–294.

Schroeder, H.A., and M. Mitchener. 1971. Toxic effects of trace elements on the reproduction of mice and rats. Arch. Environ. Health. 23:102–106.

SDFD. 2003. 1913—The great Standard Oil fire. http://www.sandiego.gov/fireandems/about/stand.shtml. Accessed on September 23, 2003. San Diego Fire-Rescue Department.

Shine, J.P. 2003. Use of receiver operating characteristic curves to evaluate sediment quality guidelines for metals. Environ. Toxicol. Chem. 22(7):1642–1648.

Smith, R.W., M. Bergen, S.B. Weisberg, D. Cadien, A. Dalkey, D. Montagne, J.K. Stull, and R.G. Velarde. 2001. Benthic response index for assessing infaunal communities on the southern California mainland shelf. Ecol. Appl. 11(4):1073–1087.

Smith, R.W., J.A. Ranasinghe, S.B. Weisberg, D.E. Montagne, D.B. Cadien, T.K. Mikel, R.G. Velarde, and A. Dalkey. 2003. Appendices C and D of benthic response index for assessing infaunal communities on the southern California mainland shelf. Southern California Coastal Water Research Project Authority.

Stahl, J.L., J.L. Gregor, and M.E. Cook. 1990. Breeding-hen and progeny performance when hens are fed excessive dietary zinc. Poult. Sci. 69:259–263.

Stehr, C.M., M.S. Myers, D.G. Burrows, M.M. Krahn, J.P. Meador, B.B. McCain, and U. Varanasi. 1997. Chemical contamination and associated liver diseases in two species of fish from San Francisco Bay and Bodega Bay. Ecotoxicology 6:33–65.

Storer, R.W., and G.L. Neuchterlein. 1992. Western and Clark's grebe. In: The Birds of North America, No. 26. A. Poole, P. Stettenheim, and F. Gill (eds). The Academy of Natural Sciences, Philadelphia, and The American Ornithologists' Union, Washington, DC.

Stout, S.A., A.D. Uhler, K.J. McCarthy, and S. Emsbo-Mattingly. 2002. Chemical fingerprinting of hydrocarbons. Chapter 6. pp. 137–260. In: Introduction to Environmental Forensics. B.L. Murphy and R.D. Morrison (eds). Academic Press, San Diego, CA.

Stull, J.K. 1995. Two decades of marine biological monitoring, Palos Verdes, California, 1972 to 1992. Bull Southern California Acad. Sci. 94(1):21-45.

Sutou, S., K. Yamamoto, H. Sendota, and M. Sugiyama. 1980. Toxicity, fertility, teratogenicity, and dominant lethal tests in rats administered cadmium subchronically. II. Fertility, teratogenicity, and dominant lethal tests. Ecotoxicol. Environ. Saf. 4:51–56.

Swartz, R.C., F.A. Cole, J.O. Lamberson, S.P. Ferraro, D.W. Schults, W.A. DeBen, H. Lee, II, and R.J. Ozretich. 1994. Sediment toxicity, contamination, and amphipod abundance at a DDT- and dieldrin-contaminated site in San Francisco Bay. Environ. Toxicol. Chem. 13:949–962.

Swatko, T.D., and S.W. Berry. 1989. Dredging techniques used to develop a man-made lake on a deep organic profile. Proceedings of the 22nd Annual Dredging Seminar sponsored by the Texas A&M University Center for Dredging Studies, Tacoma, WA.

SWRCB. 1998. Chemistry, toxicity, and benthic community conditions in sediments of the San Diego Bay region. Final Addendum Report. California State Water Resources Control Board. September 30, 1998.

Thiyagarajah, A., W.R. Hartley, and A. Abdelghani. 1998. Hepatic hemosiderosis in buffalo fish (*Ictiobus* spp.). Mar. Environ. Res. 46(1-5):203–207.

Thompson, B.C., J.A. Jackson, J. Burger, L.A. Hill, E.M. Kirsch, and J.L. Atwood. 1997. Least tern (*Sterna antillarum*). In: The Birds of North America, No. 290. A. Poole and F. Gill (eds). The Academy of Natural Sciences, Philadelphia, PA, and The American Ornithologists' Union, Washington, DC.

Tomkins, I.R. 1959. Life history notes on the least tern. Wilson Bull. 71(4):313–322.

U.S. DOI. 1996. CERCLA NRDA regulations. 43 CFR 11, Subpart E, paragraph 11.72(b)(2). U.S. Department of the Interior.

U.S. EPA. 1977. Application of numerical classification in ecological investigations of water pollution. EPA/600/3-77/033. U.S. Environmental Protection Agency, Office of Research and Development, Corvallis, OR.

U.S. EPA. 1987. Recommended protocols for sampling and analyzing subtidal benthic macroinvertebrate assemblages in Puget Sound. U.S. Environmental Protection Agency Region 10, Office of Puget Sound, Seattle, WA

U.S. EPA. 1988a. Recommendations for and documentation of biological values for use in risk assessment. EPA/600/6-87/008. U.S. Environmental Protection Agency, Office of Research and Development, Cincinnati, OH.

U.S. EPA. 1988b. Guidance for conducting remedial investigations and feasibility studies under CERCLA. U.S. Environmental Protection Agency.

U.S. EPA. 1989a. Risk assessment guidance for Superfund. Volume I: Human health evaluation manual. EPA/540/1-89/002. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

U.S. EPA. 1989b. Assessing human health risks from chemically contaminated fish and shellfish. EPA/503/8-89/002. U.S. Environmental Protection Agency, Office of Marine and Estuarine Protection and Office of Water Regulations and Standards, Washington, DC.

U.S. EPA. 1992a. An SAB Report: Review of Sediment Criteria Development Methodology for Non-Ionic Organic Contaminants. EPA-SAB-EPEC-93-002. U.S. Environmental Protection Agency, Science Advisory Board, Washington, DC. November, 1992. 12 pp.

U.S. EPA. 1992b. Sediment classification methods compendium. EPA 823-R-92-006. U.S. Environmental Protection Agency, Washington, DC.

U.S. EPA. 1992c. A supplemental guidance to RAGS: Calculating the concentration term. Publication 9285.7-081. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

U.S. EPA. 1993a. Wildlife exposure factors handbook. Volume I of II. EPA/600/R-93/187a. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

U.S. EPA. 1993b. Draft sampling and guidance manual. U.S. Environmental Protection Agency, Washington, DC. (not seen, as cited in U.S. EPA 1993c)

U.S. EPA. 1993c. Fish sampling and analysis: A guidance document for issuing fish advisories. Draft. U.S. Environmental Protection Agency, Office of Science and Technology, Washington, DC.

U.S. EPA. 1993d. Selecting remediation techniques for contaminated sediment. EPA/823/B-93/001. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. EPA. 1994. Selecting and using reference information in Superfund ecological risk assessments. Eco Update Volume 2, Number 4. EPA/540/F-94/050. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response.

U.S. EPA. 1995a. Short-term methods for estimating the chronic toxicity of effluents and receiving water to West Coast marine and estuarine organisms. EPA/600/R-95/136. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

U.S. EPA. 1995b. Great Lakes Water Quality Initiative technical support document for wildlife criteria. EPA-820-B-95-009. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. EPA. 1995c. Engineering forum issue: Determination of background concentrations of inorganics in soils and sediments at hazardous waste sites. EPA/540/5-96/500. U.S. Environmental Protection Agency, Office of Research and Development.

U.S. EPA. 1996a. PCBs: Cancer dose-response assessment and application to environmental mixtures. EPA/600/P-96/001F. U.S. Environmental Protection Agency, Office of Research and Development, National Center for Environmental Assessment, Washington, DC.

U.S. EPA. 1996b. Supplemental guidance to RAGS: Region 4 Bulletins. U.S. Environmental Protection Agency, Region 4, Waste Management Division, Office of Technical Services, Atlanta, GA.

U.S. EPA. 1997a. Ecological risk assessment guidance for Superfund: process for designing and conducting ecological risk assessments. EPA/540/R-97/0C5. U.S. Environmental Protection Agency, Solid Waste and Emergency Response, Washington, DC.

U.S. EPA. 1997b. Arsenic and fish consumption. EPA-822-R-97-003. U.S. Environmental Protection Agency, Office of Science and Technology, Office of Water, Washington, DC.

U.S. EPA. 1998a. Guidelines for ecological risk assessment. EPA/630/R-95/002F. U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC.

U.S. EPA. 1998b. Idling Vehicle Emissions. U.S. Environmental Protection Agency, Air and Radiation, Office of Transportation and Air Quality. EPA420-F98-014.

U.S. EPA. 1998c. EPA's contaminated sediment management strategy. EPA/823/R-98/001. U.S. Environmental Protection Agency, Office of Water.

U.S. EPA. 1999a. Issuance of final guidance: Ecological risk assessment and risk management principles for Superfund sites: OSWER directive 9285.28P. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

U.S. EPA. 1999b. Update to supplemental guidance to RAGS: Region 4 bulletins. August 1999 revision. U.S. Environmental Protection Agency, Region 4, Waste Management Division, Office of Technical Services, Atlanta, GA.

U.S. EPA. 2000a. Water quality standards: Establishment of numeric criteria for priority toxic pollutants for the state of California. Rule. U.S. Environmental Protection Agency.

U.S. EPA. 2000b. Development of a framework for evaluating numerical sediment quality targets and sediment contamination in the St. Louis River area of concern. EPA/905/R-00/008. U.S. Environmental Protection Agency, Great Lakes National Program Office, Chicago, IL.

U.S. EPA. 2000c. Update to supplemental guidance to RAGS: Region 4 bulletins. March 2000 revision. U.S. Environmental Protection Agency, Region 4, Waste Management Division, Office of Technical Services, Atlanta, GA.

U.S. EPA. 2000d. Guidance for assessing chemical contaminant data for use in fish advisories. Volume 1: Fish sampling and analysis. Third edition. EPA/823/B-00/008. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. EPA. 2000e. Guidance for assessing chemical contaminant data for use in fish advisories. Volume 2: Risk assessment and fish consumption limits. Third edition. EPA/823/B-00/008. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. EPA. 2001a. Methods for collection, storage, and manipulation of sediments for chemical and toxicological analyses: technical manual. EPA-823-B-01-002. U.S. Environmental Protection Agency, Office of Water.

U.S. EPA. 2001b. Update to supplemental guidance to RAGS: Region 4 bulletins. April 2001 revision. U.S. Environmental Protection Agency, Region 4, Waste Management Division, Office of Technical Services, Atlanta, GA.

U.S. EPA. 2001c. Update to supplemental guidance to RAGS: Region 4 bulletins. November 2001 revision. U.S. Environmental Protection Agency, Region 4, Waste Management Division, Office of Technical Services, Atlanta, GA.

U.S. EPA. 2002a. Guidance for comparing background and chemical concentrations in soil for CERCLA. EPA/540/R-01/003. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

U.S. EPA. 2002b. Calculating exposure point concentrations at hazardous waste sites. Draft. OSWER 9285.6-10. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

U.S. EPA. 2003. Integrated Risk Information System. Available at: www.epa.gov/iris/. U.S. Environmental Protection Agency, Office of Research and Development.

U.S. EPA/Corps. 1991. Evaluation of dredged material proposed for ocean disposal. EPA/503/8-91/001. U.S. Environmental Protection Agency, Office of Water and U.S. Army Corps of Engineers.

U.S. EPA/Corps. 1998. Evaluation of dredged material proposed for discharge in waters of the U.S.—testing manual (Inland Testing Manual). Draft Report. U.S. Environmental Protection Agency, Washington, DC, and U.S. Army Corps of Engineers.

URS. 2002. Inventory of waterfront structures, Southwest Marine Shipyard. URS Corporation.

Vermeer, K., and N. Bourne. 1984. The white-winged scoter diet in British Columbia waters: resource partitioning with other scoters. pp. 30–38. In: Marine Birds: Their Feeding Ecology and Commercial Fisheries Relationships. D.N. Nettleship, G.A. Sanger, and P.F. Springer (eds). Minister of Supply and Services Canada.

Verschuuren, H.G., R. Kroes, E.M. den Tonkelaar, J.M. Berkvens, P.W. Helleman, A.G. Rauws, P.L. Schuller, and G.J. van Esch. 1976. Toxicity of methylmercury chloride in rats. II: reproduction study. Toxicology 6:97–106.

Vogelbein, W.K. 1993. Ultrastructural and histological characterization of hepatic neoplasms in mummichog *Fundulus heteroclitus* inhabiting a creosote-contaminated environment. Mar. Environ. Res. 35(1-2):197–198.

Vos, J.G., H.L. van der Maas, A. Musch, and E. Ram. 1971. Toxicity of hexachlorobenzene in Japanese quail with special reference to porphyria, liver damage, reproduction, and tissue residues. Toxicol. Appl. Pharmacol. 18:944–957.

Whitaker, J.O., Jr. 1997. National Audubon Society field guide to North American mammals. Alfred A. Knopf, New York, NY. 937 pp.

White, D.H., and M.T. Finley. 1978. Uptake and retention of dietary cadmium in mallard ducks. Environ. Res. 17:53–59.

Whiting, S.D., and J.D. Miller. 1998. Short term foraging ranges of adult green turtles (*Chelonia mydas*). J. Herpetol. 32(3):330–337.

Williams, T.P., J.M. Bubb, and J.N Lester. 1994. Metal accumulation within salt marsh environments: A review. Mar. Pollut. Bull. 28:277–290.

Wilson, C.M. 2002. Personal communication (letter to J. Robertus, San Diego, CA, dated February 22, 2002, regarding applicability of state board resolution 92-49 in setting sediment cleanup levels). State Water Resources Control Board, Sacramento, CA.

Wilson, N.D., N.M. Shear, D.J. Paustenbach, and P.S. Price. 1998. The effects of cooking practices on the concentration of DDT and PCB compounds in the edible tissue of fish. J. Exposure Anal. Environ. Epidemiol. 8(3):423–436.

Woodward-Clyde. 1995. Historical occupancy search, Southwest Marine, foot of Sampson Street, San Diego, California. Prepared for Southwest Marine, Inc., San Diego, CA. Woodward-Clyde Consultants, San Diego, CA. 15 pp.

Ydenberg, R.C., and L.S. Forbes. 1988. Diving and foraging in the western grebe. Ornis Scandinavica 19:129–133.

Zar, J.H. 1996. Biostatistical analysis. Third Edition. Prentice-Hall, Upper Saddle River, NJ.

Zenetos, A., and E. Papathanassiou. 1989. Community parameters and multivariate analysis as a means of assessing the effects of tannery effluents on macrobenthos. Mar. Poll. Bull. 20(4):176–181.

Zeng, E.Y., and C.L. Vista. 1997. Organic pollutants in the coastal environment off San Diego, California. 1. Source identification and assessment by compositional indices of polycyclic aromatic hydrocarbons. Environ. Toxicol. Chem. 16(2):179–188.

Figures

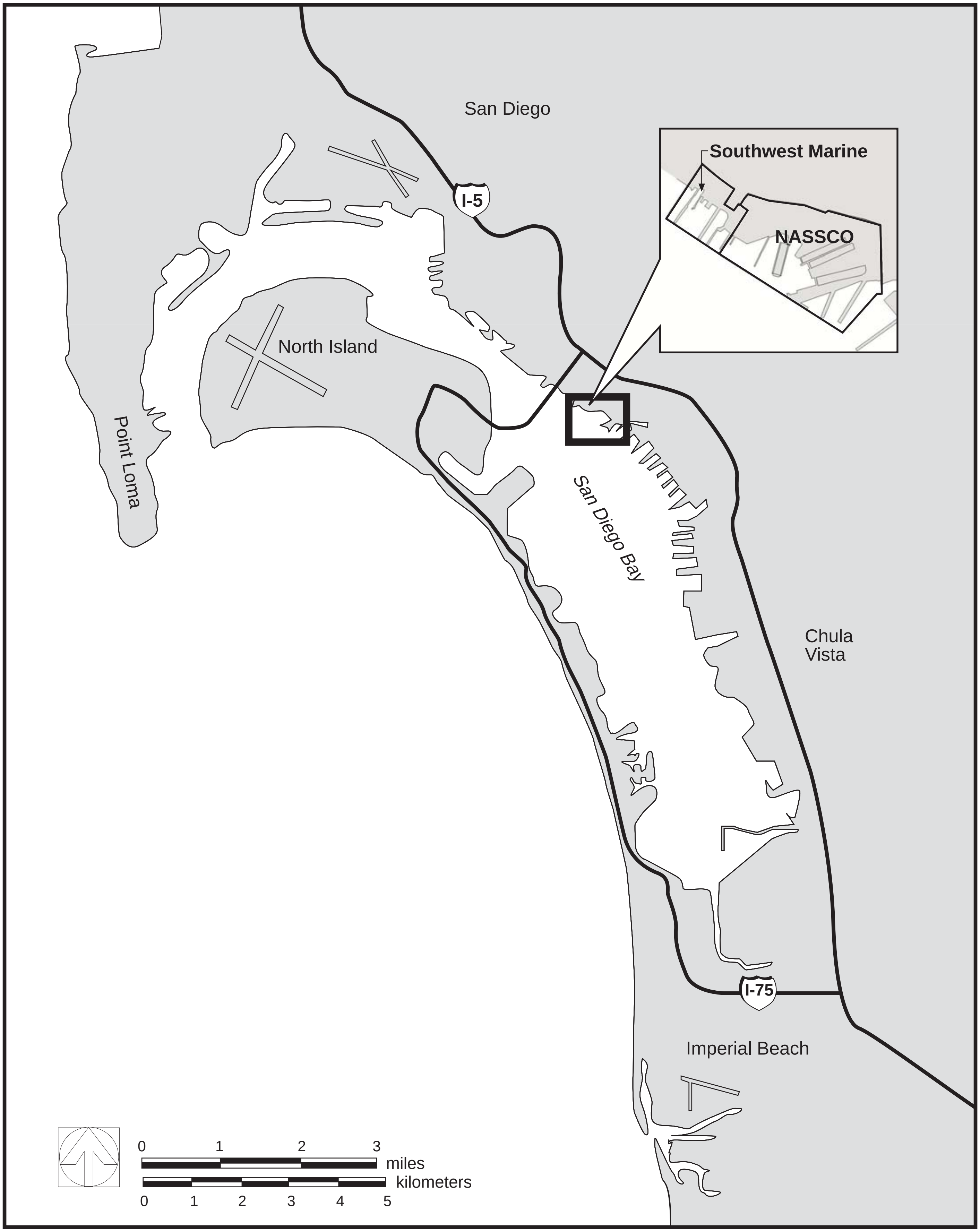

Figure 1-1. Site location

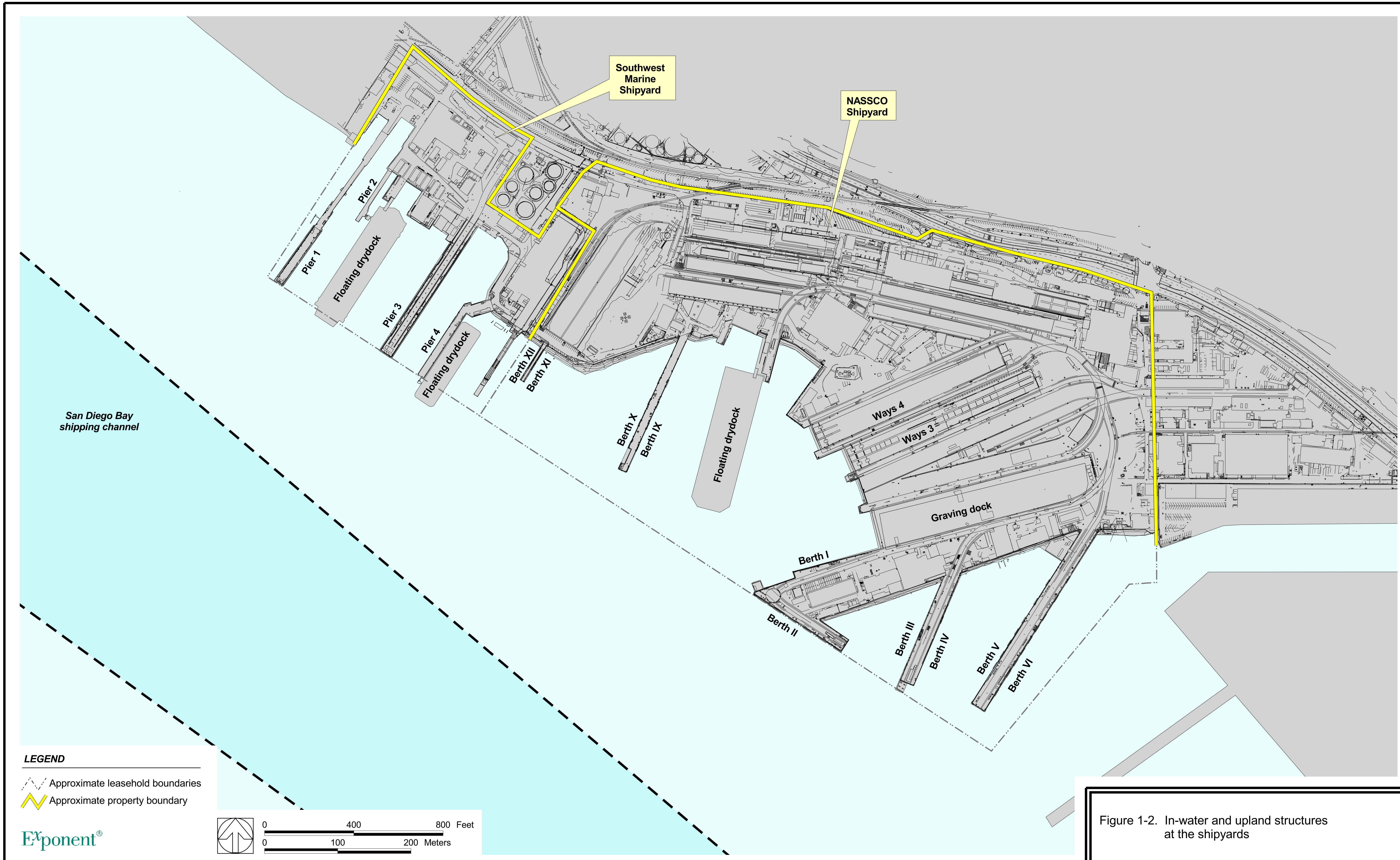

Figure 1-2. In-water and upland structures at the shipyards

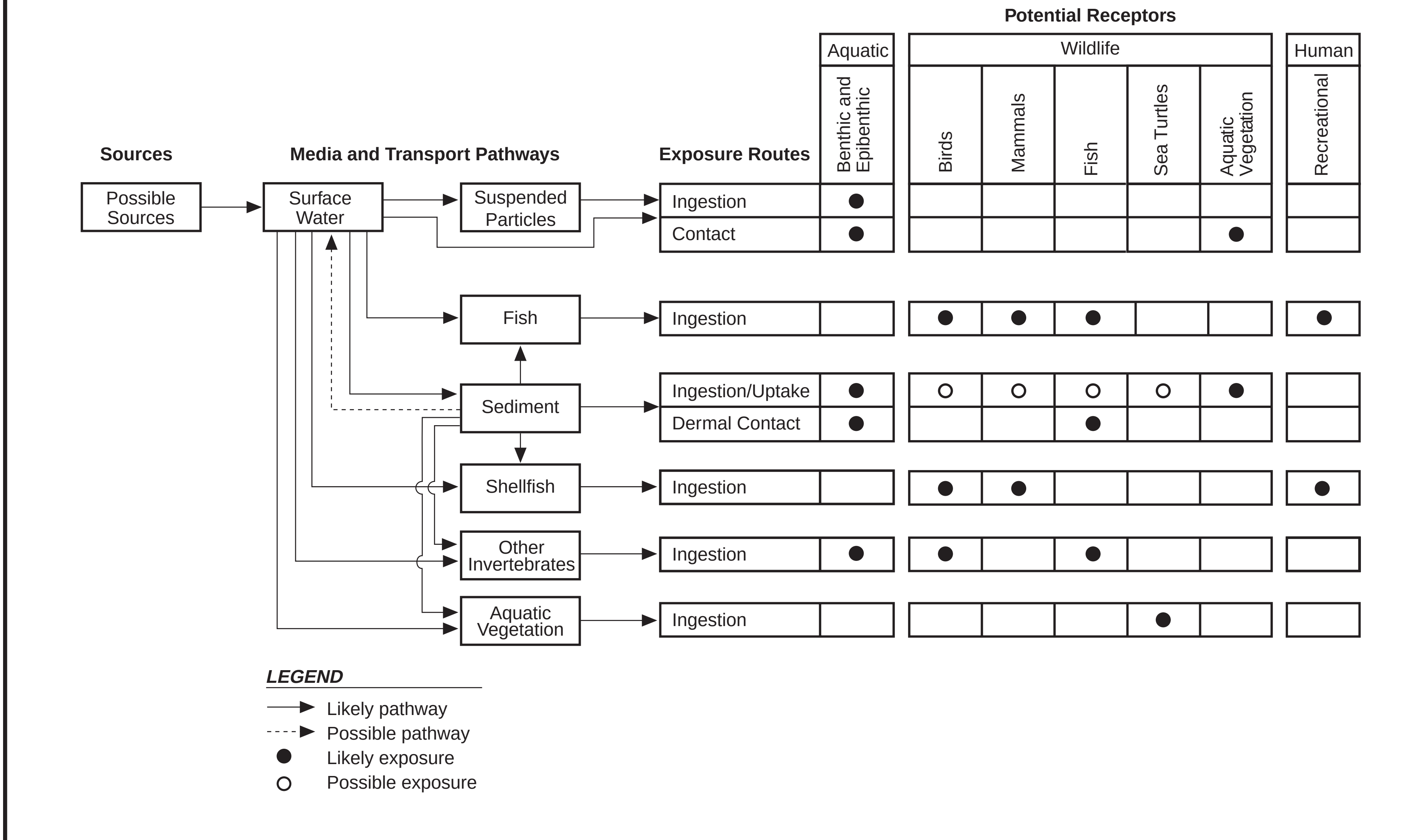

Figure 1-3. Conceptual site model

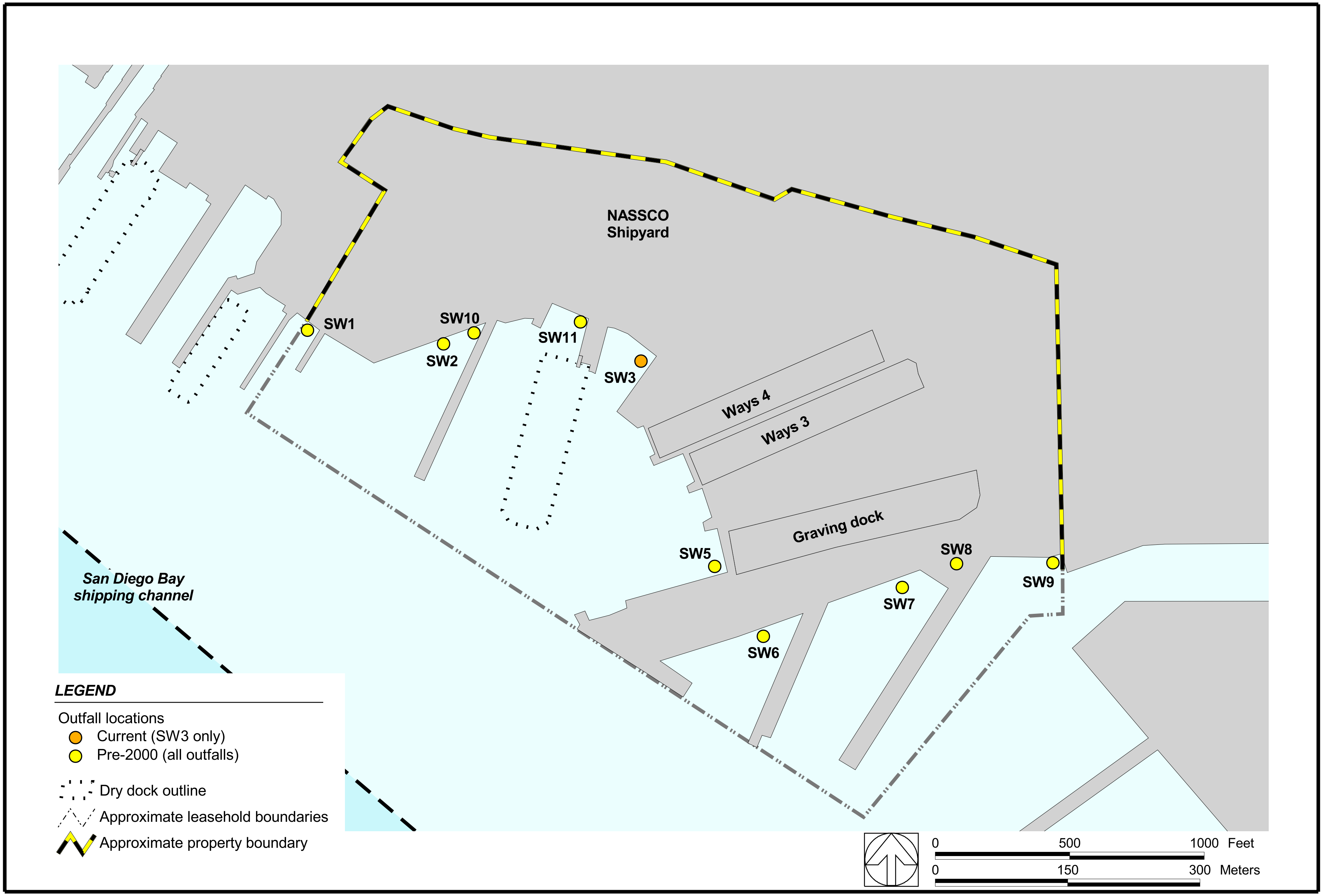

Figure 1-4. Current and previous stormwater discharge locations at NASSCO

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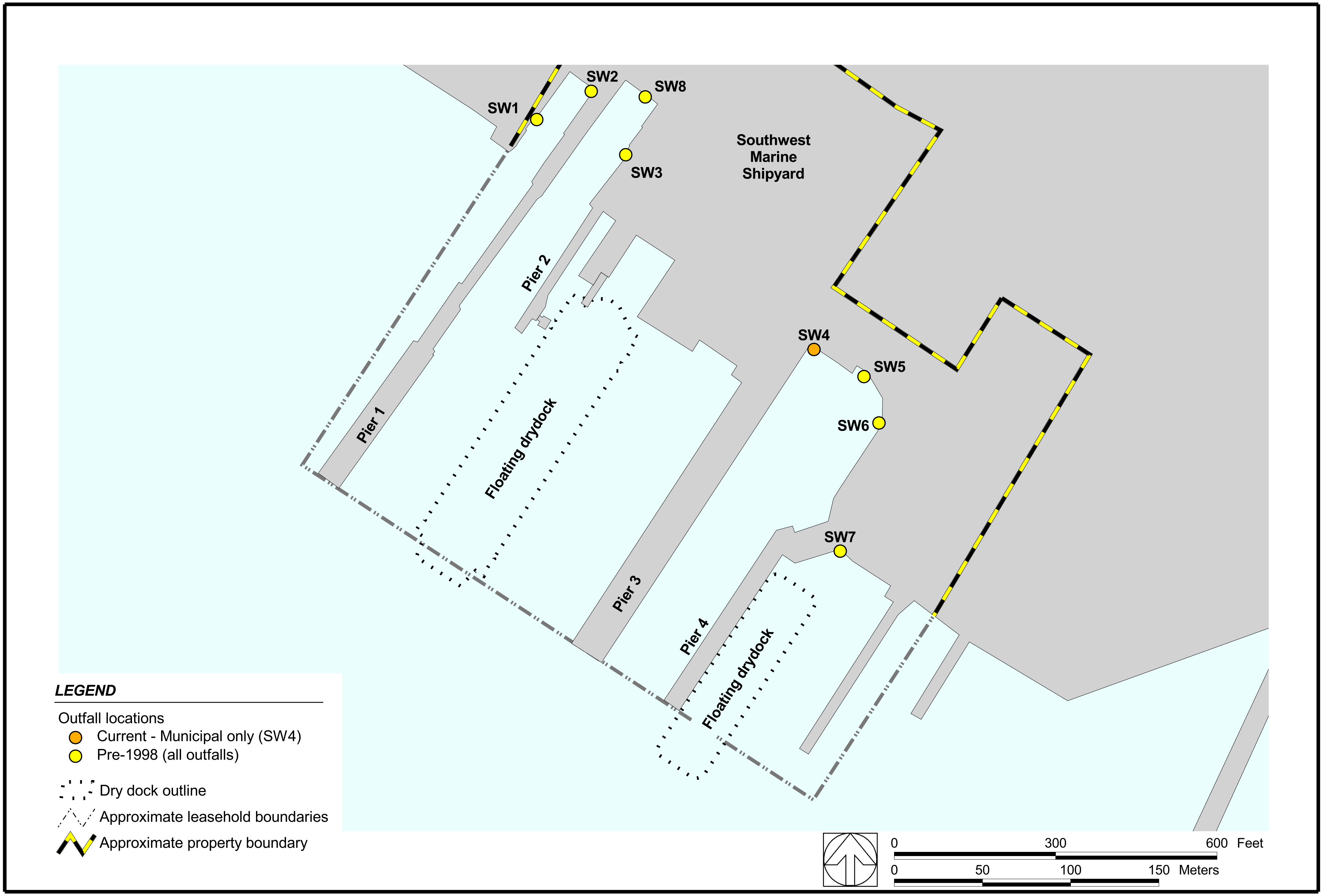

Figure 1-5. Current and previous stormwater discharge locations at Southwest Marine

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8601718.002 1201 | Sep 22, 2003 | fig 01-5 outfall SWM view | Fig 01-5 SWM outfall layout | g:/swm_nassco/projects/shipyards_rpt.apr

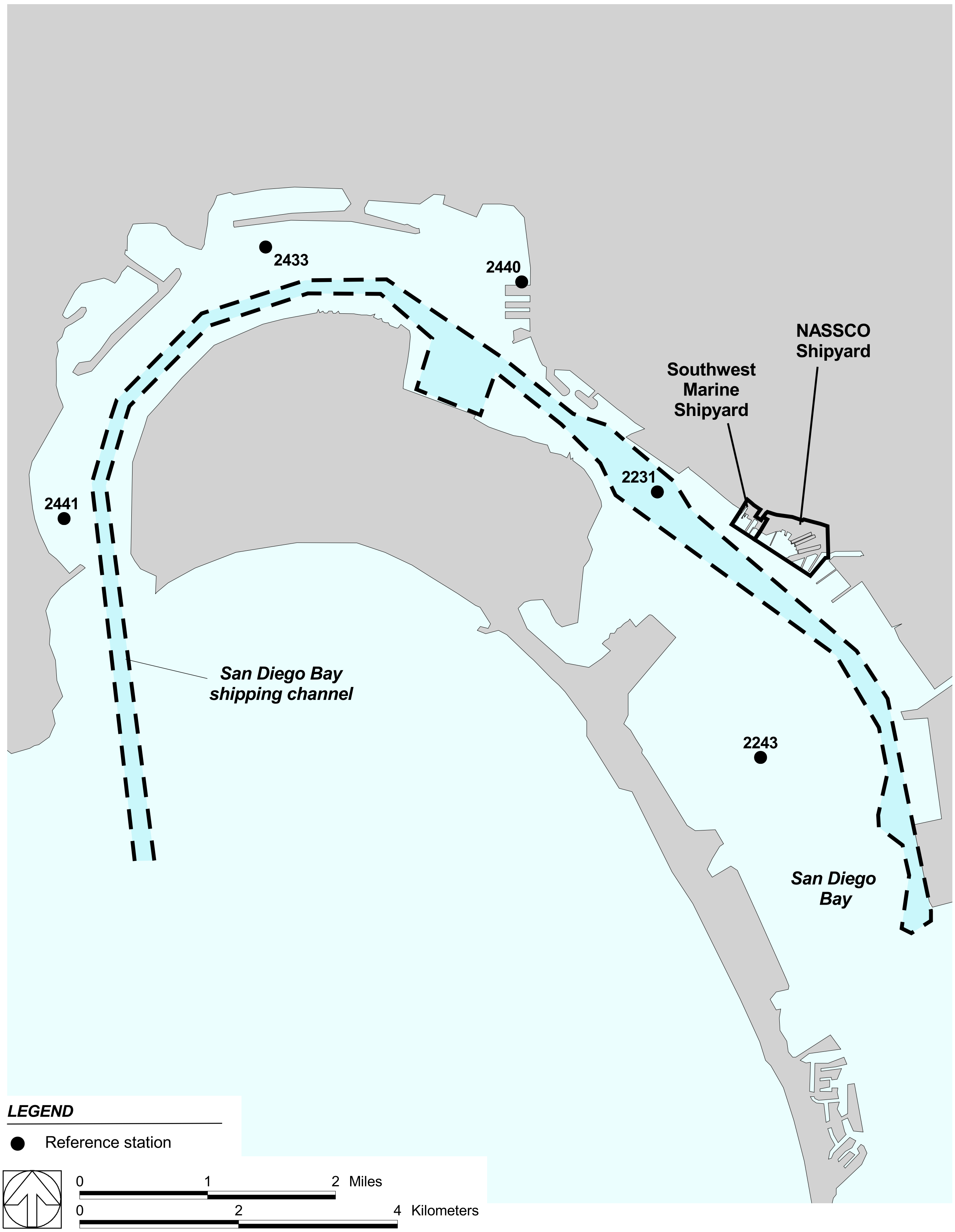


Figure 2-1. Reference station locations

Exponent®

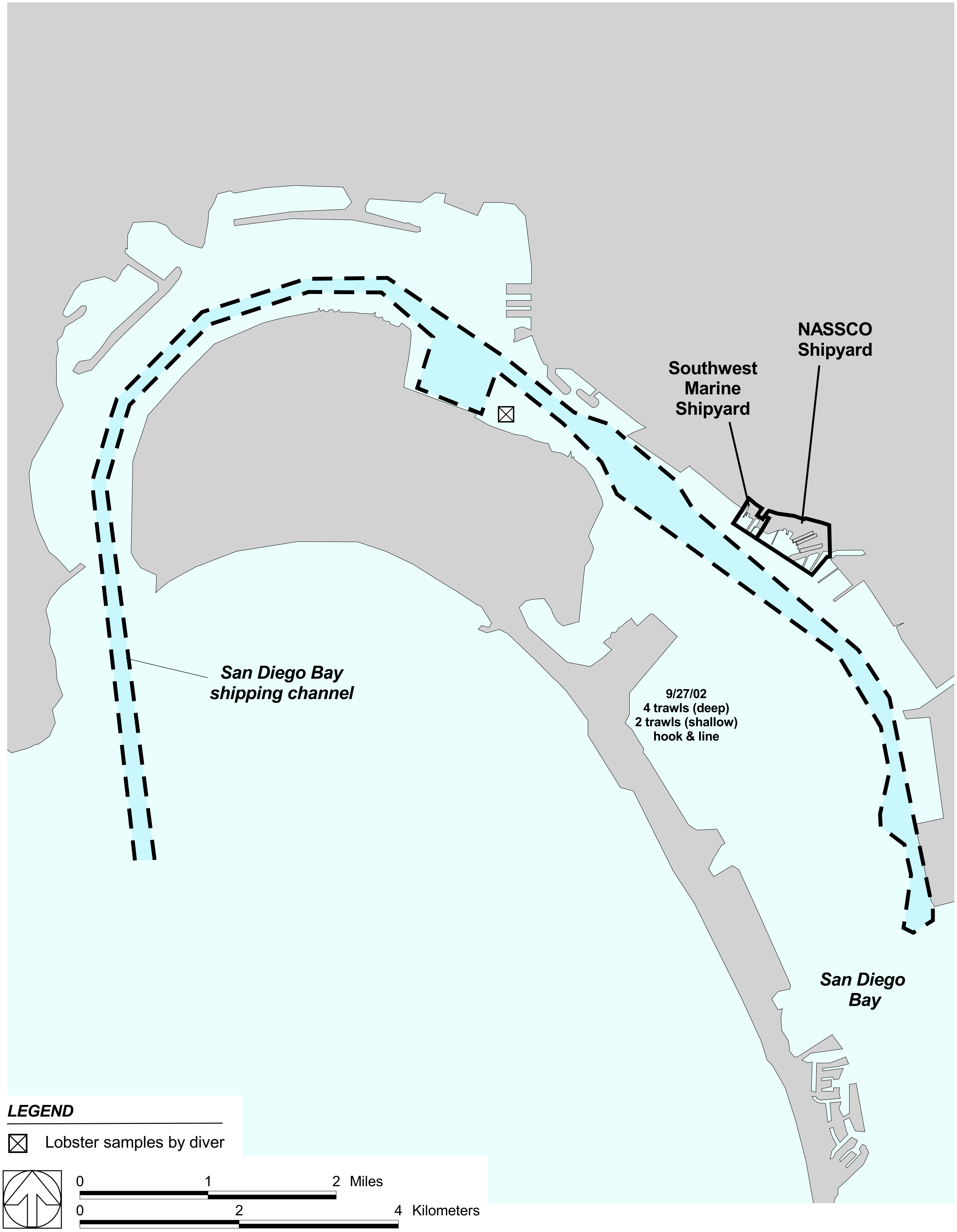


Figure 2-2. Biota sampling reference locations

Exponent®

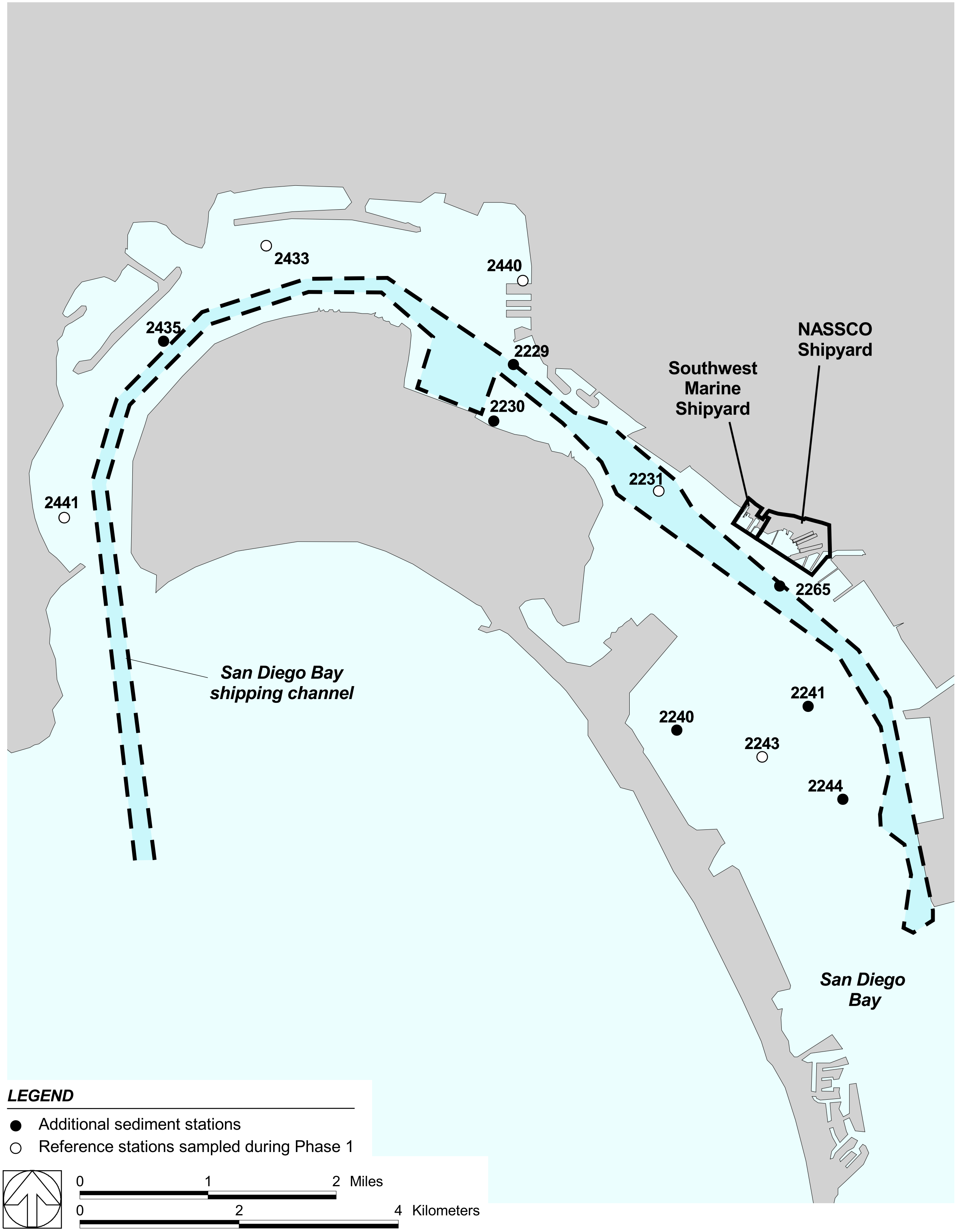


Figure 2-3. Additional sediment sampling locations in San Diego Bay

Exponent®

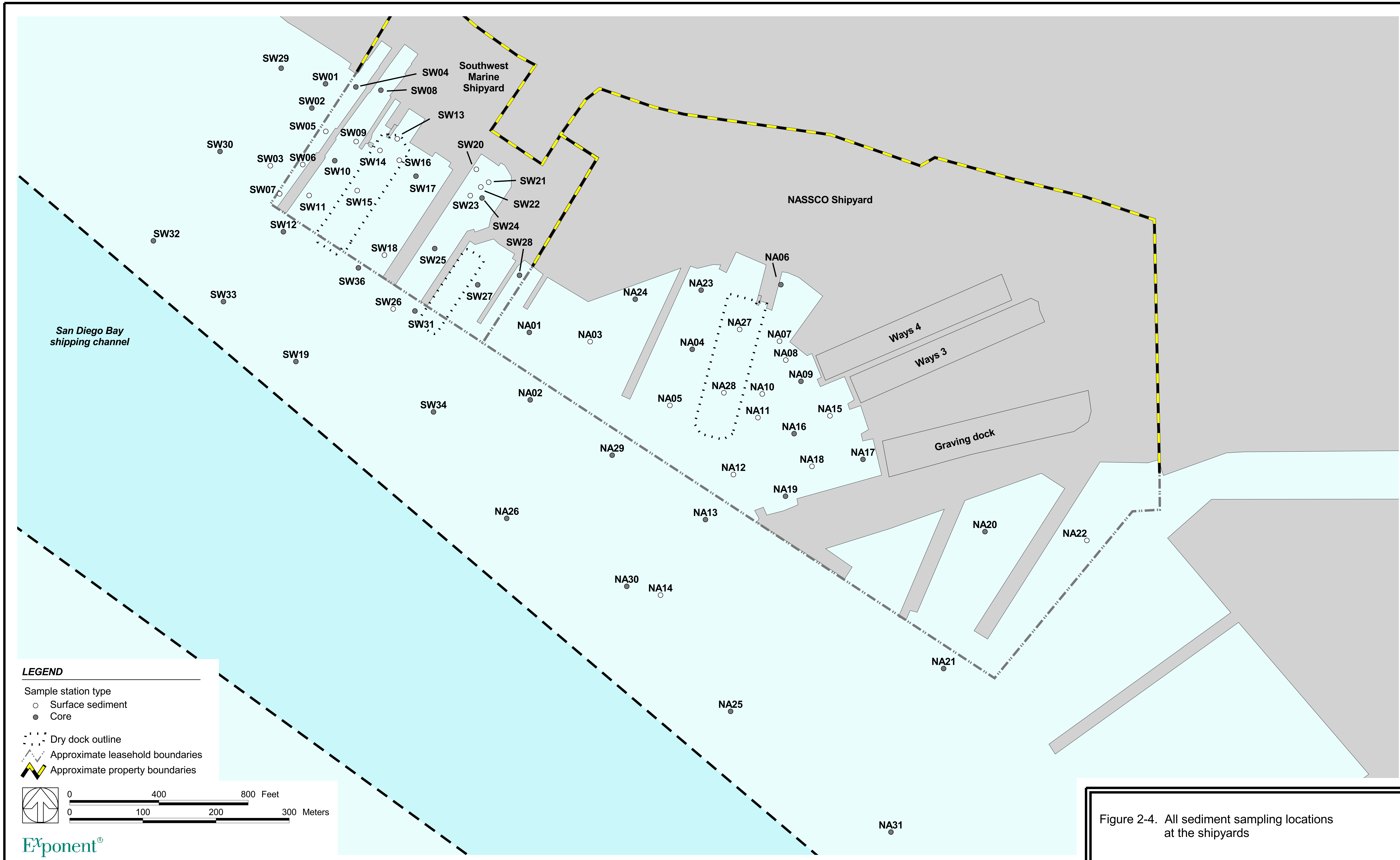


Figure 2-4. All sediment sampling locations at the shipyards

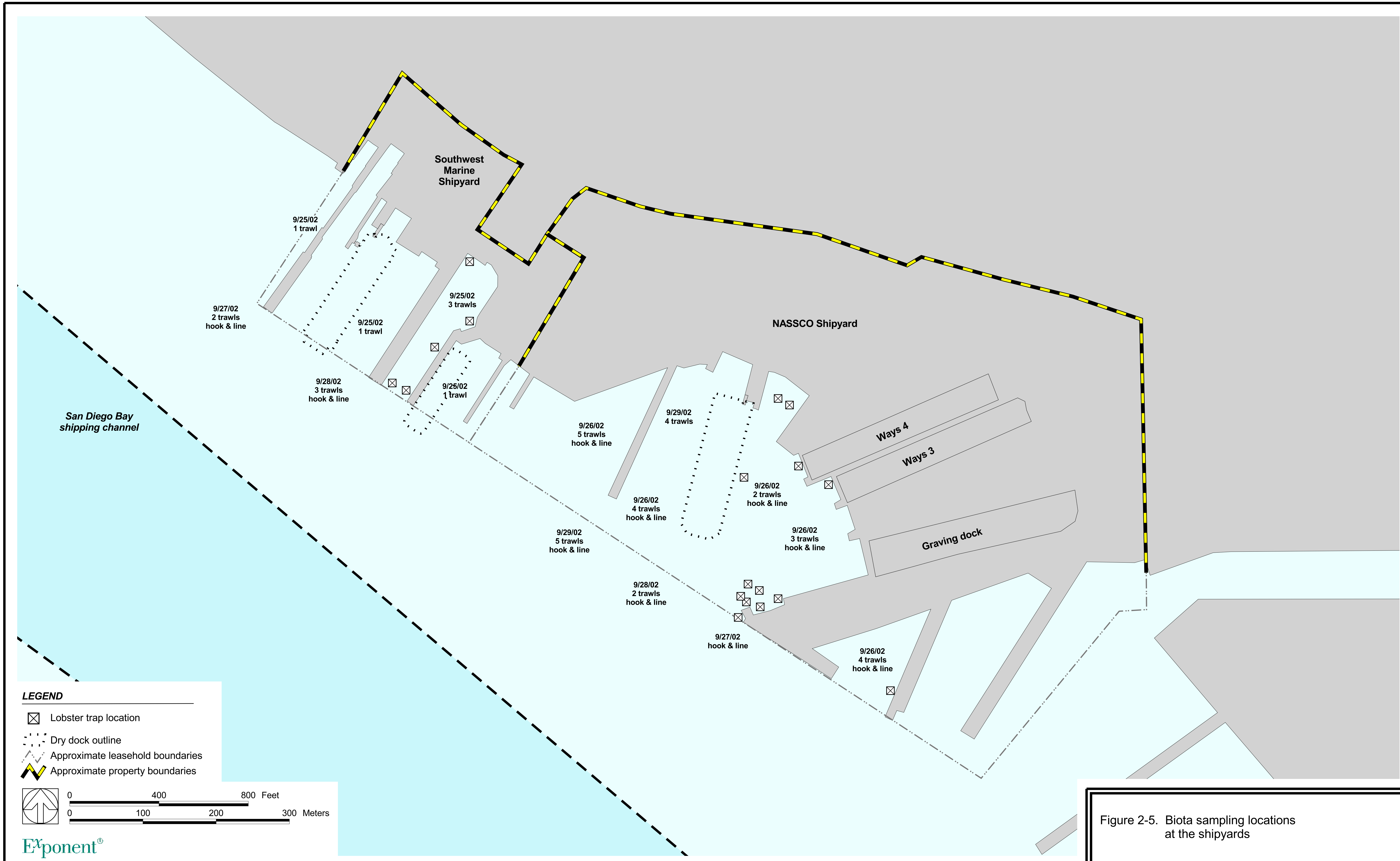


Figure 2-5. Biota sampling locations at the shipyards

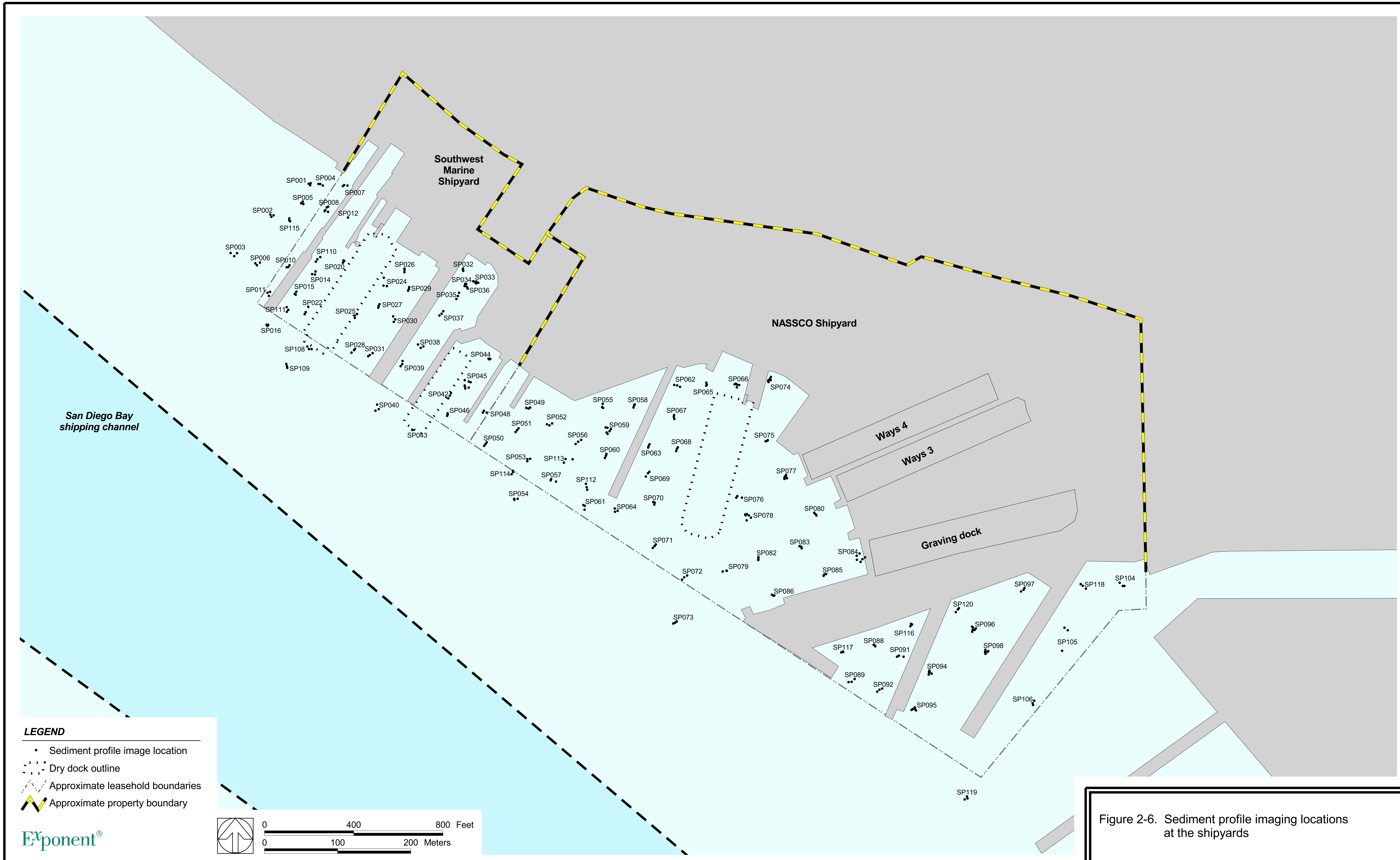


Figure 2-6. Sediment profile imaging locations at the shipyards

Exponent®

8601718/1731.002 1201 | Sep 03, 2003 | spi actual view | Fig 2-6 SPI locations layout | g:/swm_nassco/projects/shipyards_rpt.apr

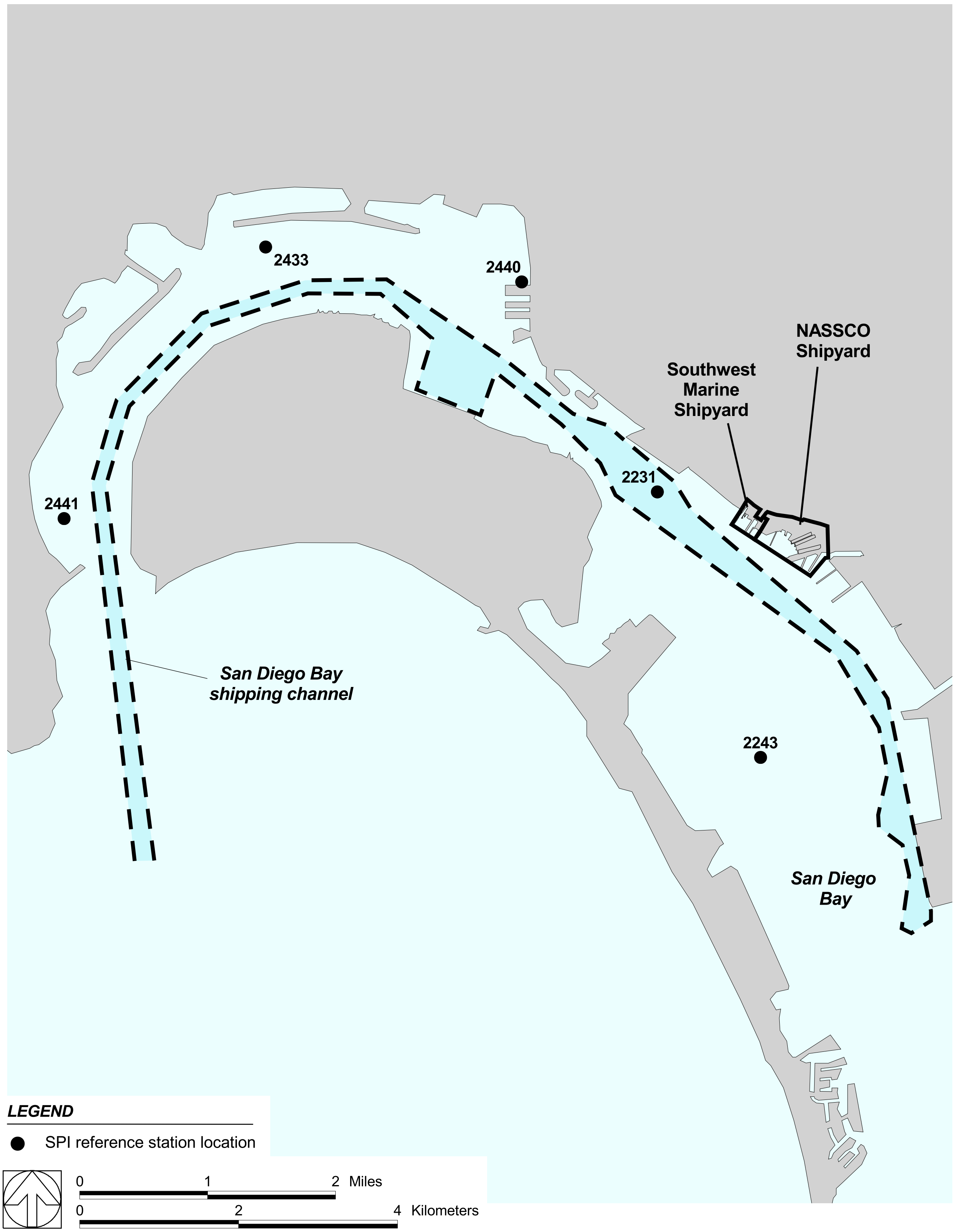


Figure 2-7. Reference station locations for sediment profile imaging

Exponent®

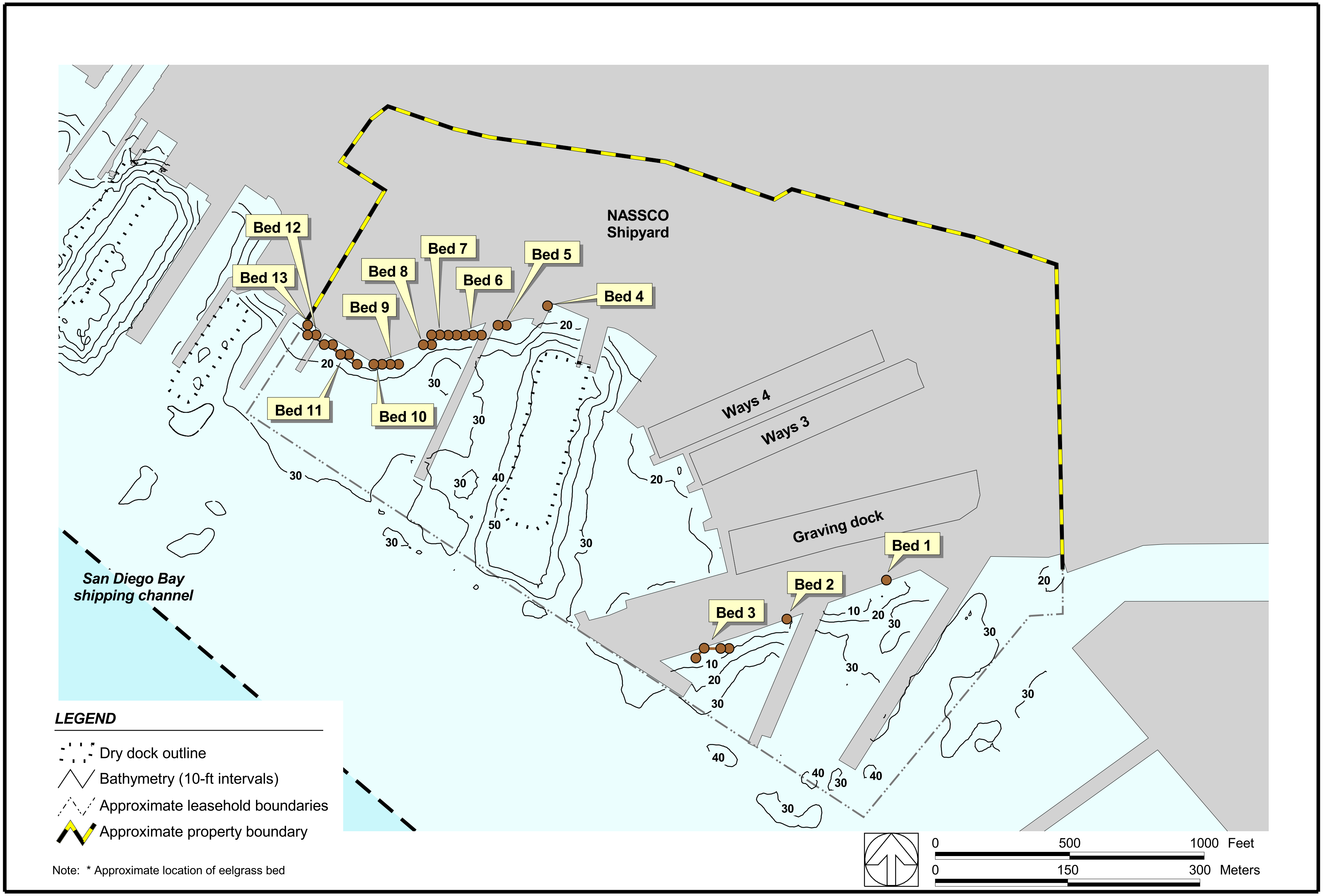


Figure 2-8. Distribution of eelgrass at NASSCO

Exponent®

8601718.002 1201 | Sep 22, 2003 | fig 02-8 eel grass NASSCO view | Fig 02-8 NASSCO eel grass layout | g:/swm_nassco/projects/shipyards_rpt.apr

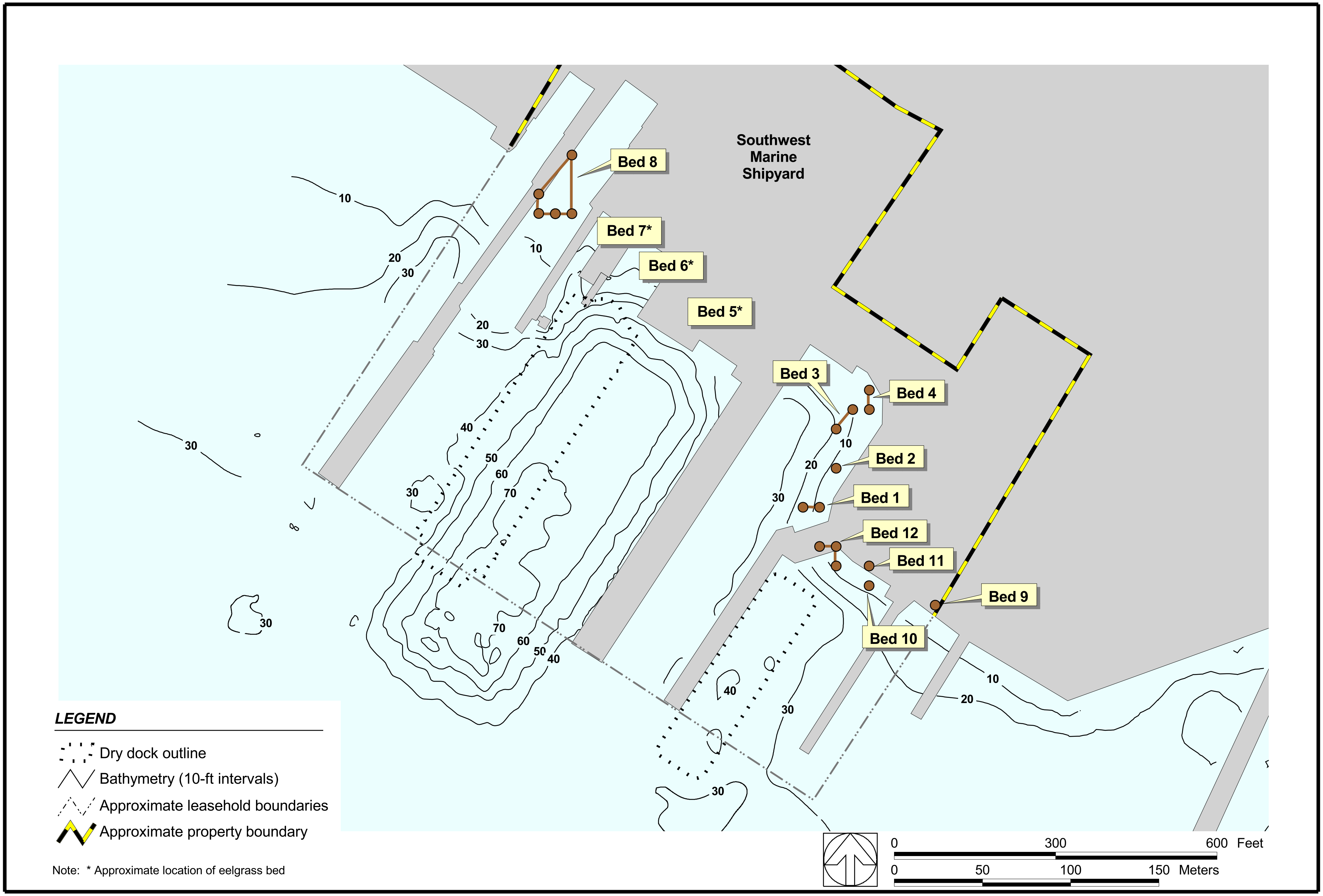


Figure 2-9. Distribution of eelgrass at Southwest Marine

Exponent®

8601718.002 1201 | Sep 22, 2003 | fig 02-9 eel grass SWM view | Fig 02-9 SWM eel grass layout | g:/swm_nassco/projects/shipyards_rpt.apr

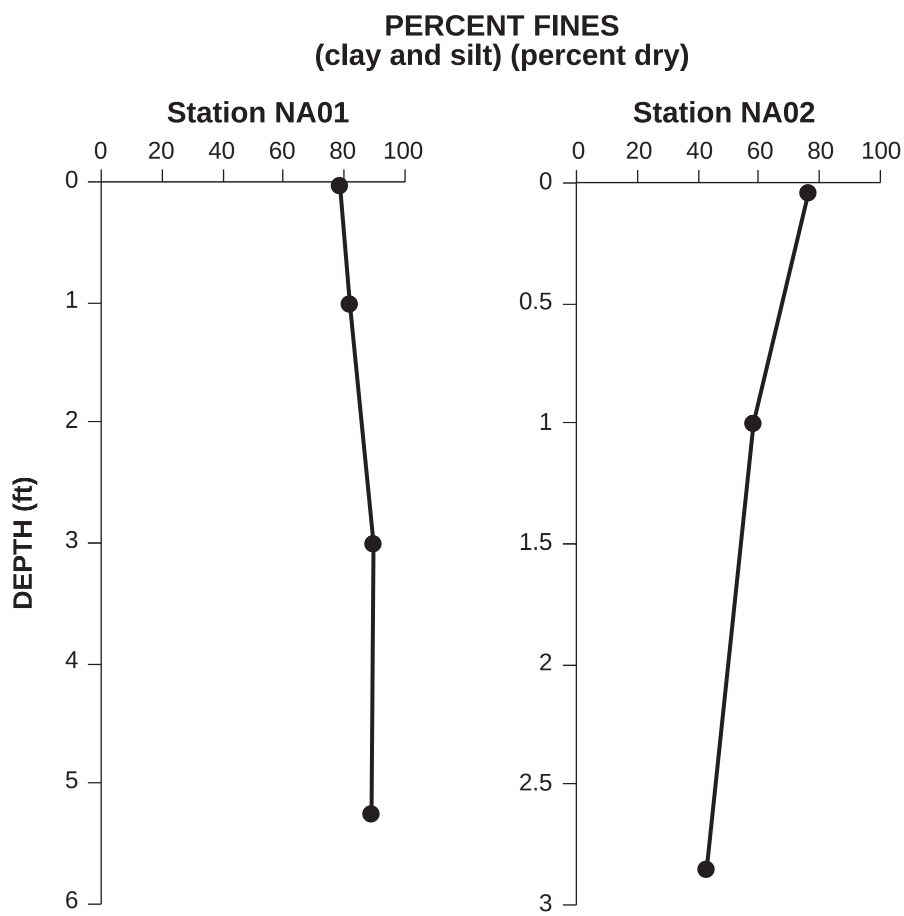


Figure 4-1. Example of ungraded and graded bedding in sediment grain size profiles

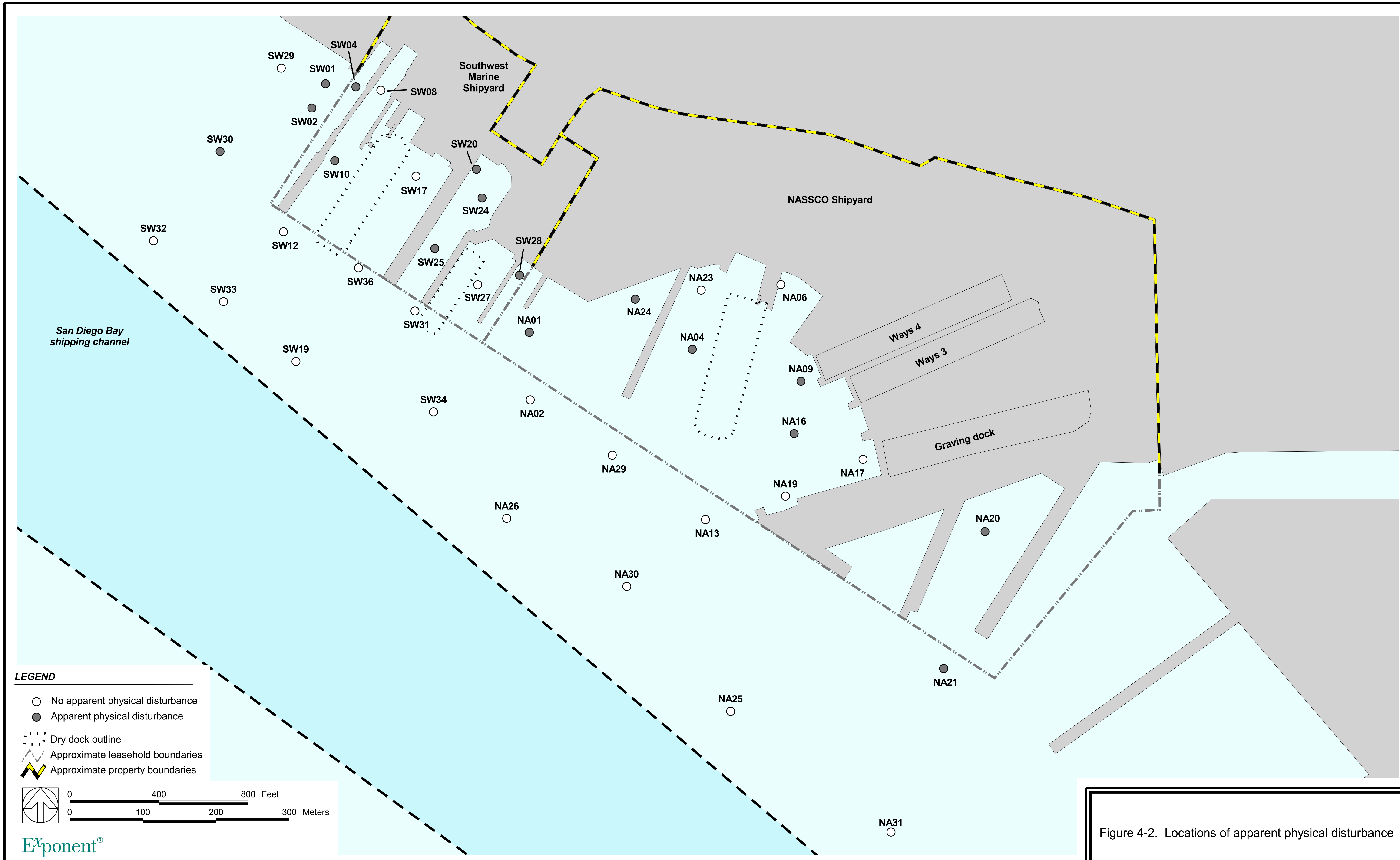


Figure 4-2. Locations of apparent physical disturbance

8601718/8601731.002 1201 | Oct 08, 2003 | fig 04-2 disturbance view | Fig 04-2 disturbance layout | g:/swm_nassco/projects/shipyards_rpt.apr

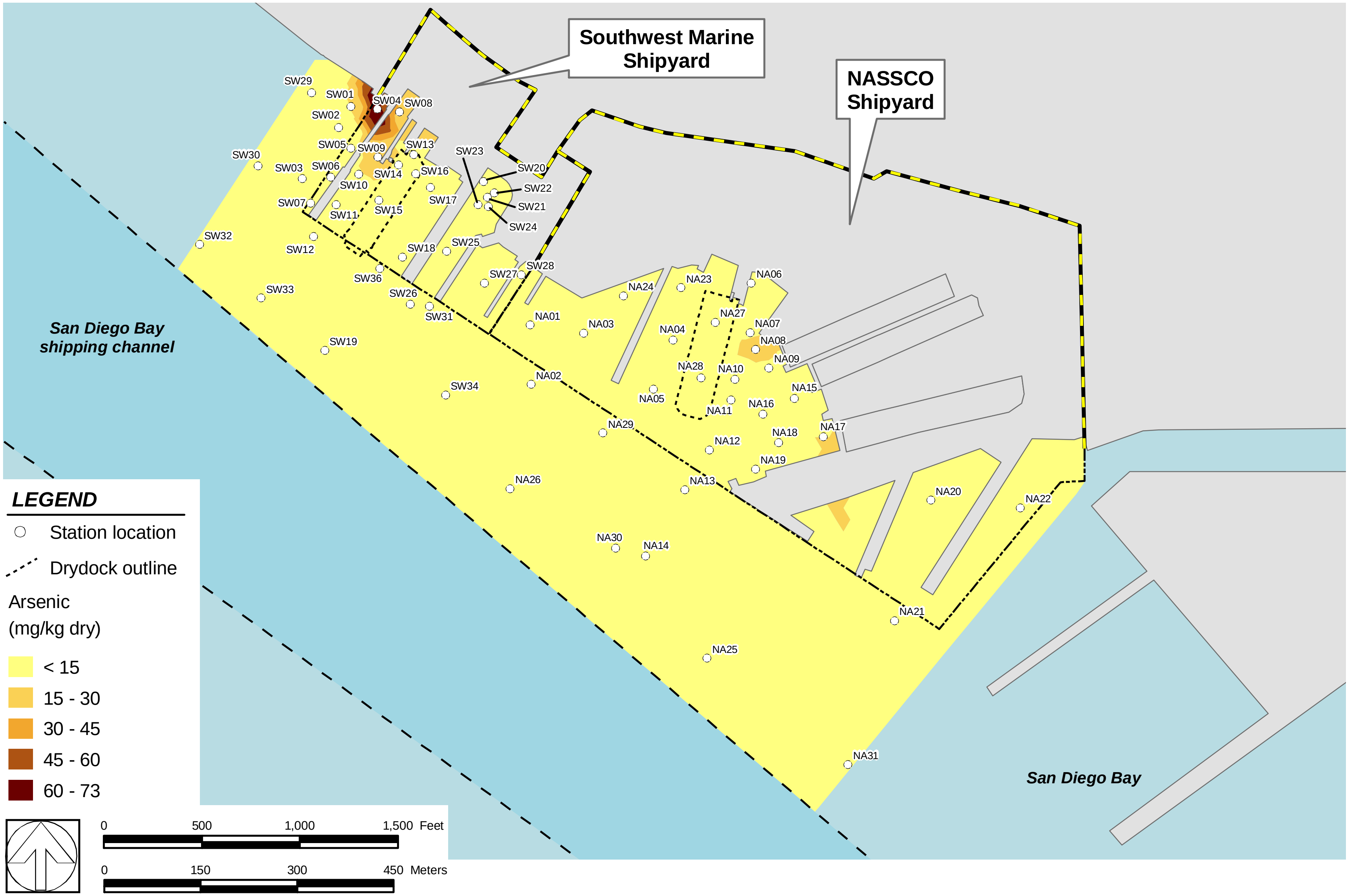


Figure 4-3. Surface sediment concentrations of arsenic

Exponent®

8601718/8601731.002 1201 | August 21, 2003 | G:\swm_nassco\projects\surface_data_cntrs_RRO.mxd

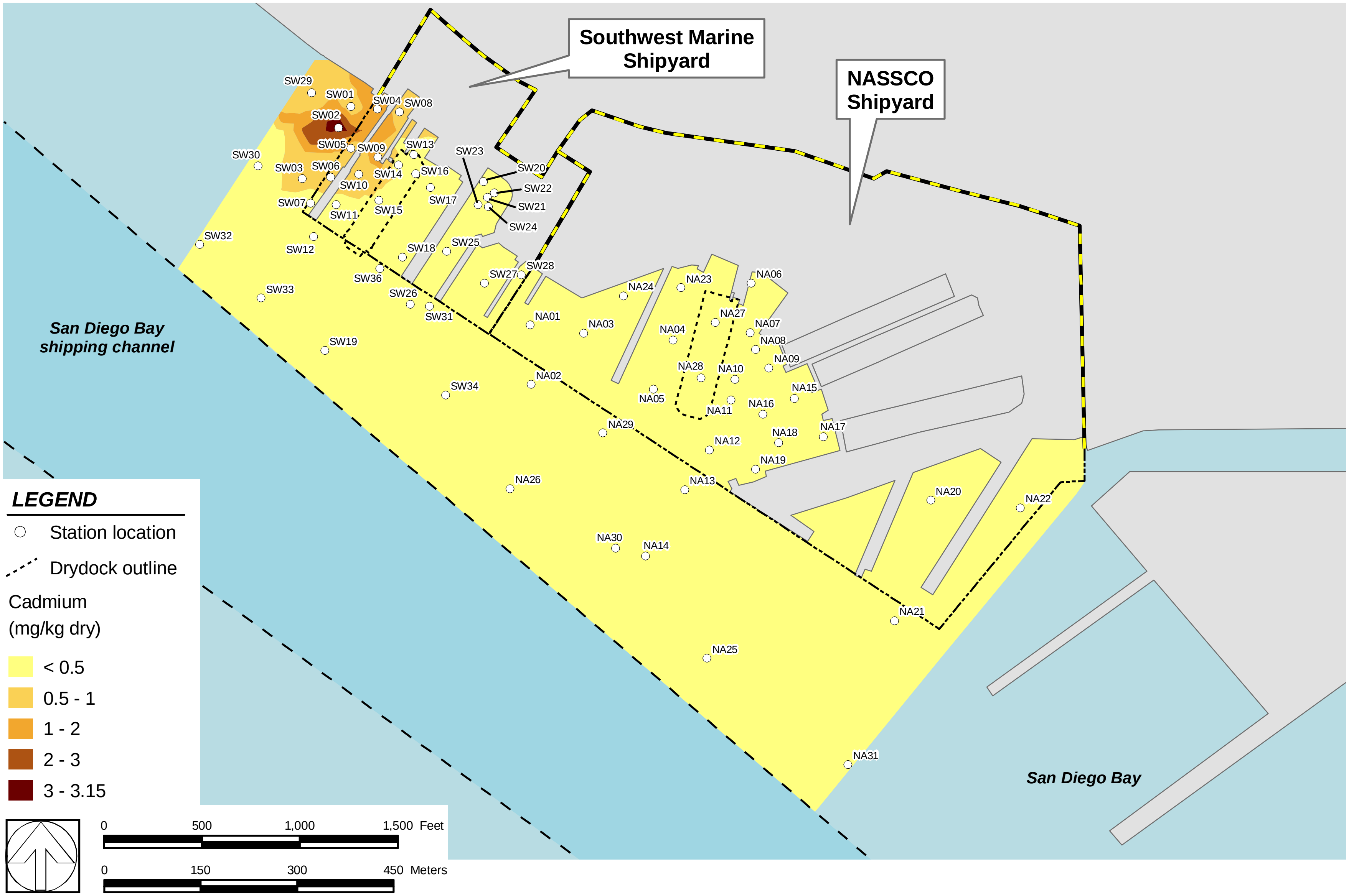


Figure 4-4. Surface sediment concentrations of cadmium

Exponent®

8601718/8601731.002 1201 | August 21, 2003 | G:\swm_nassco\projects\surface_data_cntrs_Cd.mxd

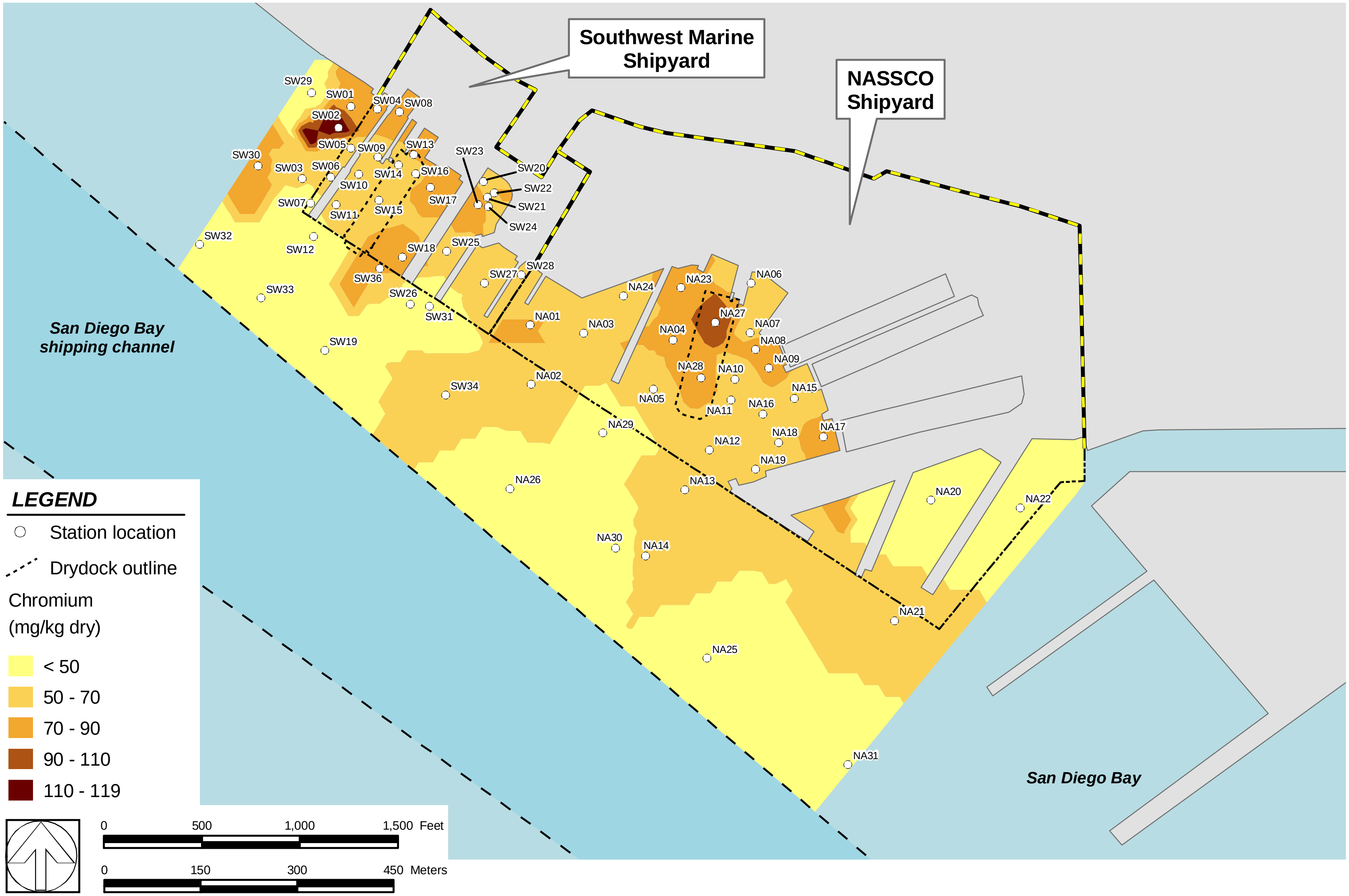


Figure 4-5. Surface sediment concentrations of chromium

Exponent®

8601718/8601731.002 1201 | August 21, 2003 | G:\swm_nassco\projects\surface_data_cntrs_Cr.mxd

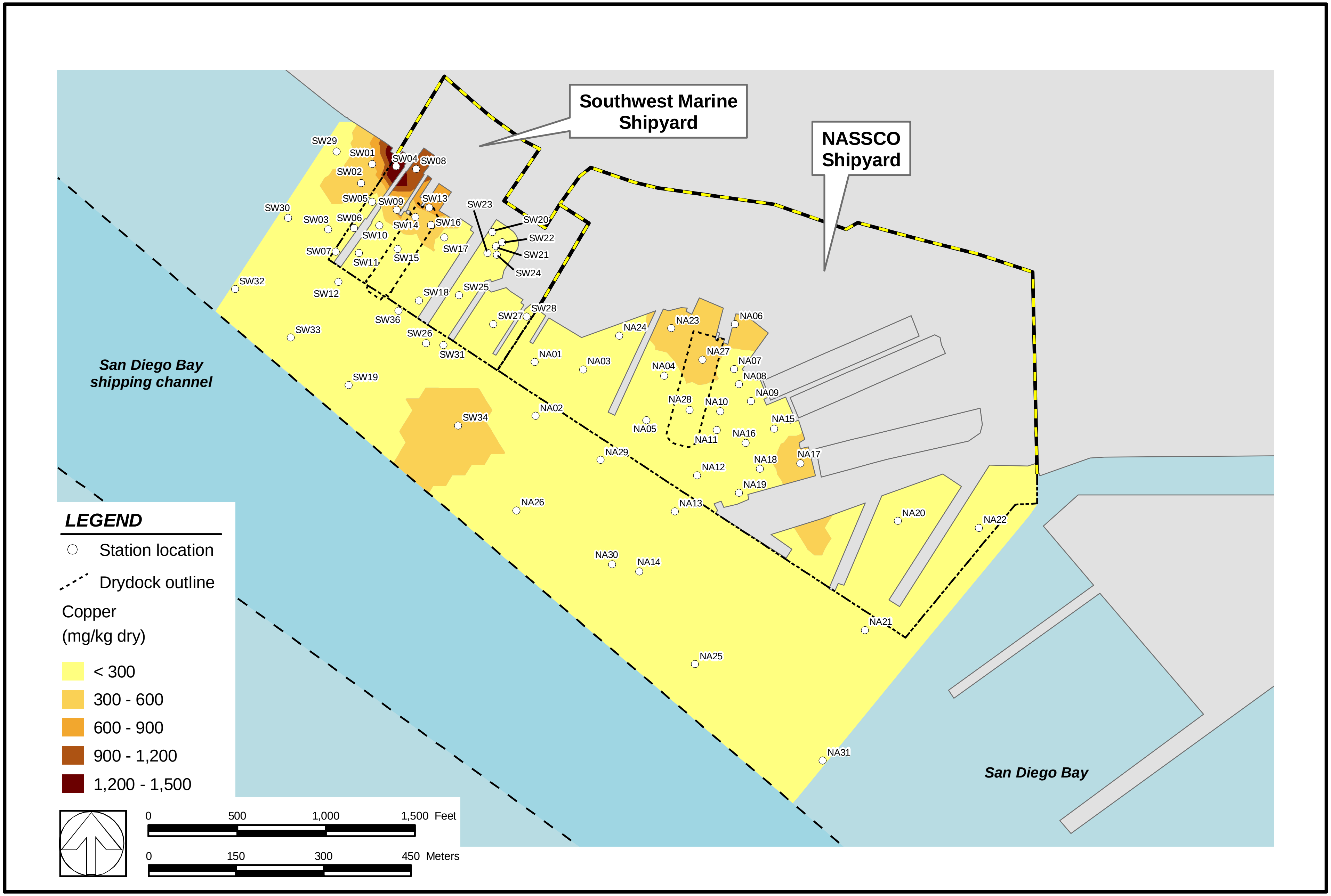


Figure 4-6. Surface sediment concentrations of copper

Exponent®

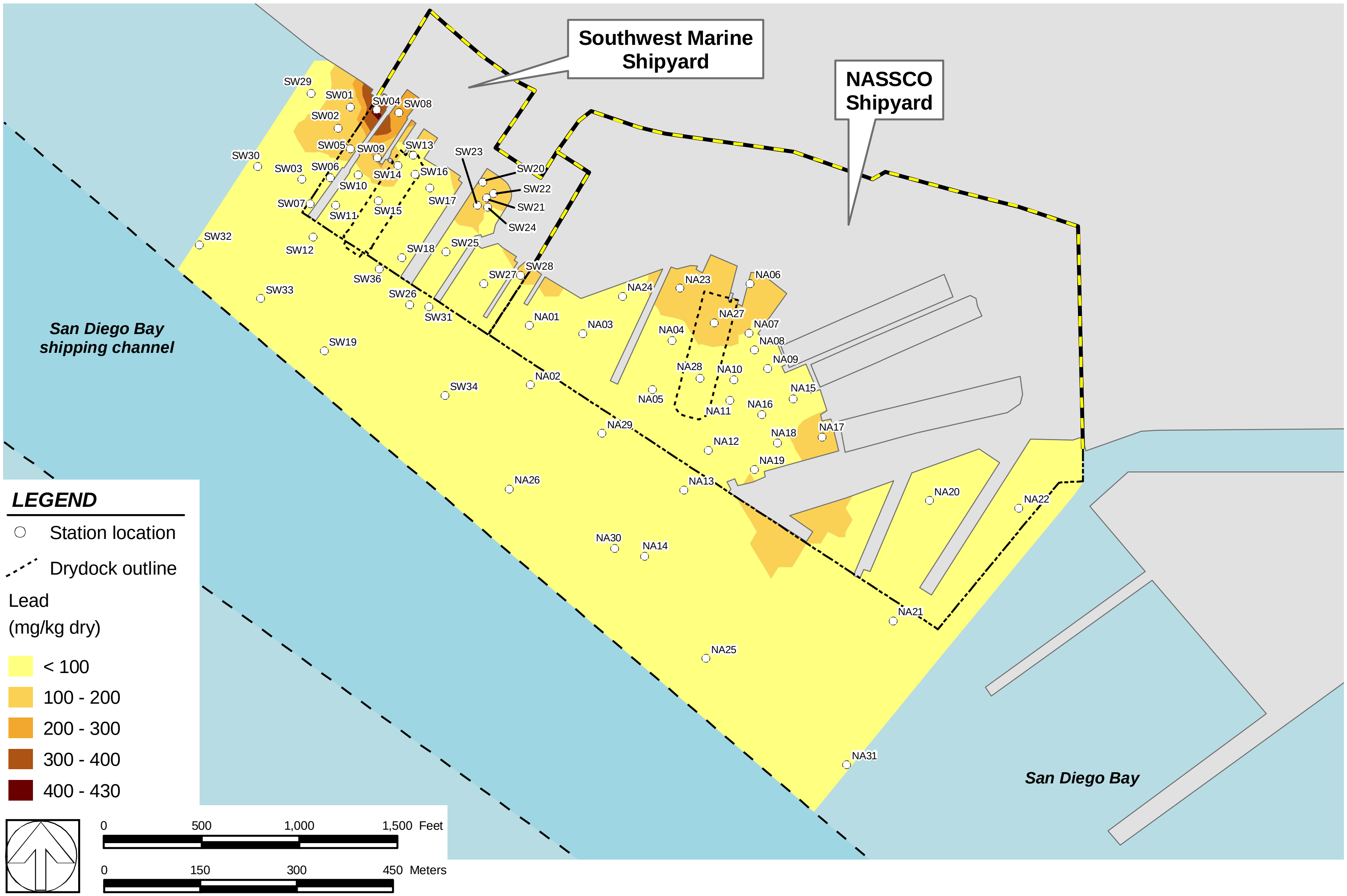


Figure 4-7. Surface sediment concentrations of lead

Exponent®

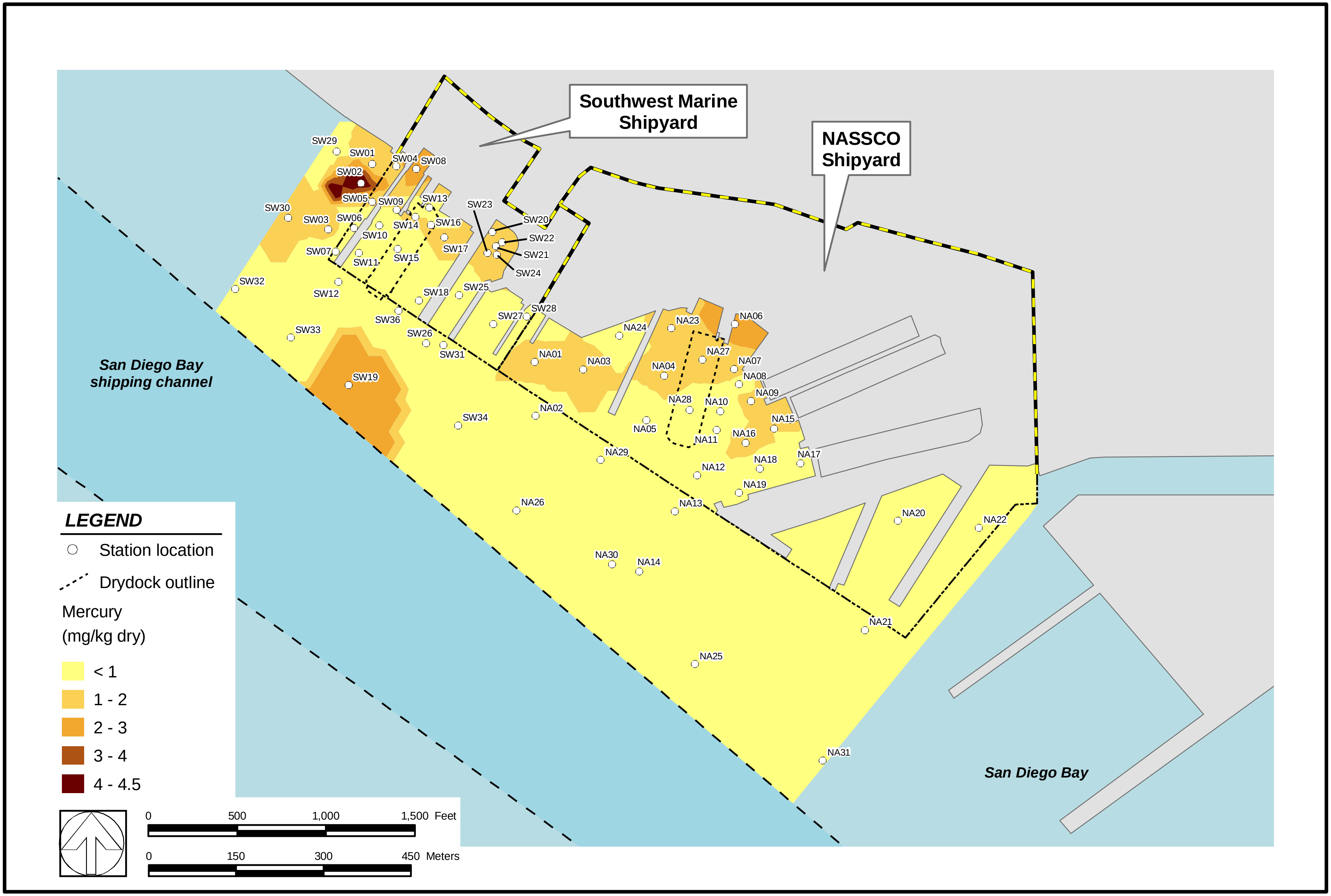


Figure 4-8. Surface sediment concentrations of mercury

Exponent®

8601718/8601731.002 1201 | August 22, 2003 | G:\swm_nassco\projects\surface_data_cntrs_Hg.mxd

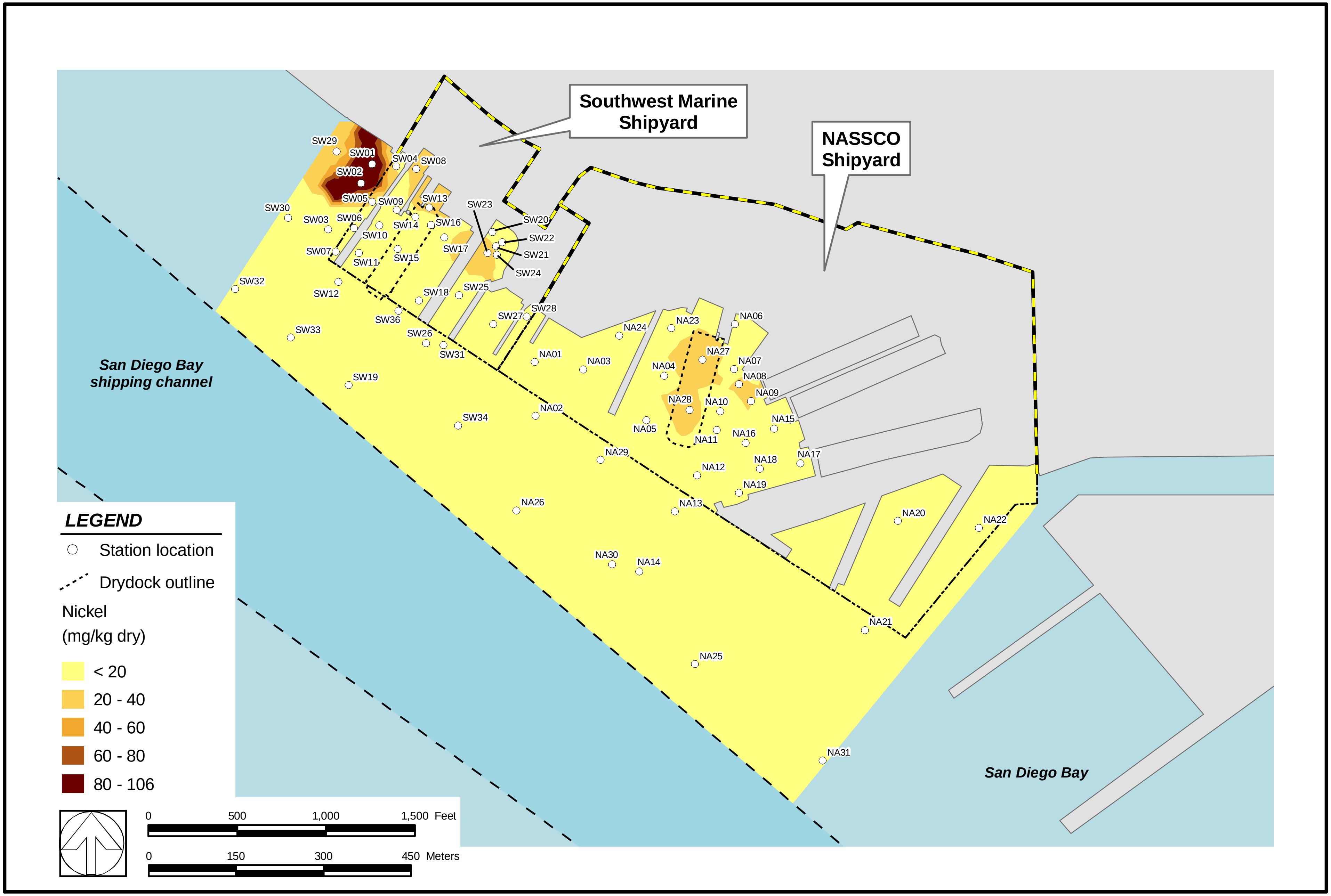


Figure 4-9. Surface sediment concentrations of nickel

Exponent®

8601718/8601731.002 1201 | August 22, 2003 | G:\swm_nassco\projects\surface_data_cntrs_Ni.mxd

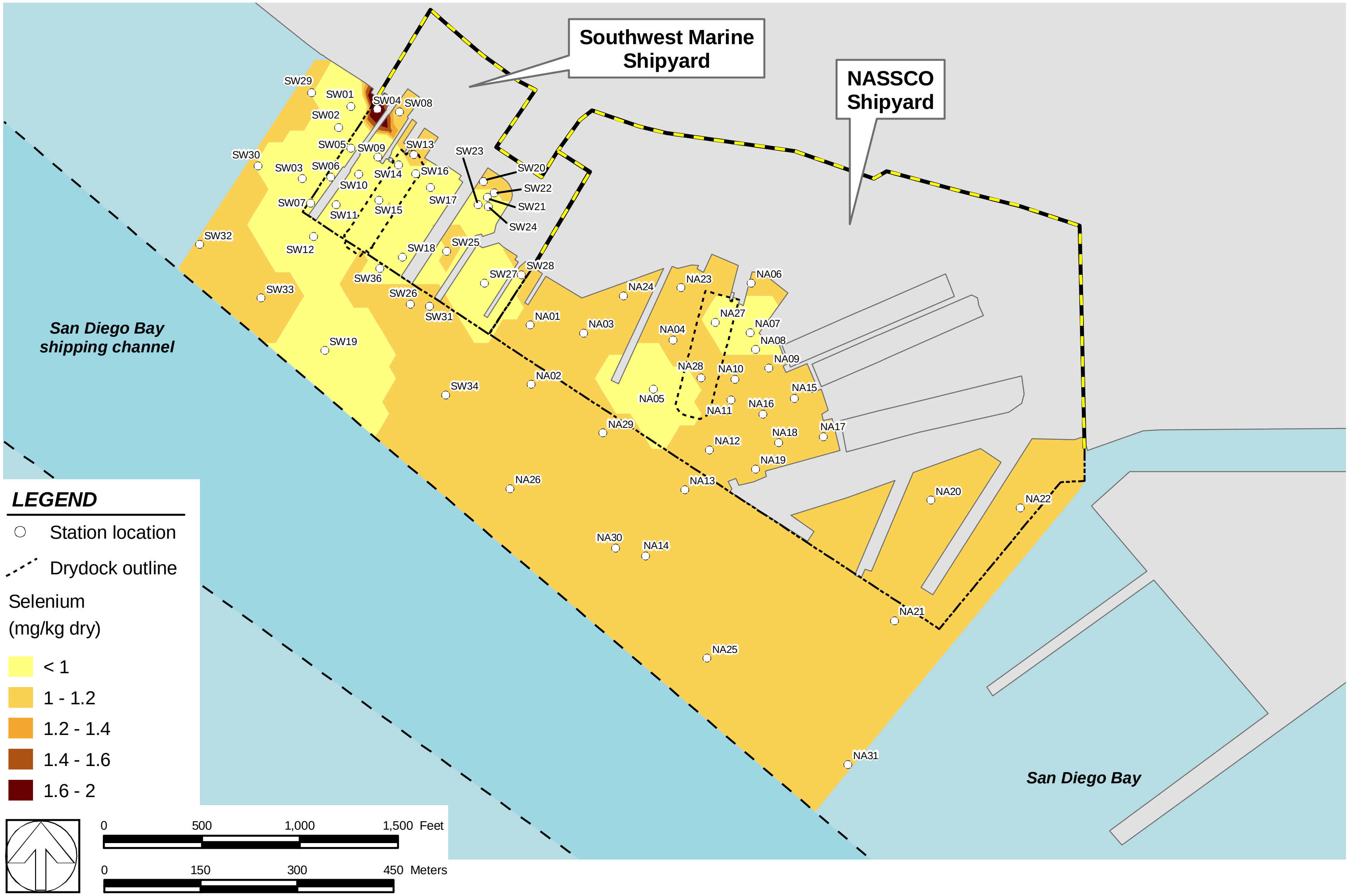


Figure 4-10. Surface sediment concentrations of selenium

Exponent®

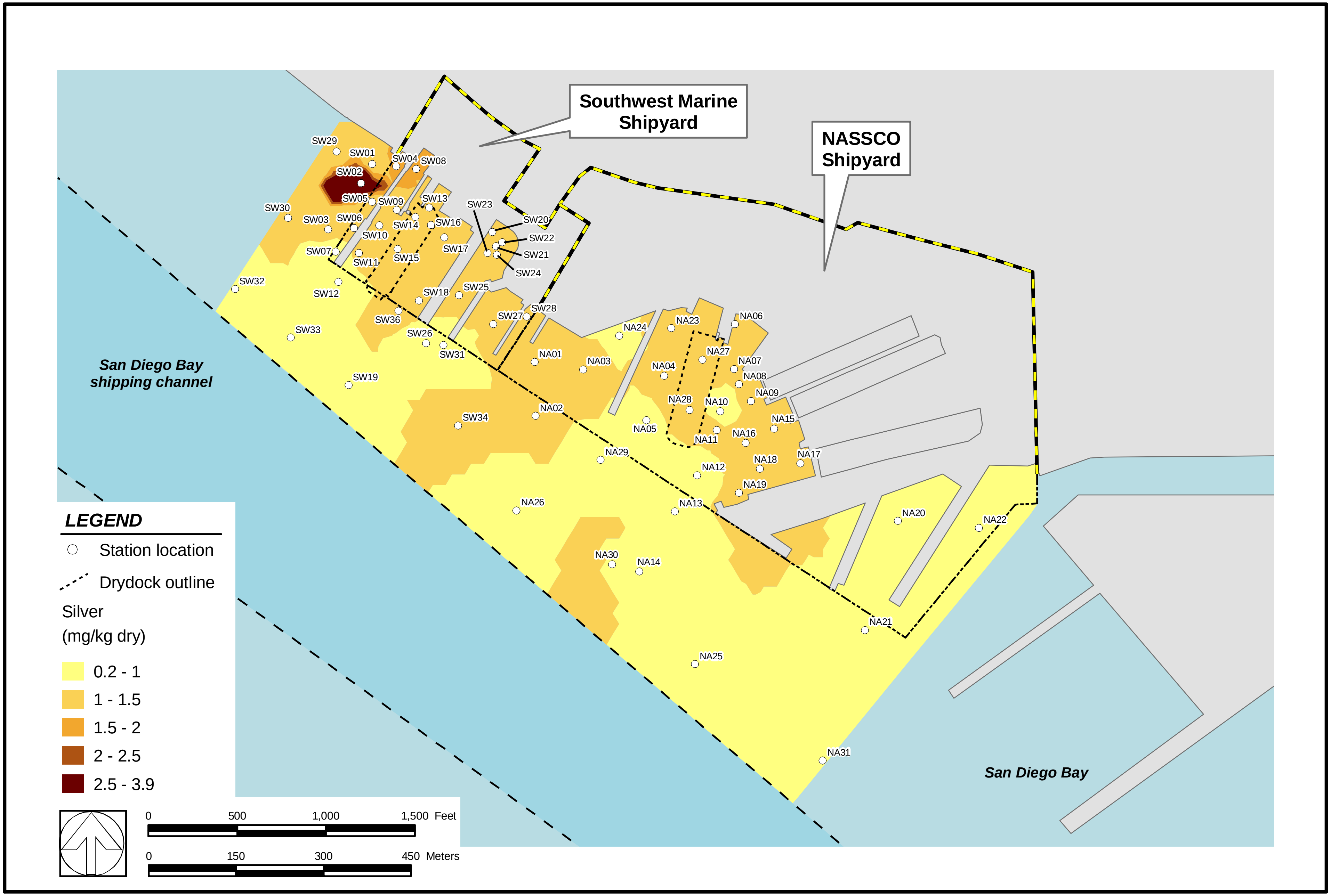


Figure 4-11. Surface sediment concentrations of silver

Exponent®

8601718/8601731.002 1201 | August 22, 2003 | G:\swm_nassco\projects\surface_data_cntrs_Ag.mxd

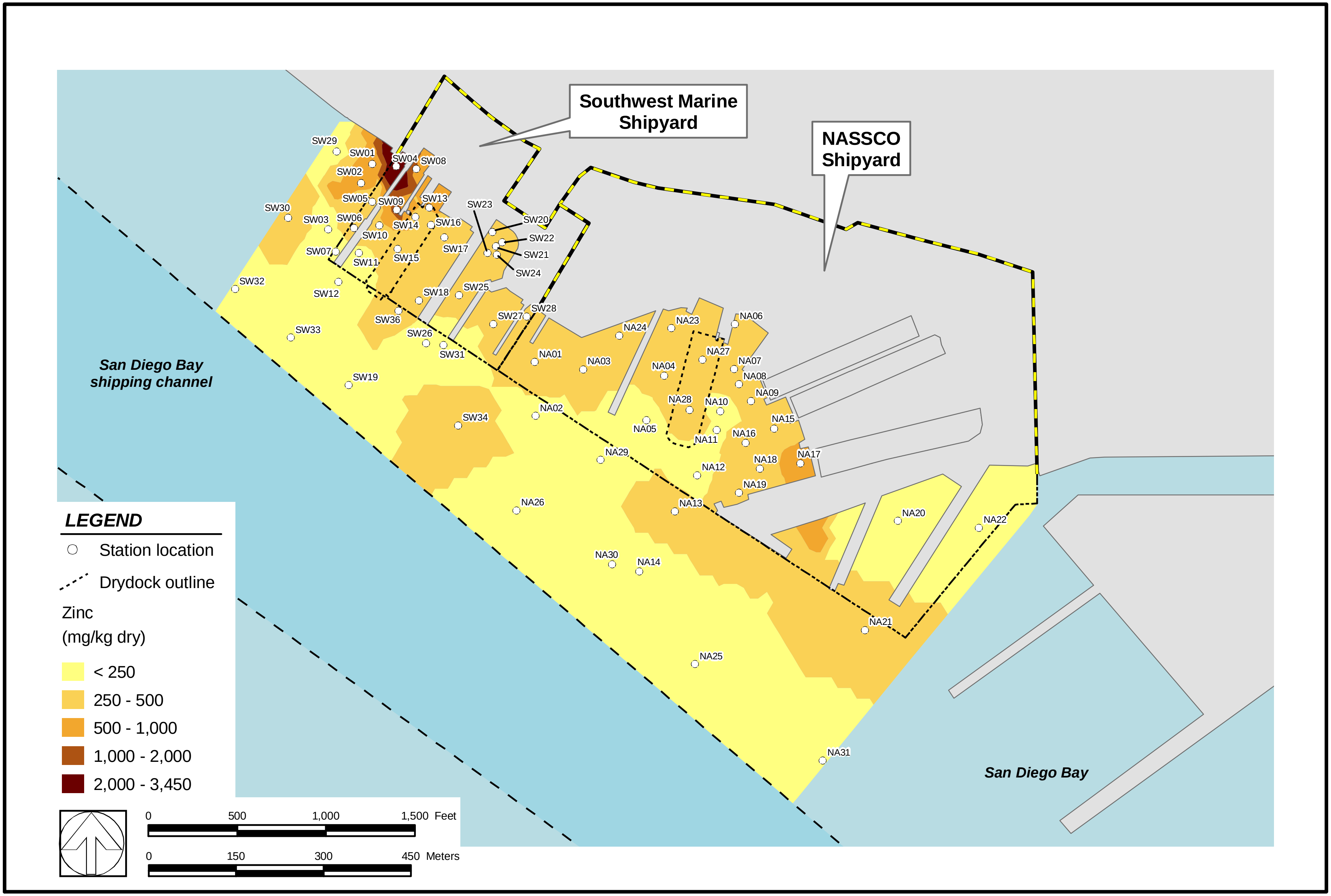


Figure 4-12. Surface sediment concentrations of zinc

Exponent®

8601718/8601731.002 1201 | August 22, 2003 | G:\swm_nassco\projects\surface_data_cntrs_Se.mxd

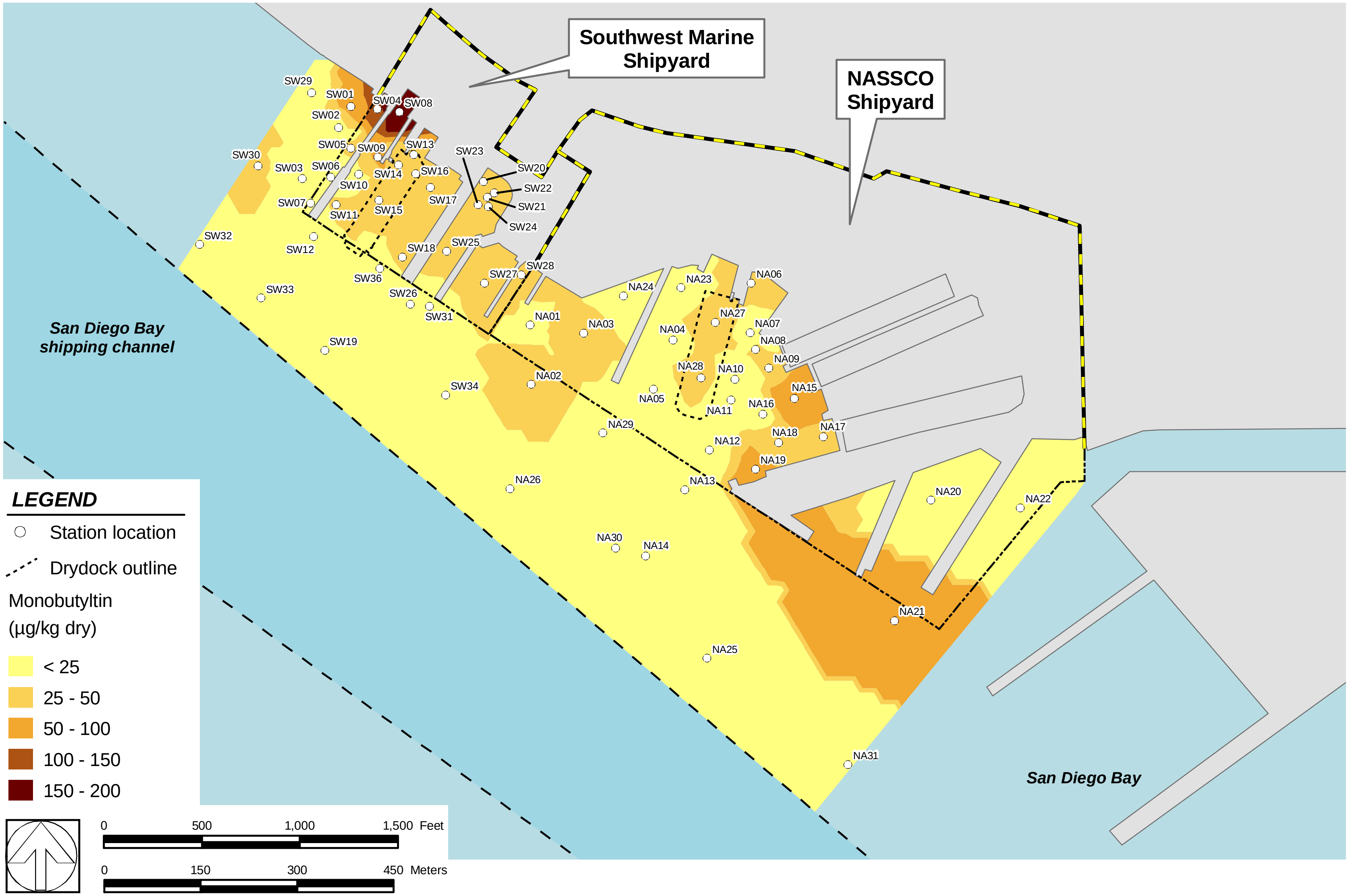


Figure 4-13. Surface sediment concentrations of monobutyltin

Exponent®

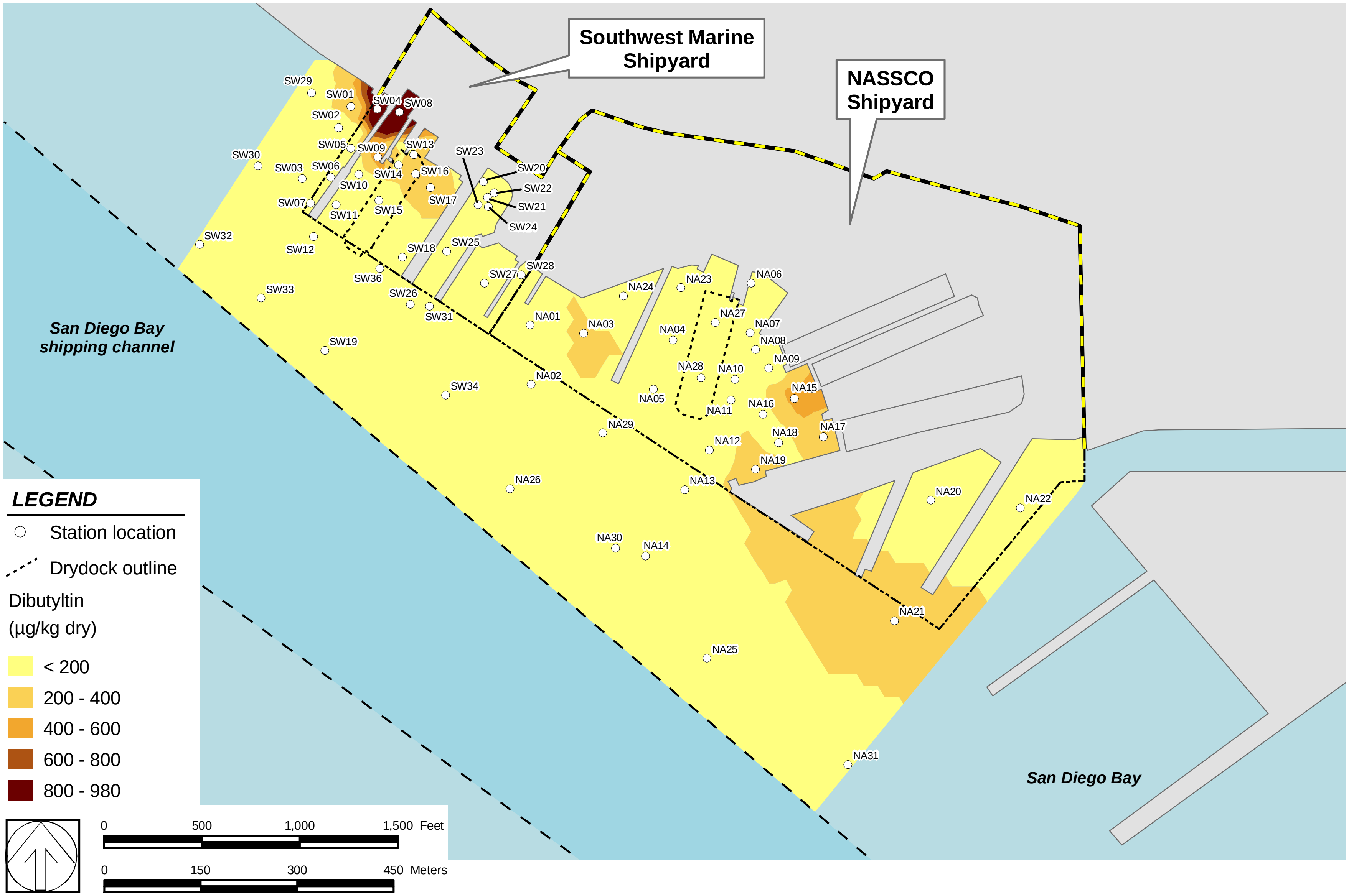


Figure 4-14. Surface sediment concentrations of dibutyltin

Exponent®

8601718/8601731.002 1201 | August 22, 2003 | G:\swm_nassco\projects\surface_data_cntrs_dibutyltin.mxd

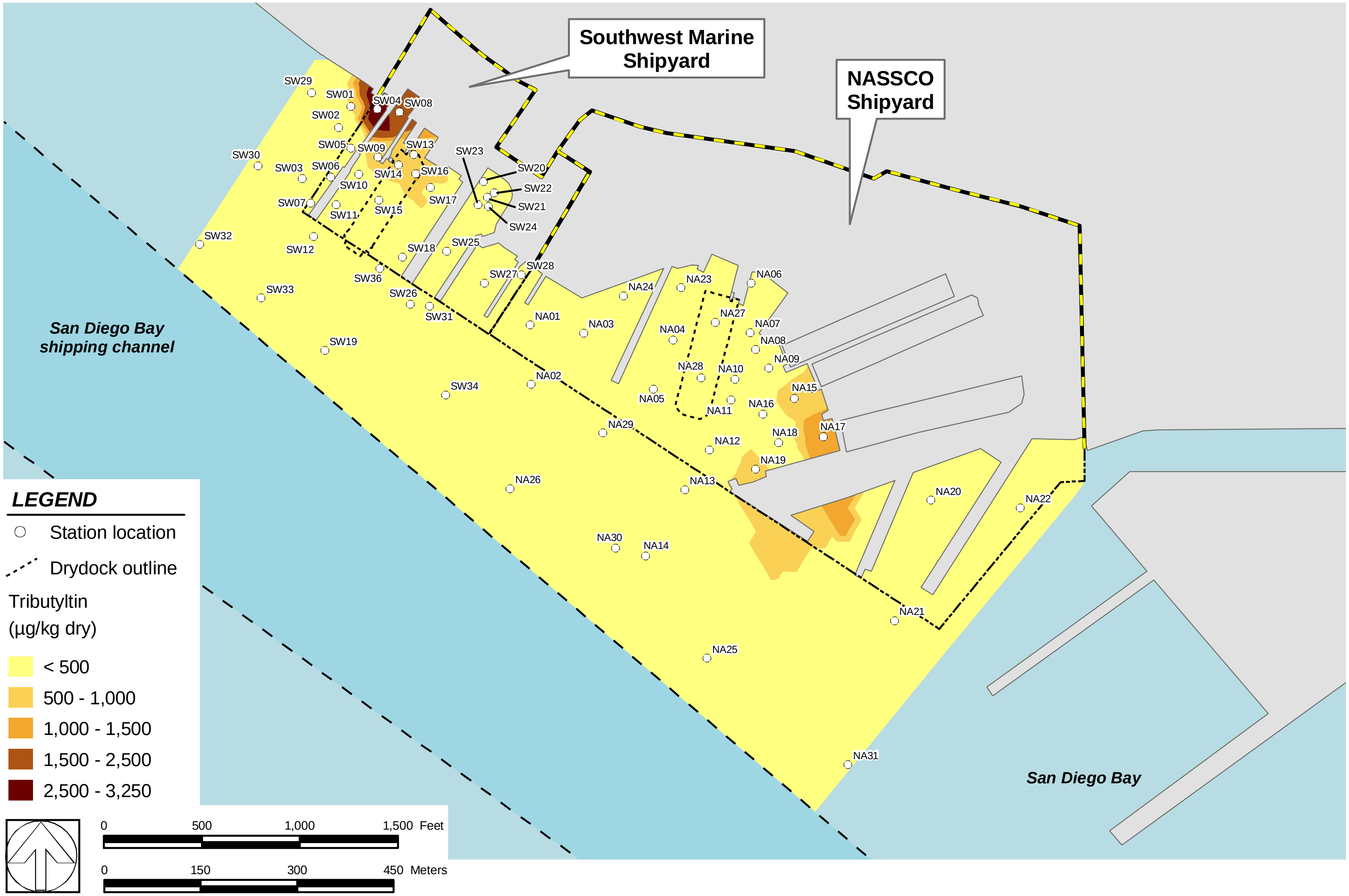


Figure 4-15. Surface sediment concentrations of tributyltin

Exponent®

8601718/8601731.002 1201 | August 22, 2003 | G:\swm_nassco\projects\surface_data_cntrs_tbt.mxd

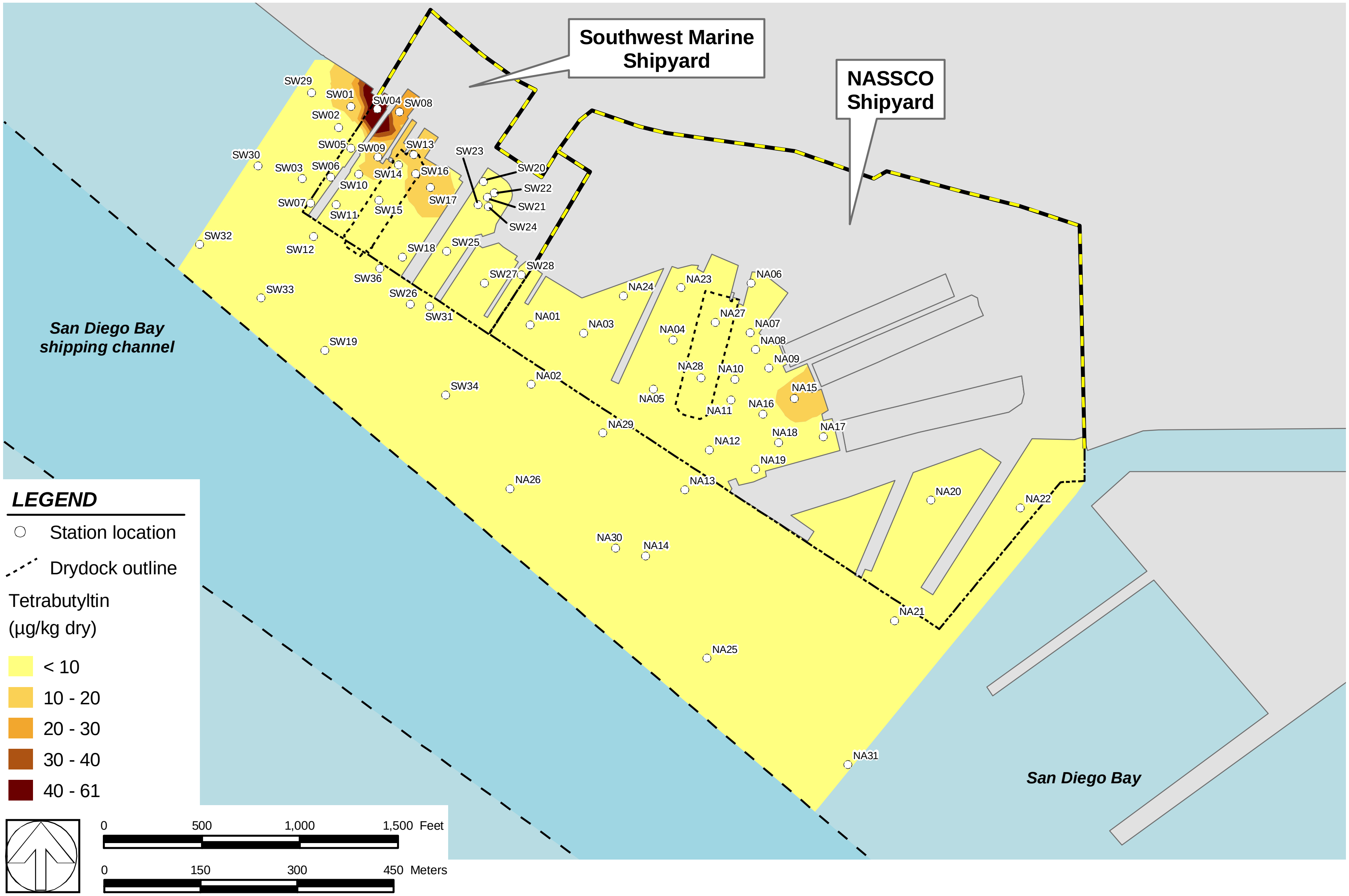


Figure 4-16. Surface sediment concentrations of tetrabutyltin

Exponent®

8601718/8601731.002 1201 | August 22, 2003 | G:\swm_nassco\projects\surface_data_cntrs_tetrabutyltin.mxd

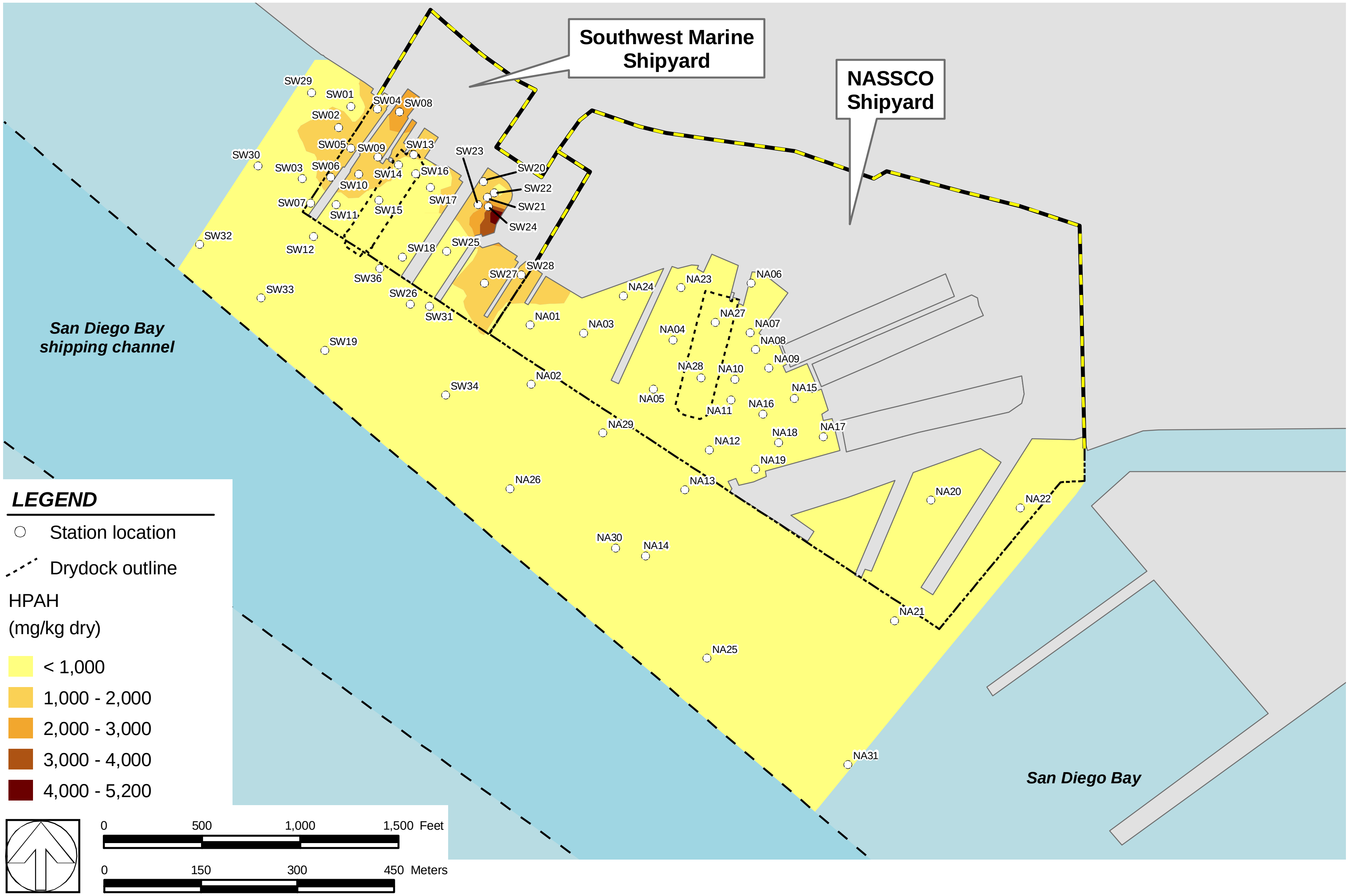


Figure 4-17. Surface sediment concentrations of high molecular weight polycyclic aromatic hydrocarbon (HPAH)

Exponent®

8601718/8601731.002 1201 | August 22, 2003 | G:\swm_nassco\projects\surface_data_cntrs_HPAH.mxd

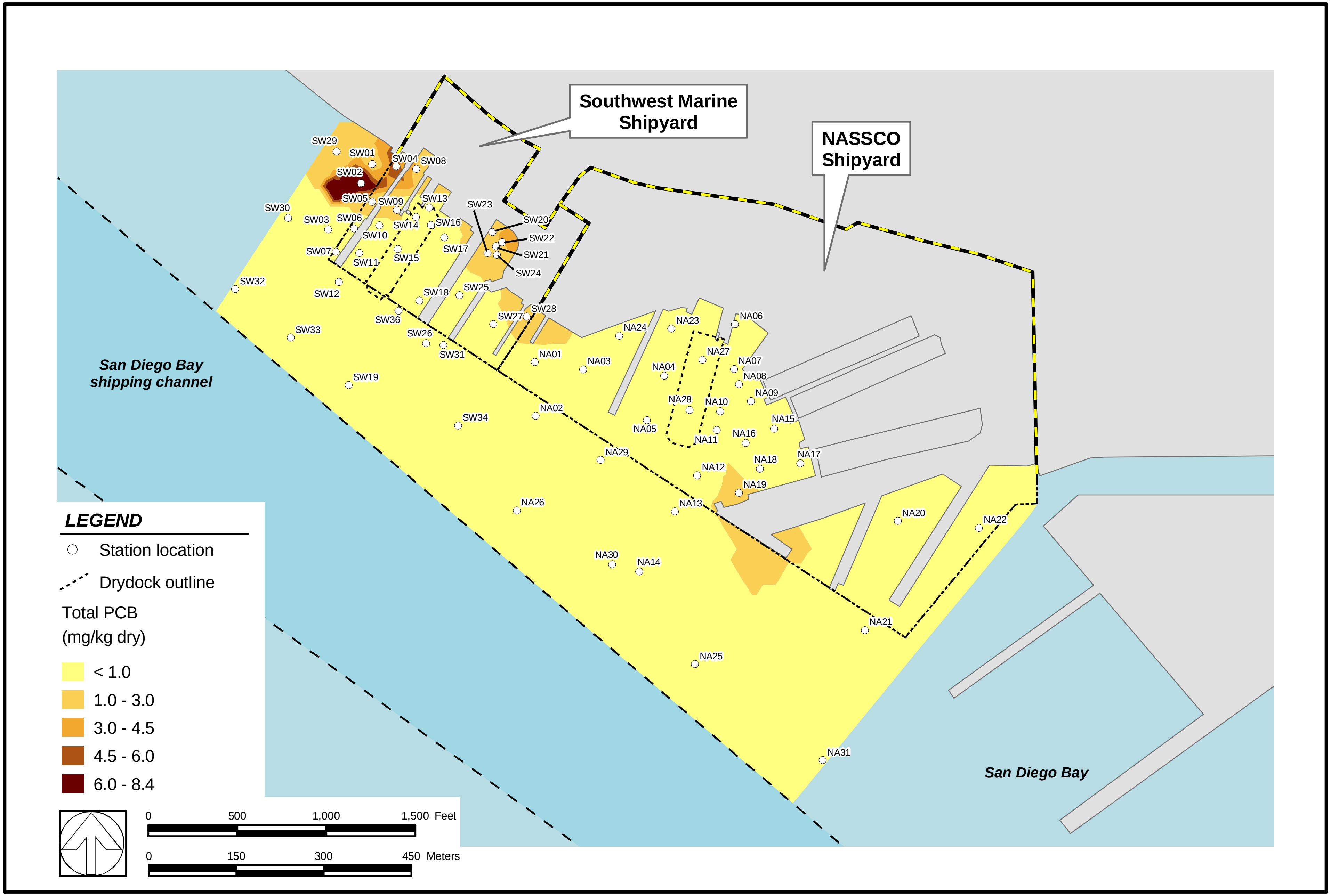


Figure 4-18. Surface sediment concentrations of total PCB homologs

Exponent®

8601718/8601731.002 1201 | August 22, 2003 | G:\swm_nassco\projects\surface_data_cntrs_PCB.mxd

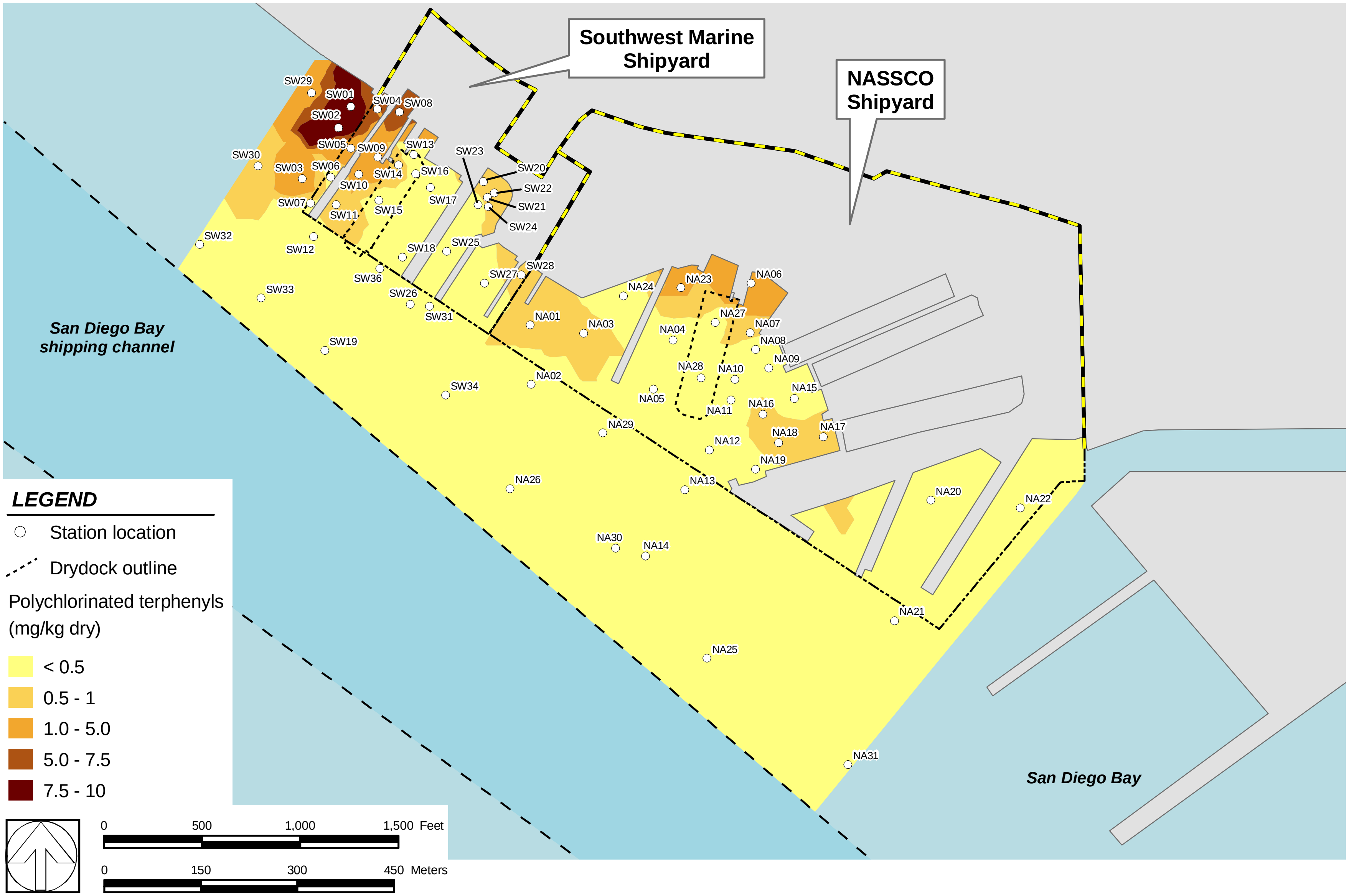


Figure 4-19. Surface sediment concentrations of polychlorinated terphenyls

Exponent®

8601718/8601731.002 1201 | August 22, 2003 | G:\swm_nassco\projects\surface_data_cntrs_PCT.mxd

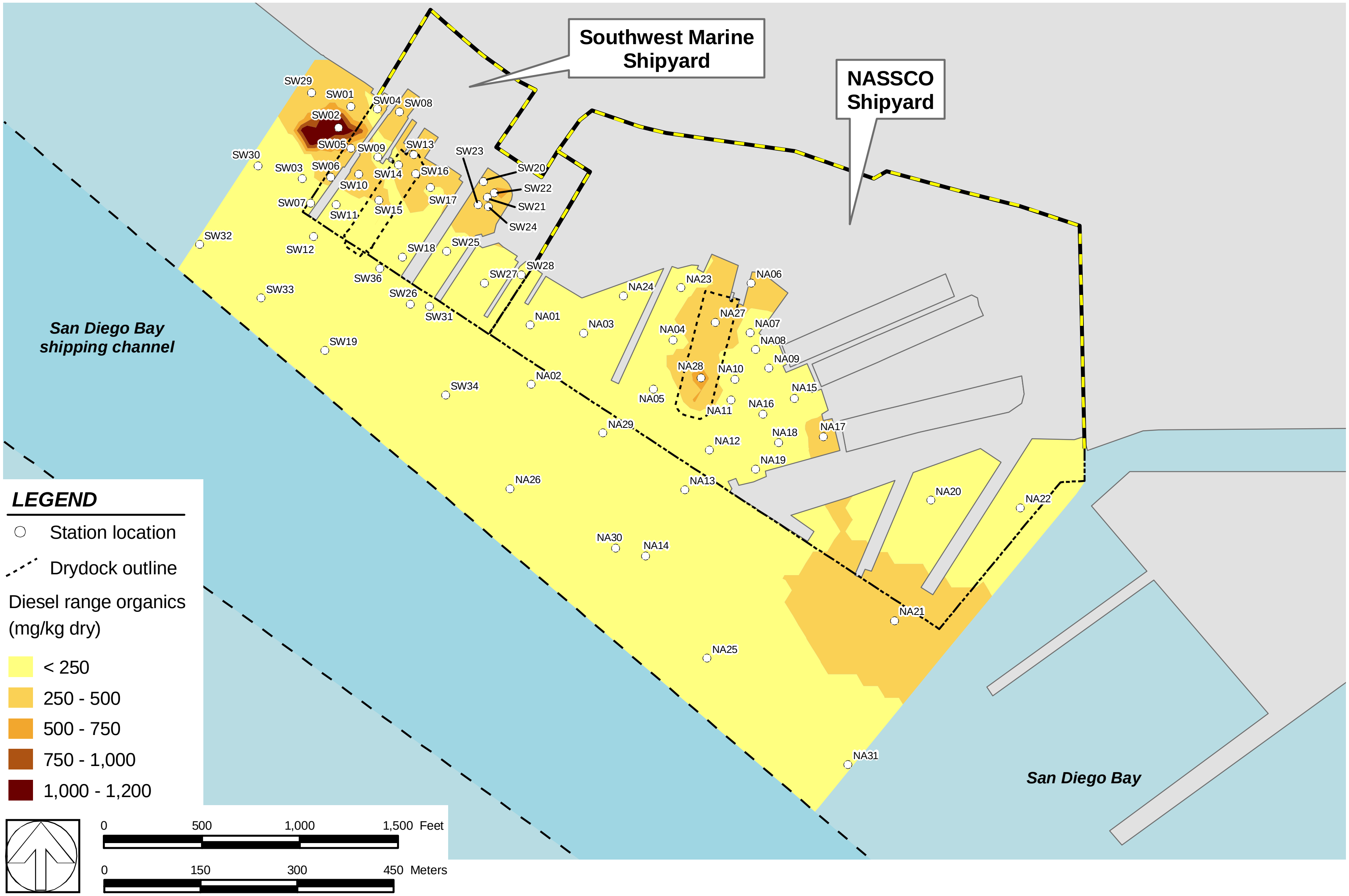


Figure 4-20. Surface sediment concentrations of diesel range organics

Exponent®

8601718/8601731.002 1201 | August 21, 2003 | G:\swm_nassco\projects\surface_data_cntrs_DRO.mxd

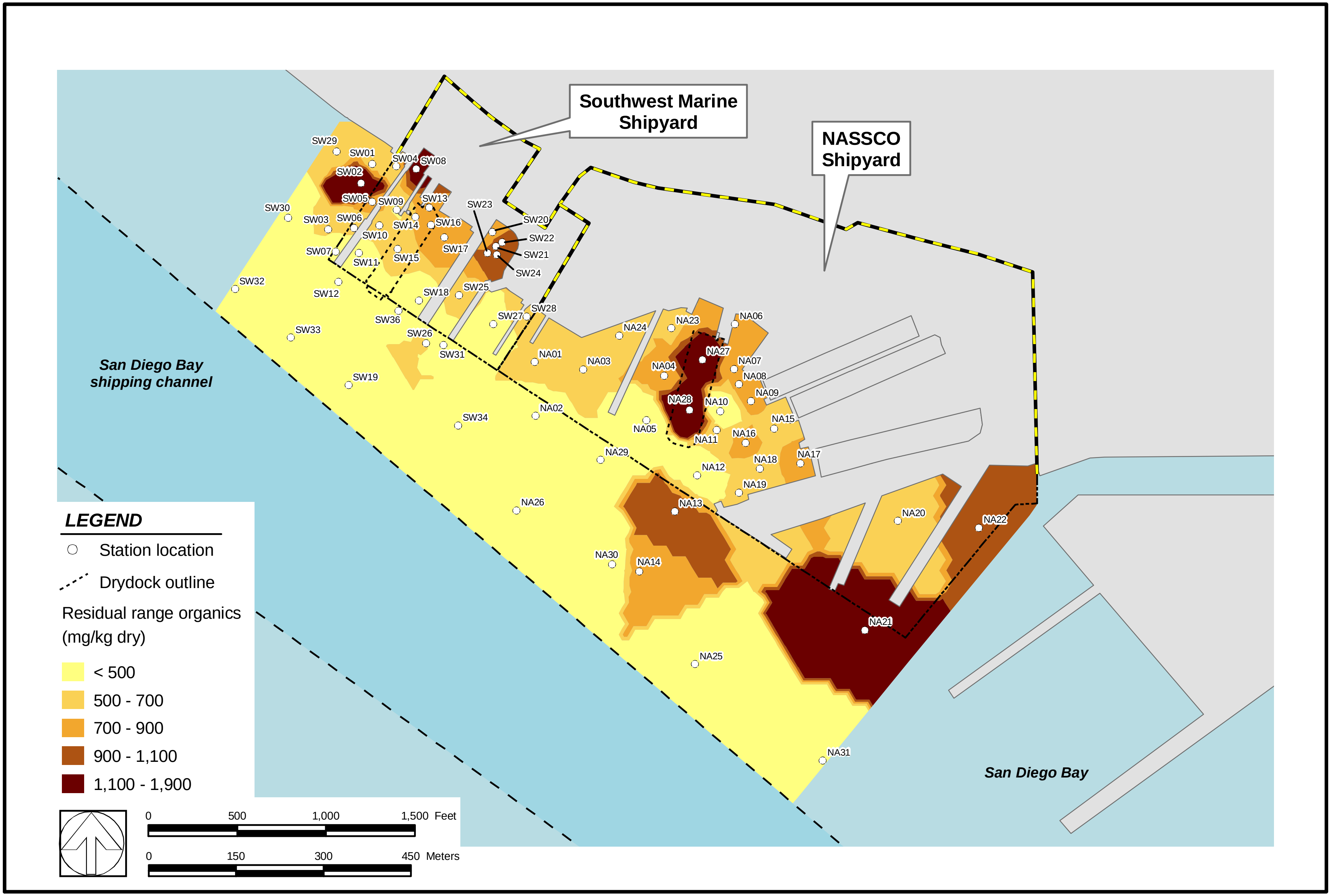


Figure 4-21. Surface sediment concentrations of residual range organics

Exponent®

8601718/8601731.002 1201 | August 21, 2003 | G:\swm_nassco\projects\surface_data_cntrs_RRO.mxd

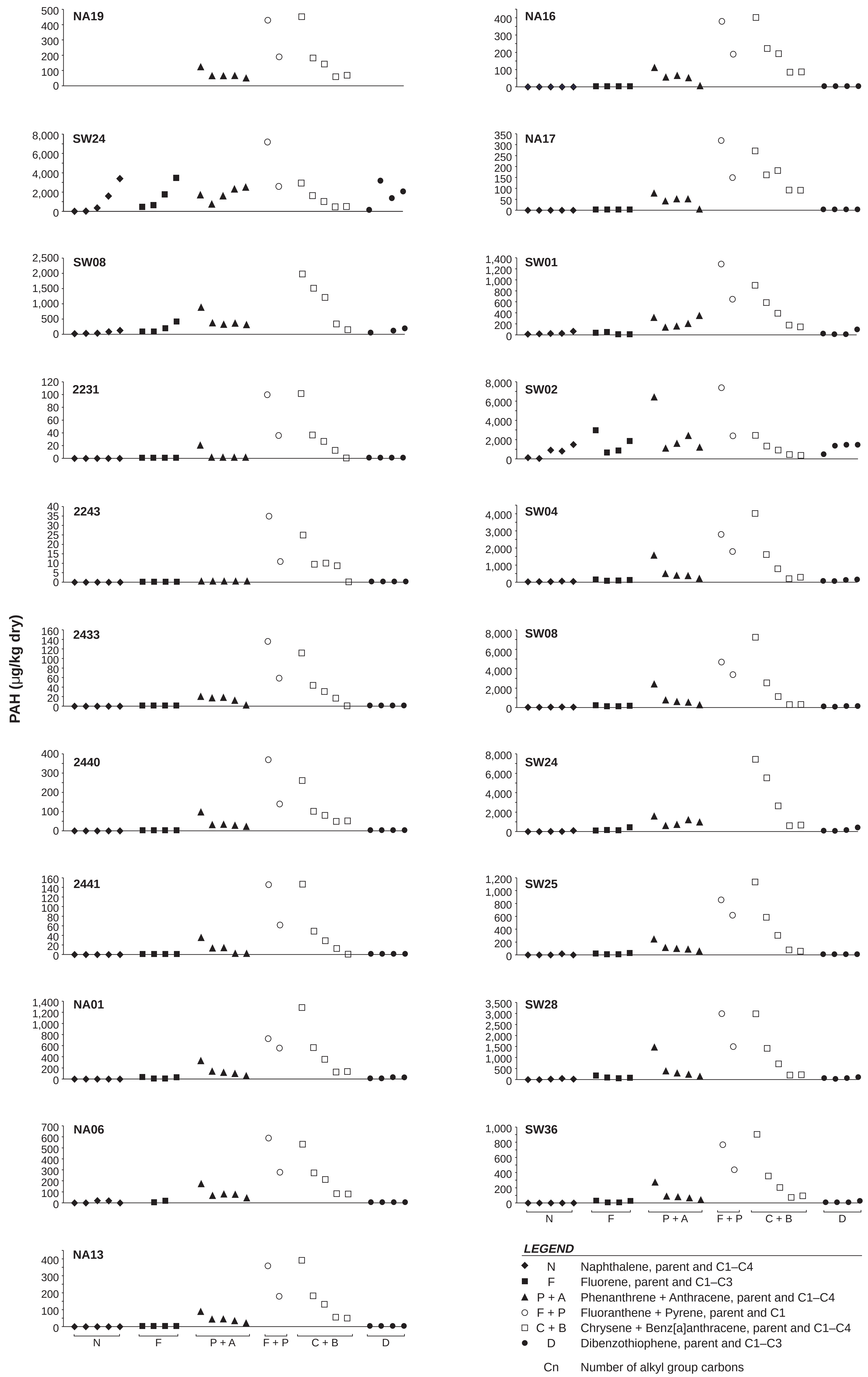


Figure 4-22. Concentrations of alkylated and non-alkylated PAH

8601718.002 1201 - 8601731.002 1201 10/08/03 WA

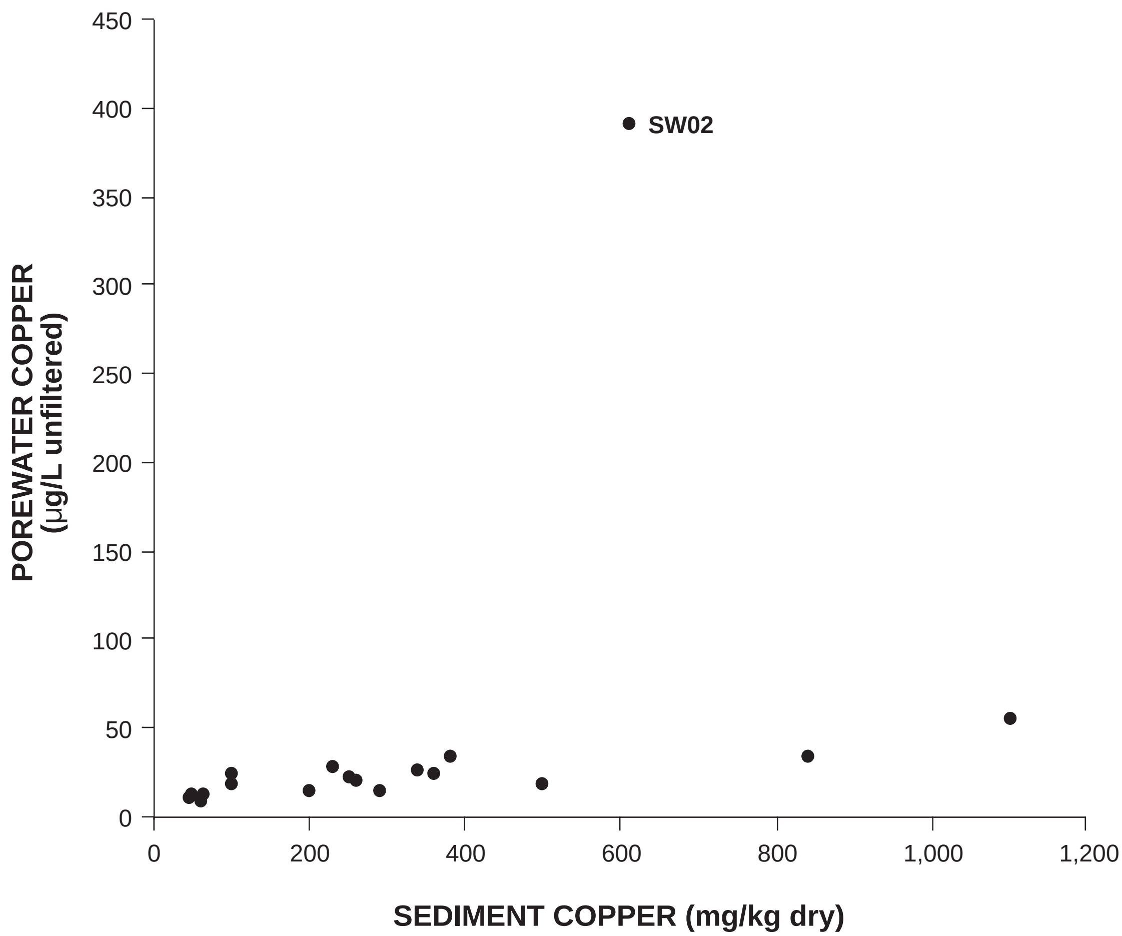


Figure 4-23. Copper concentrations in sediment and associated pore water

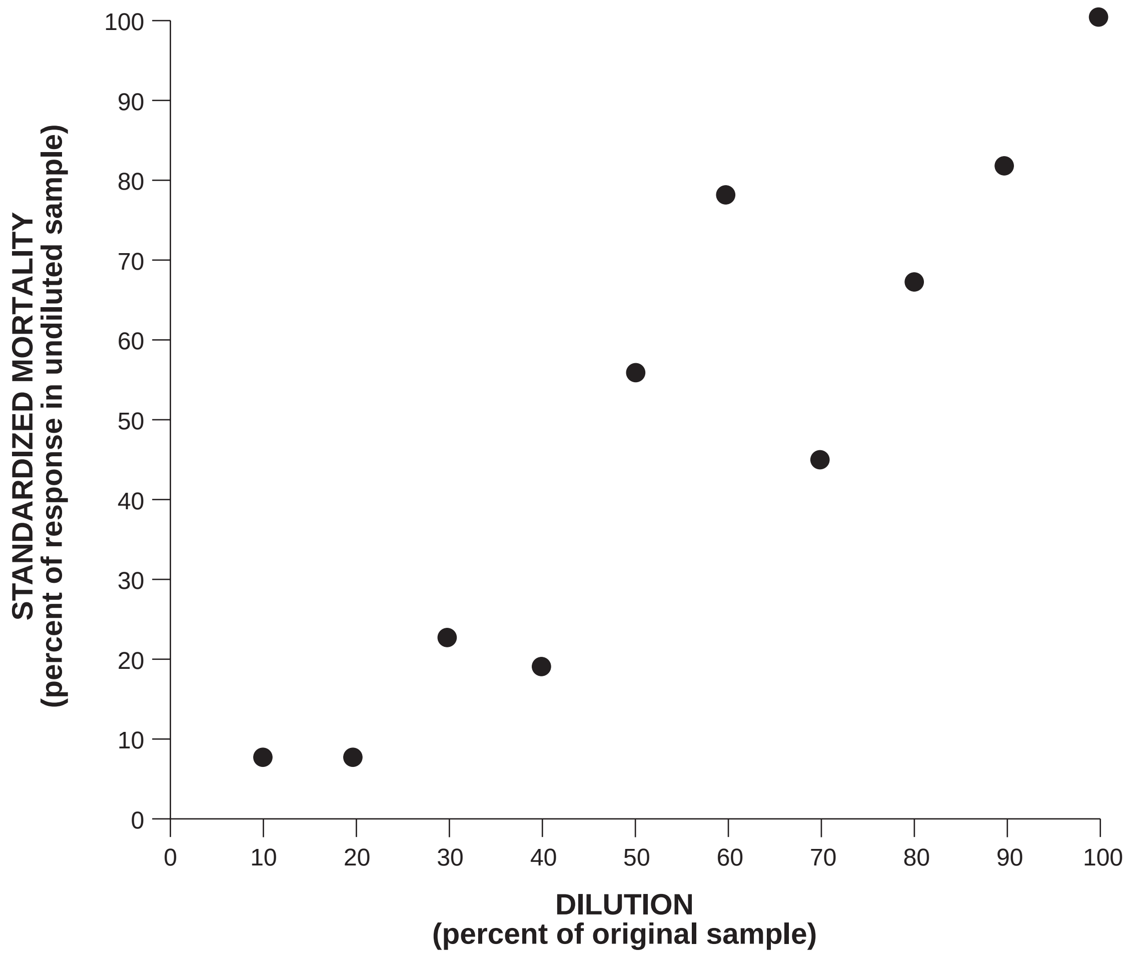


Figure 6-1. Amphipod serial dilution response, Station NA07

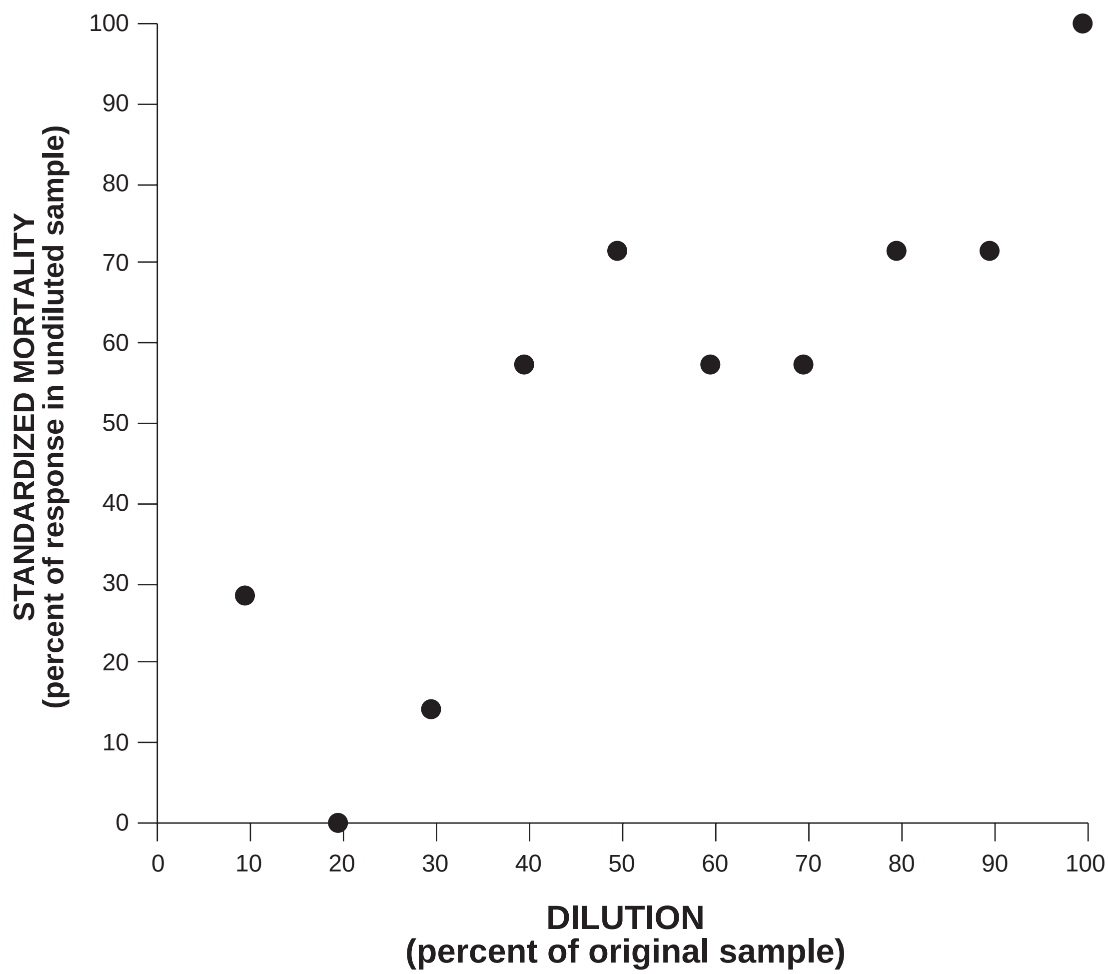


Figure 6-2. Amphipod serial dilution response, Station SW04

8601718.002 1201 - 8601731.002 1201 10/08/03 WA

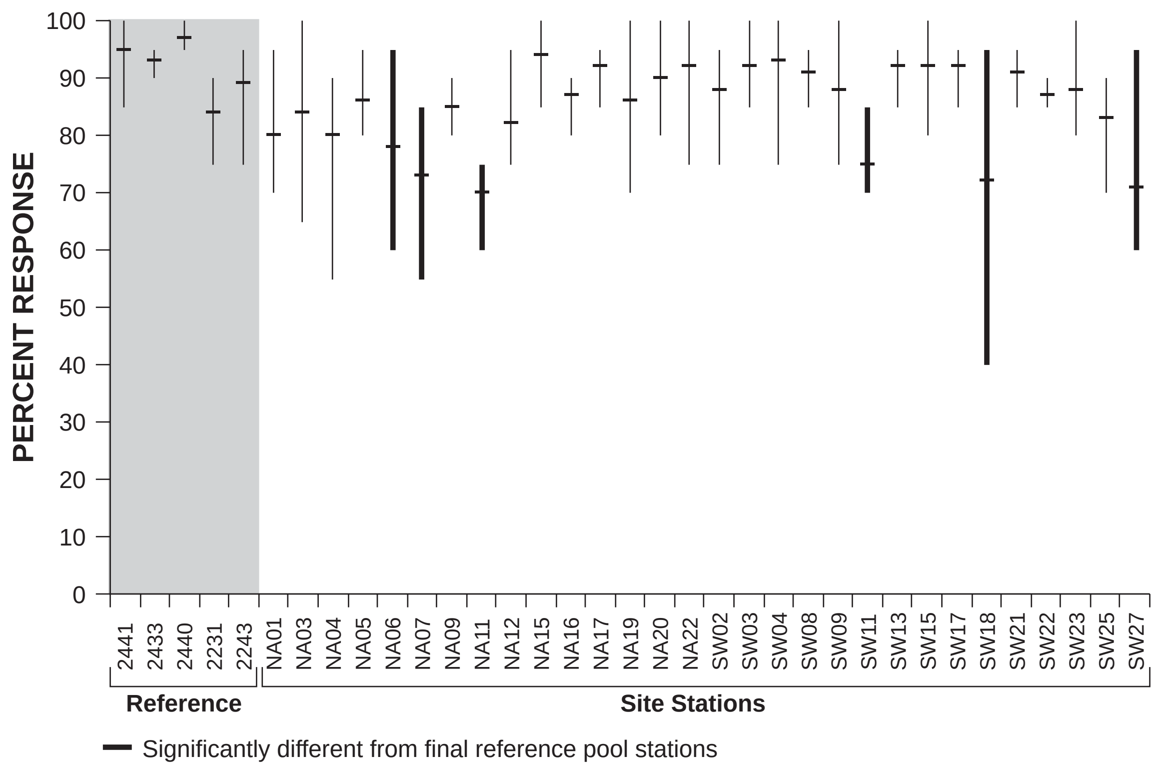


Figure 6-3. Amphipod survival response by station

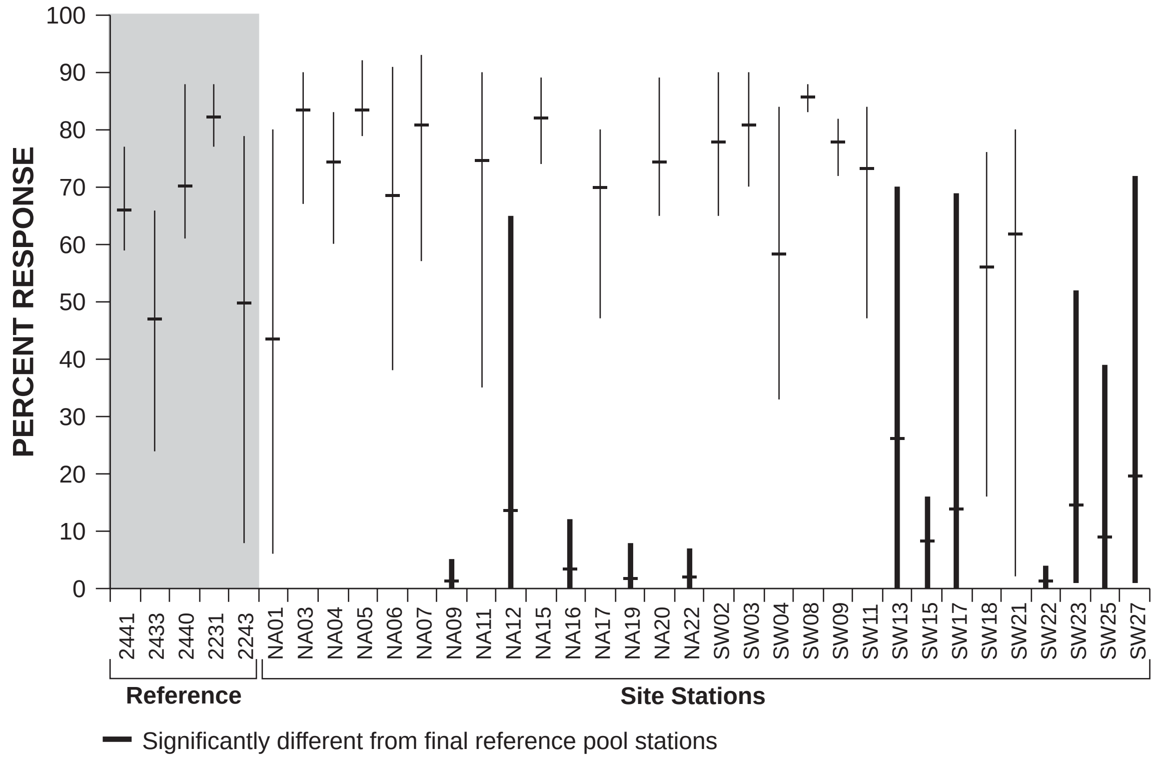


Figure 6-4. Bivalve normality response by station

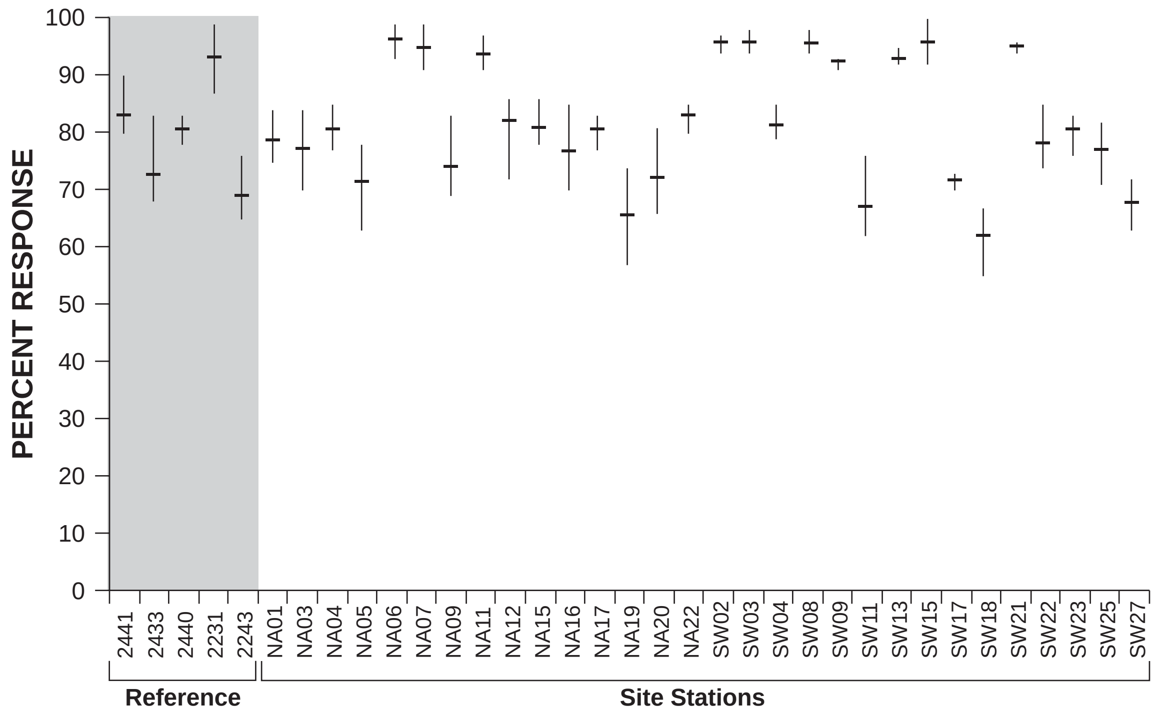


Figure 6-5. Echinoderm fertility response by station

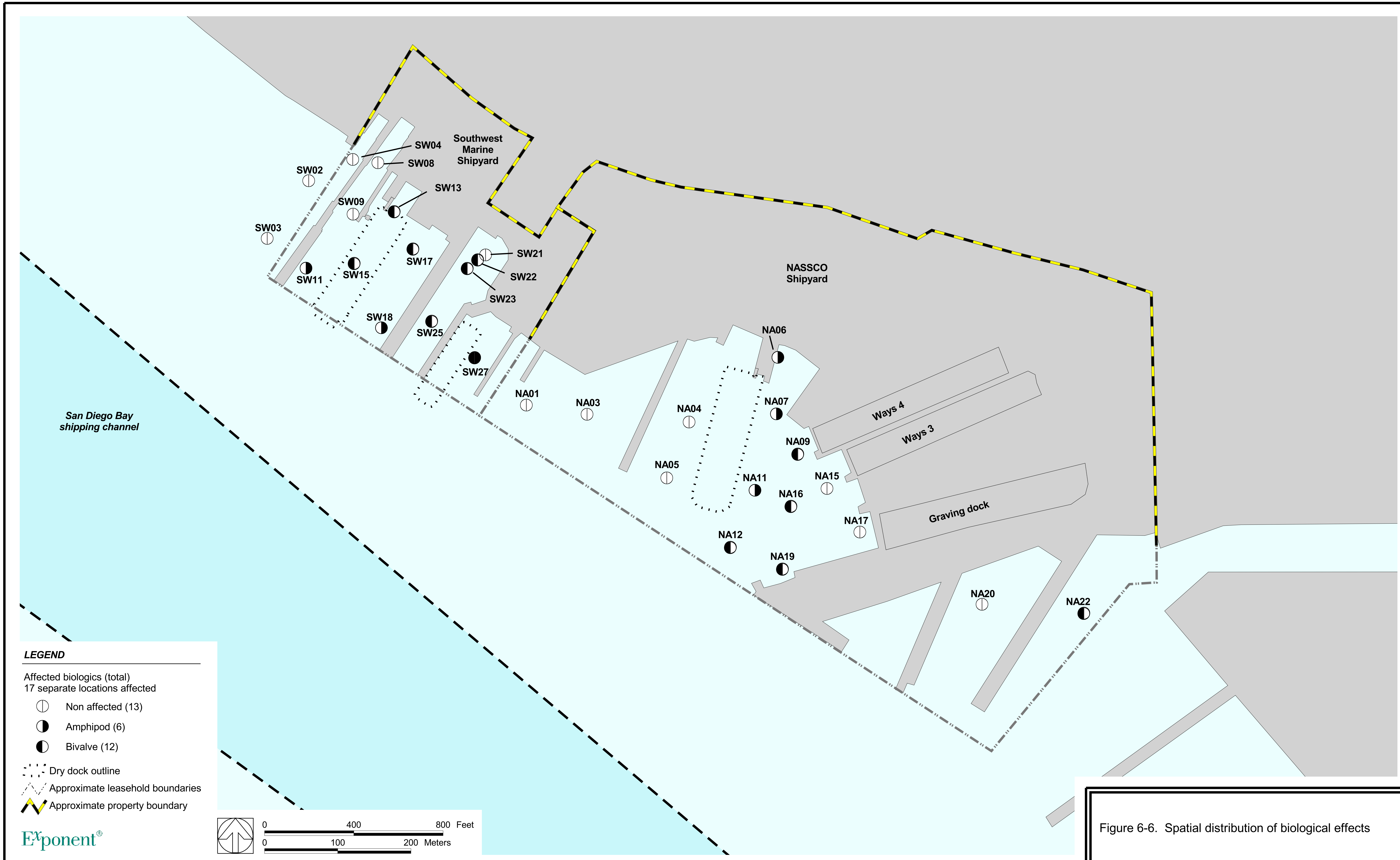


Figure 6-6. Spatial distribution of biological effects

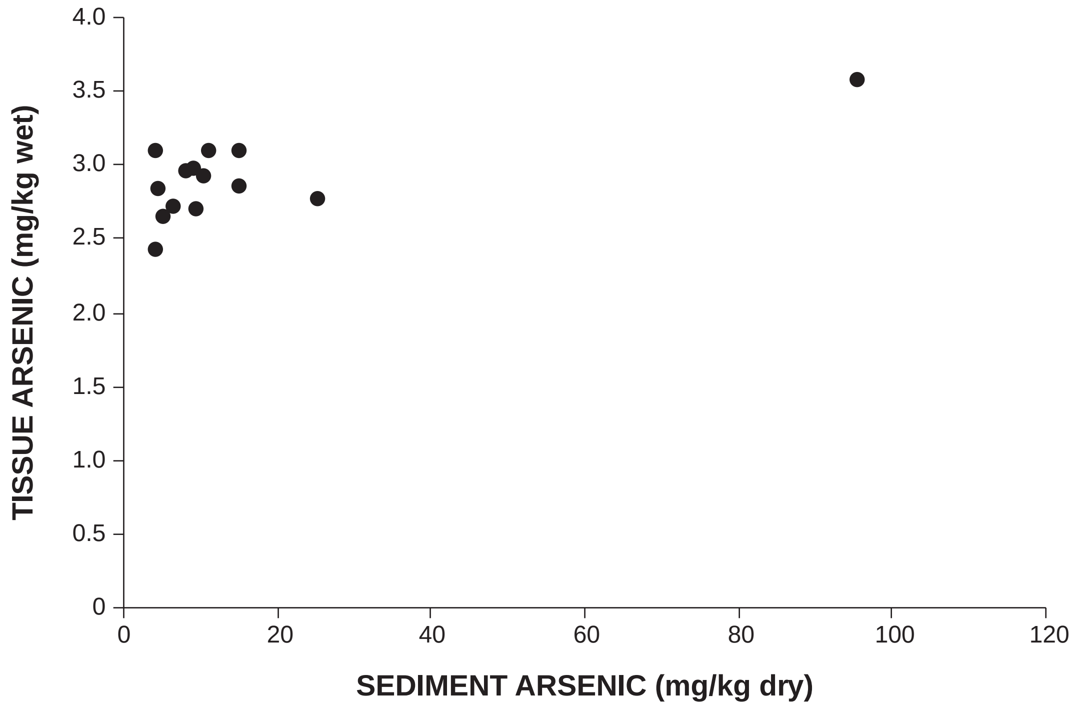


Figure 7-1. Tissue and sediment data for arsenic

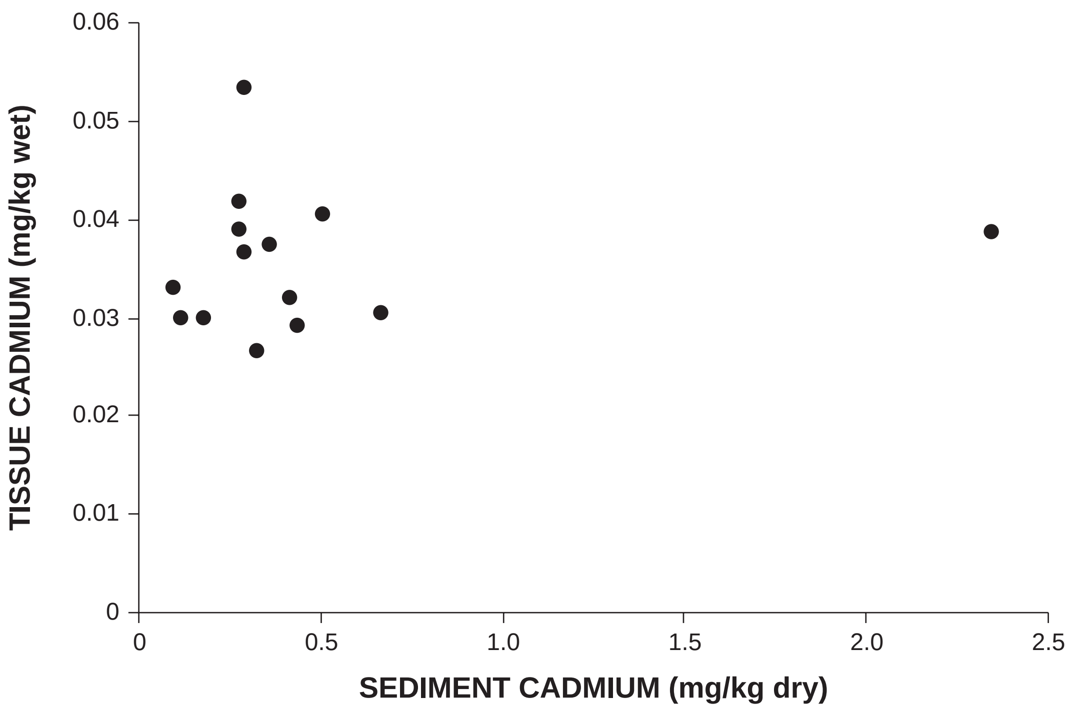


Figure 7-2. Tissue and sediment data for cadmium

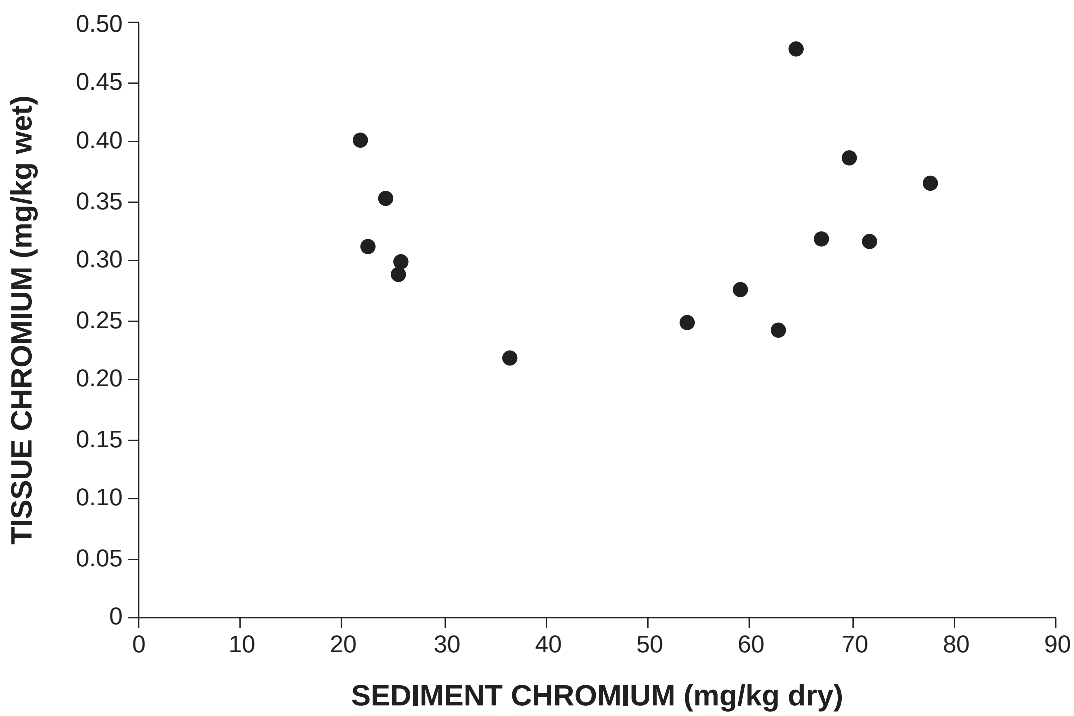


Figure 7-3. Tissue and sediment data for chromium

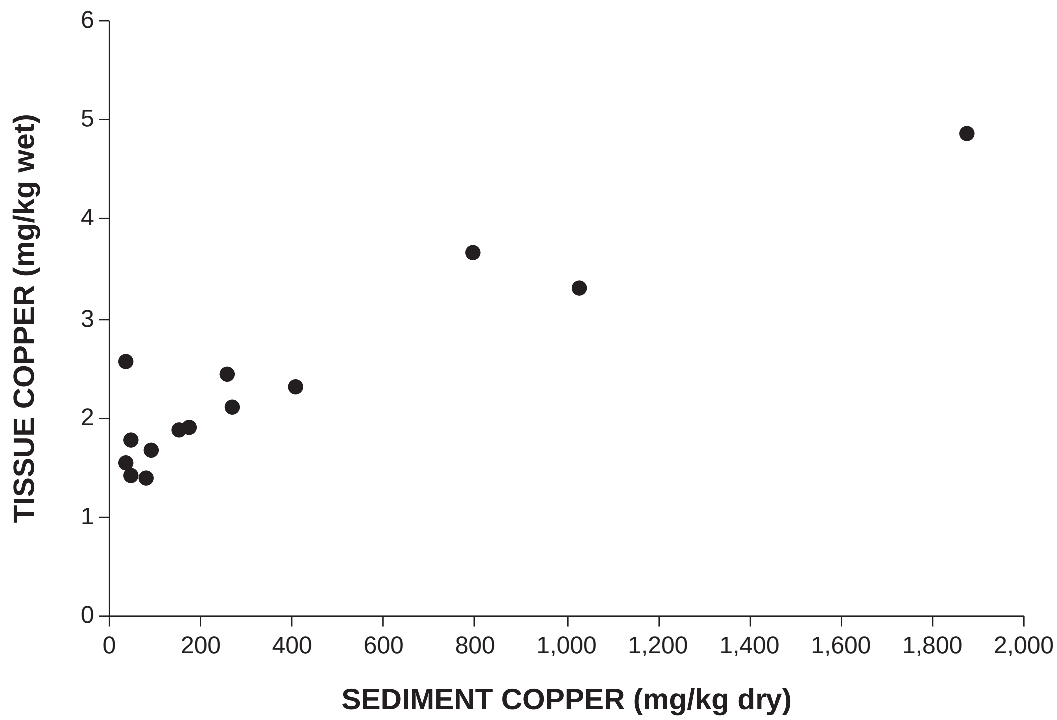


Figure 7-4. Tissue and sediment data for copper

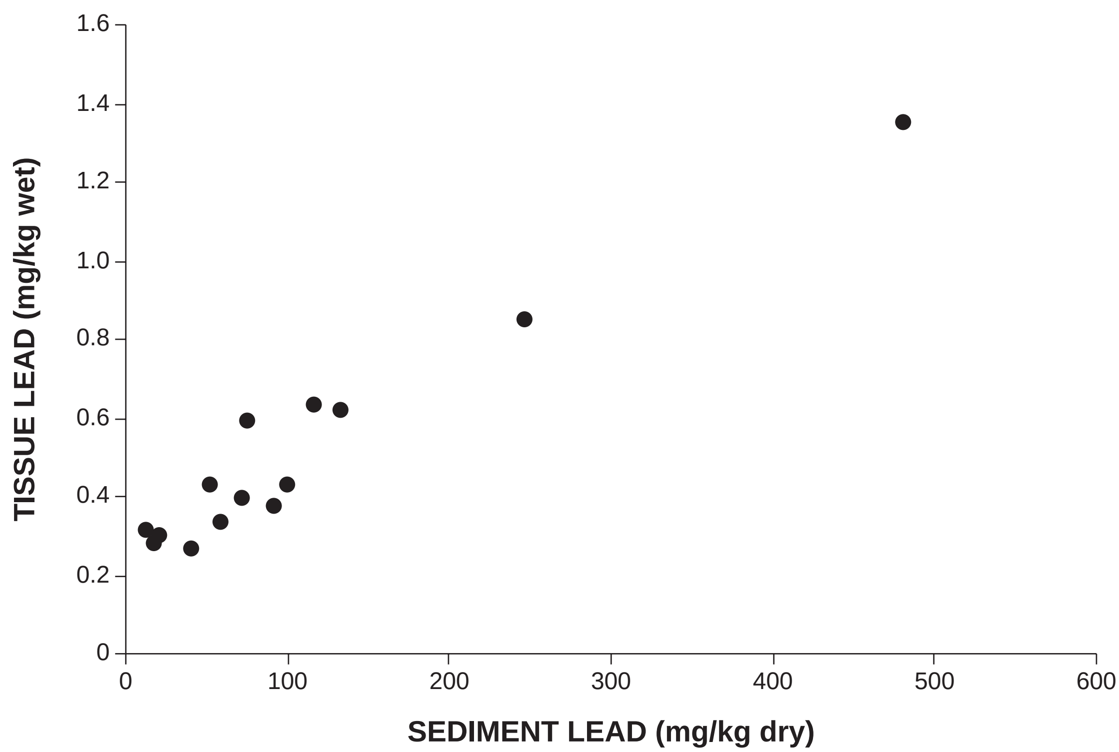


Figure 7-5. Tissue and sediment data for lead

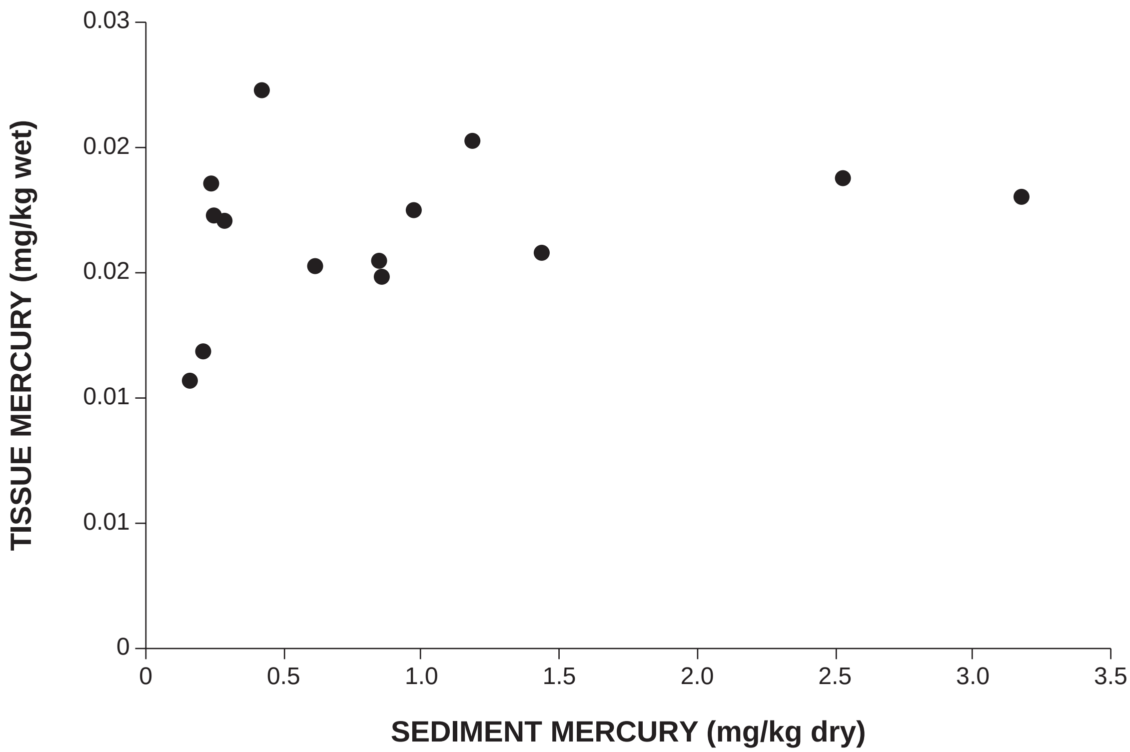


Figure 7-6. Tissue and sediment data for mercury

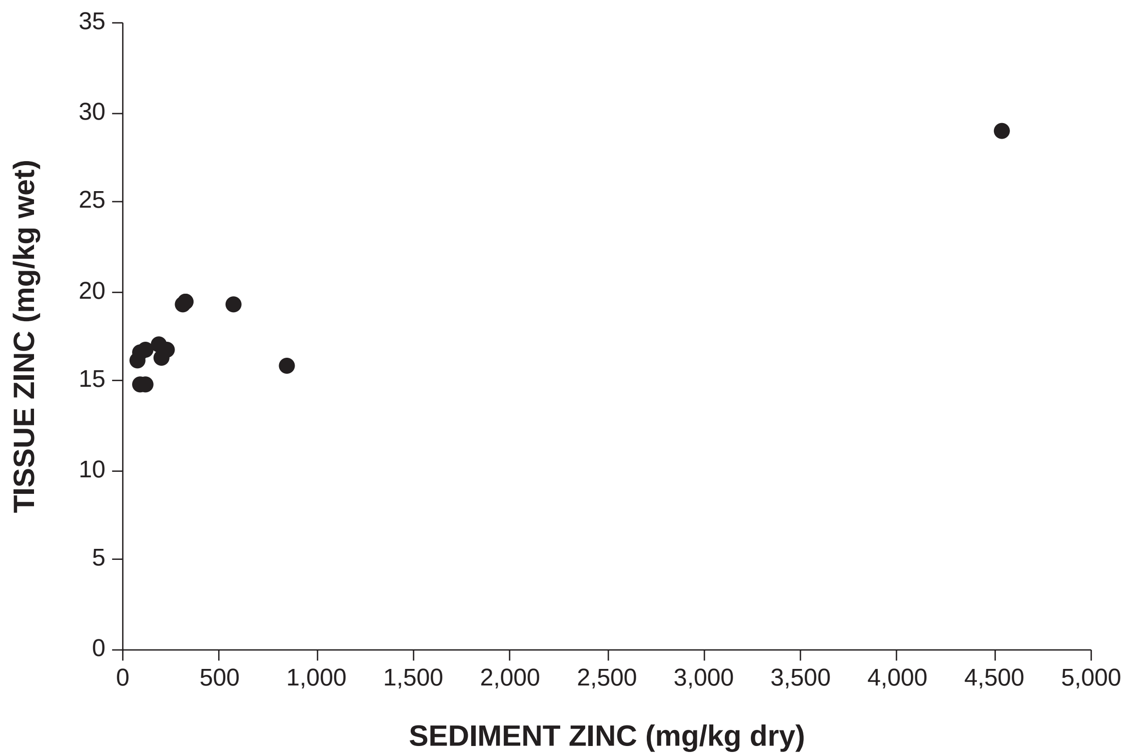


Figure 7-7. Tissue and sediment data for zinc

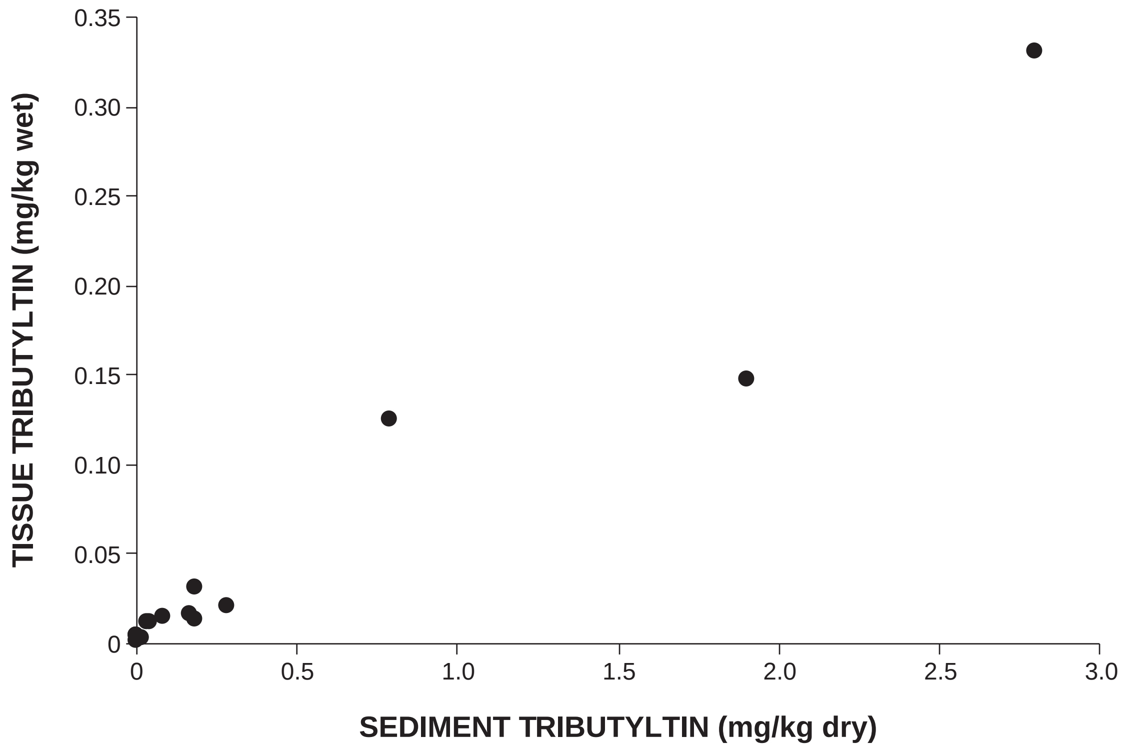


Figure 7-8. Tissue and sediment data for tributyltin

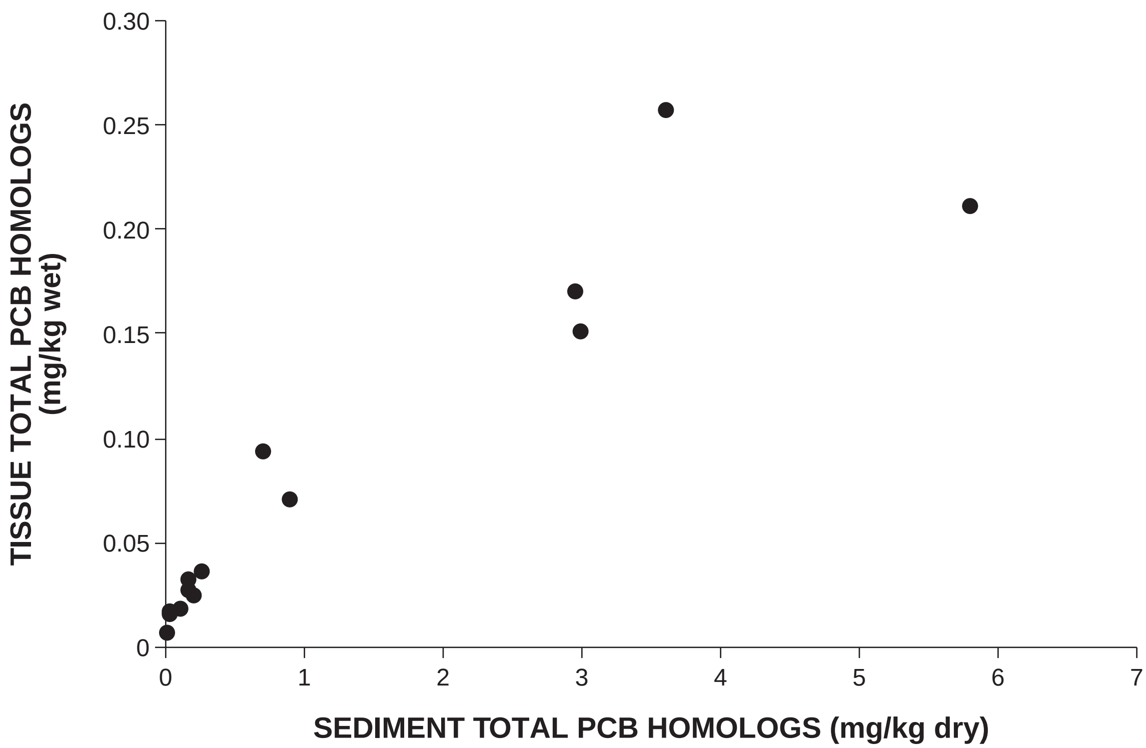


Figure 7-9. Tissue and sediment data for total PCB homologs

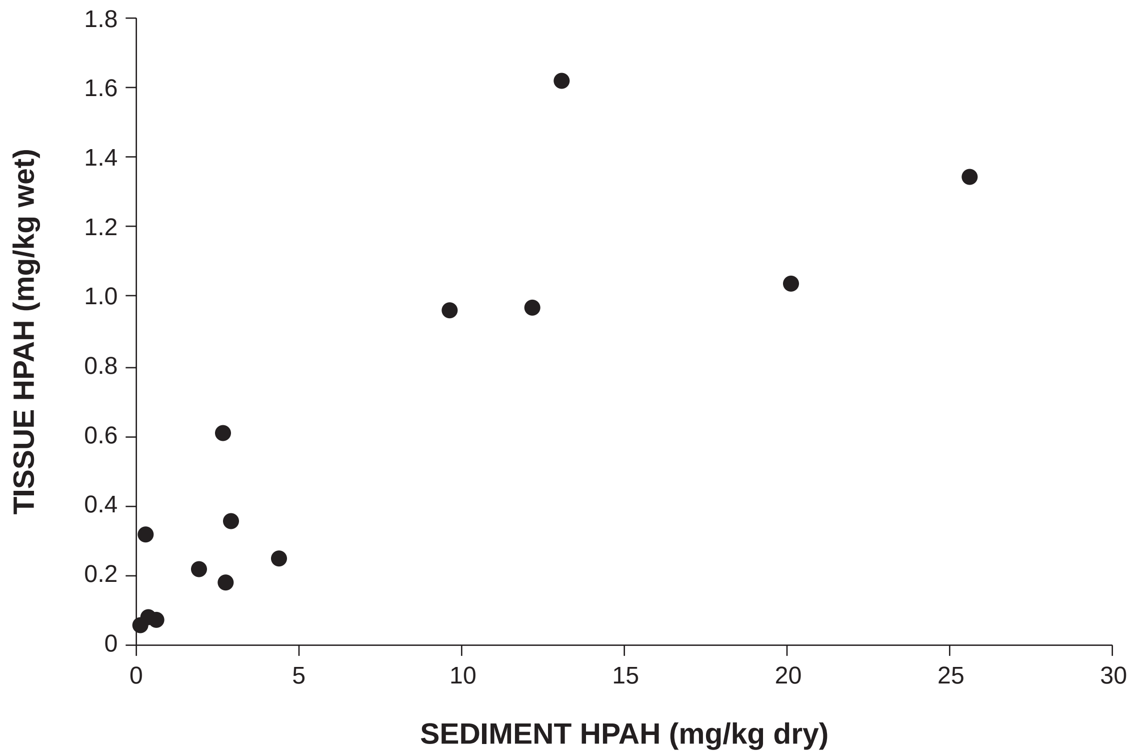


Figure 7-10. Tissue and sediment data for high-molecular-weight polycyclic aromatic hydrocarbons (HPAHs)

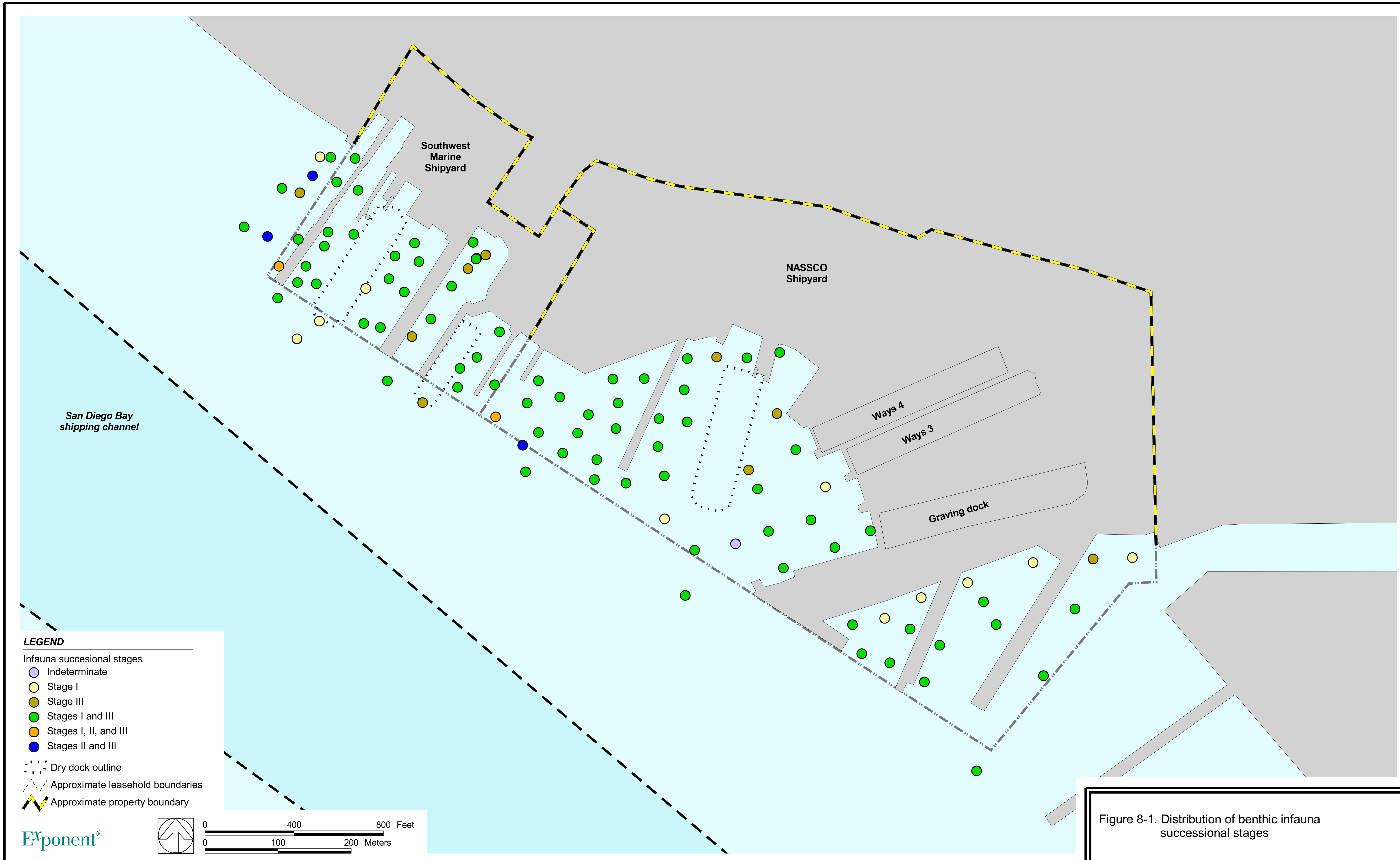


Figure 8-1. Distribution of benthic infauna successional stages

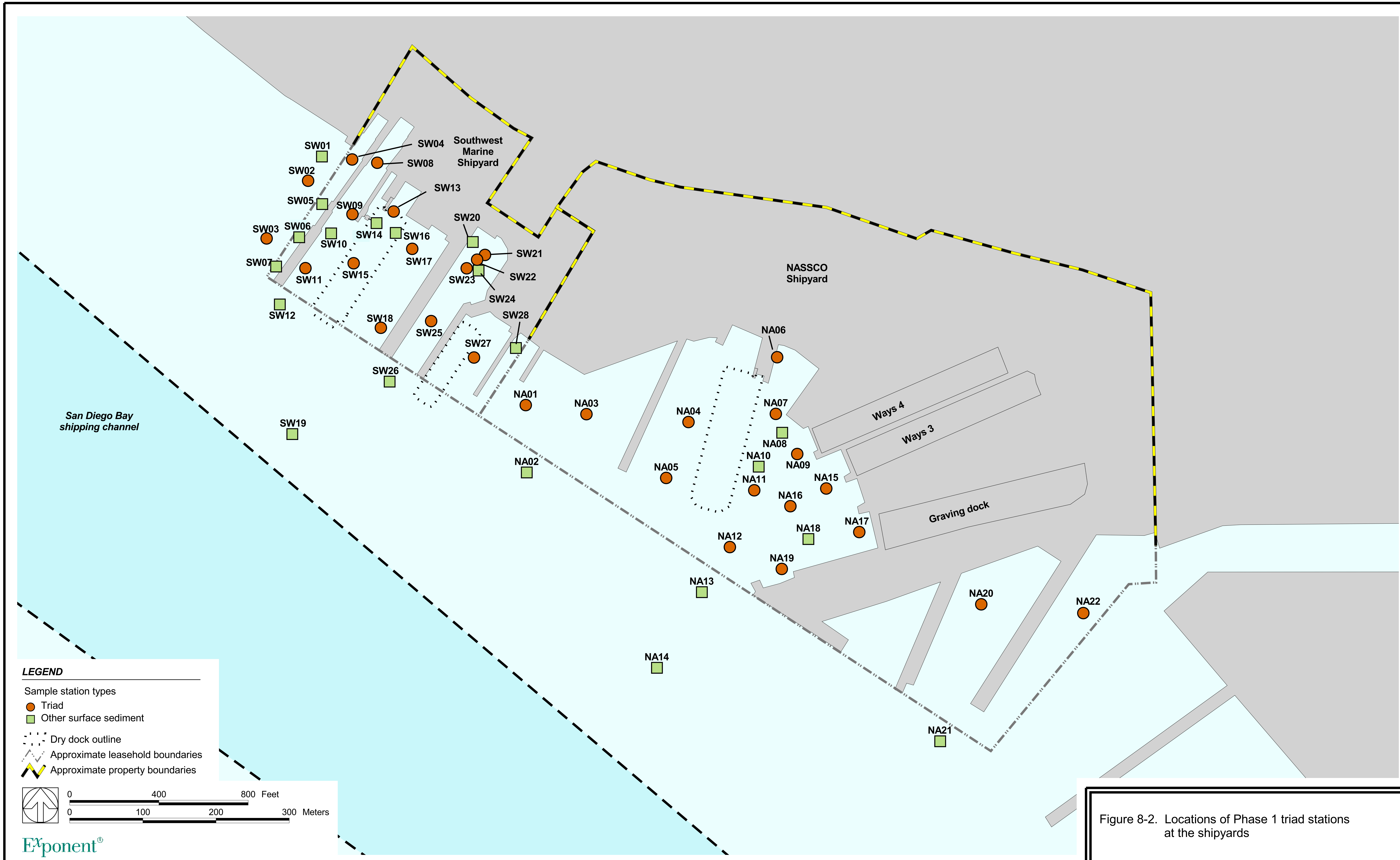
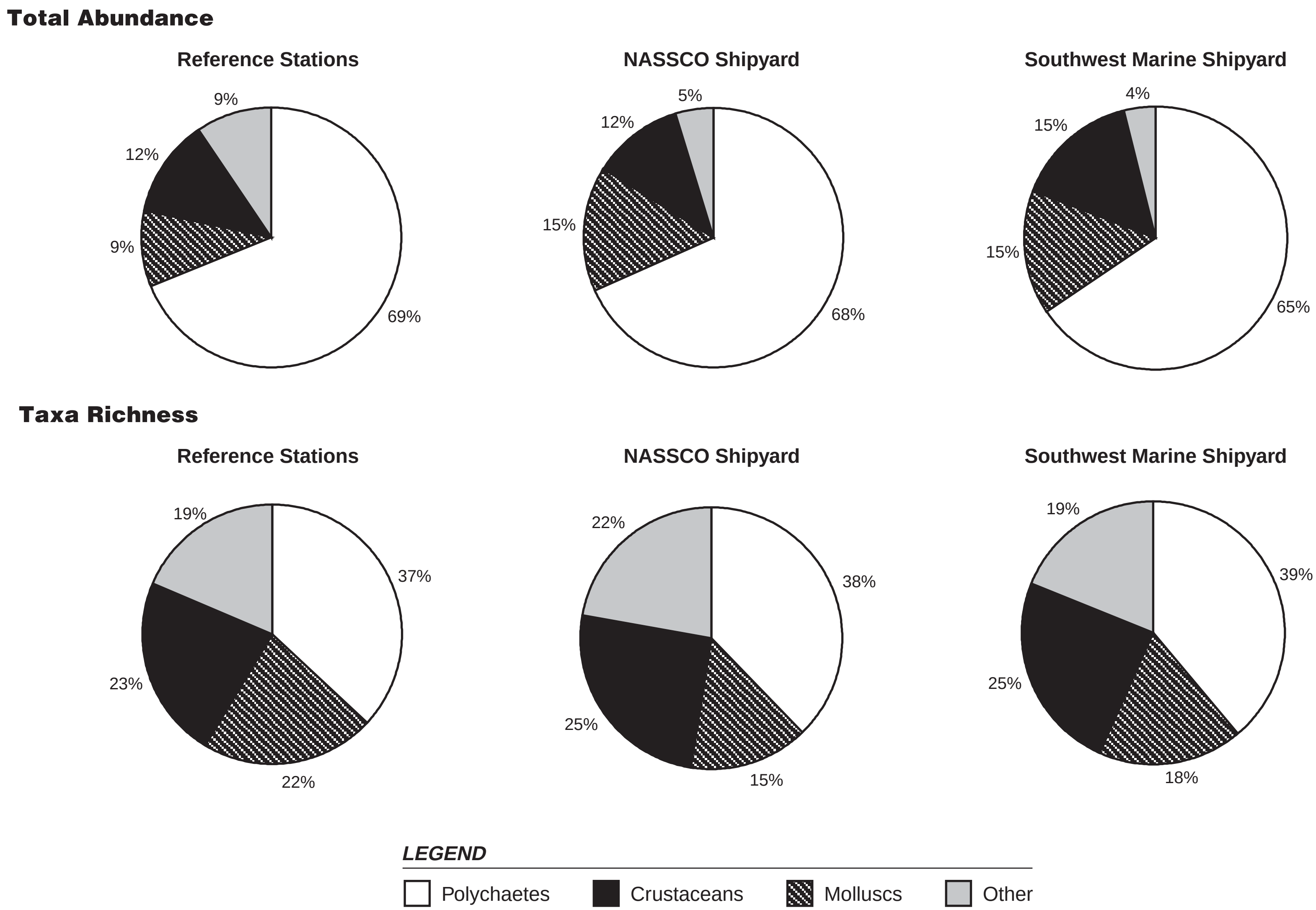


Figure 8-2. Locations of Phase 1 triad stations at the shipyards

8601718/8601731.002 1201 | Sep 22, 2003 | fig 08-2 Triad sample view | Fig 08-2 TRIAD locations layout | g:/swm_nassco/projects/shipyards_rpt.apr



Note: Data were pooled for all stations sampled in each area.

Figure 8-3. Comparison of percent community composition represented by major benthic taxa at the shipyard sites and reference areas

8601718.002 1201 - 8601731.002 1201 10/08/03 WA

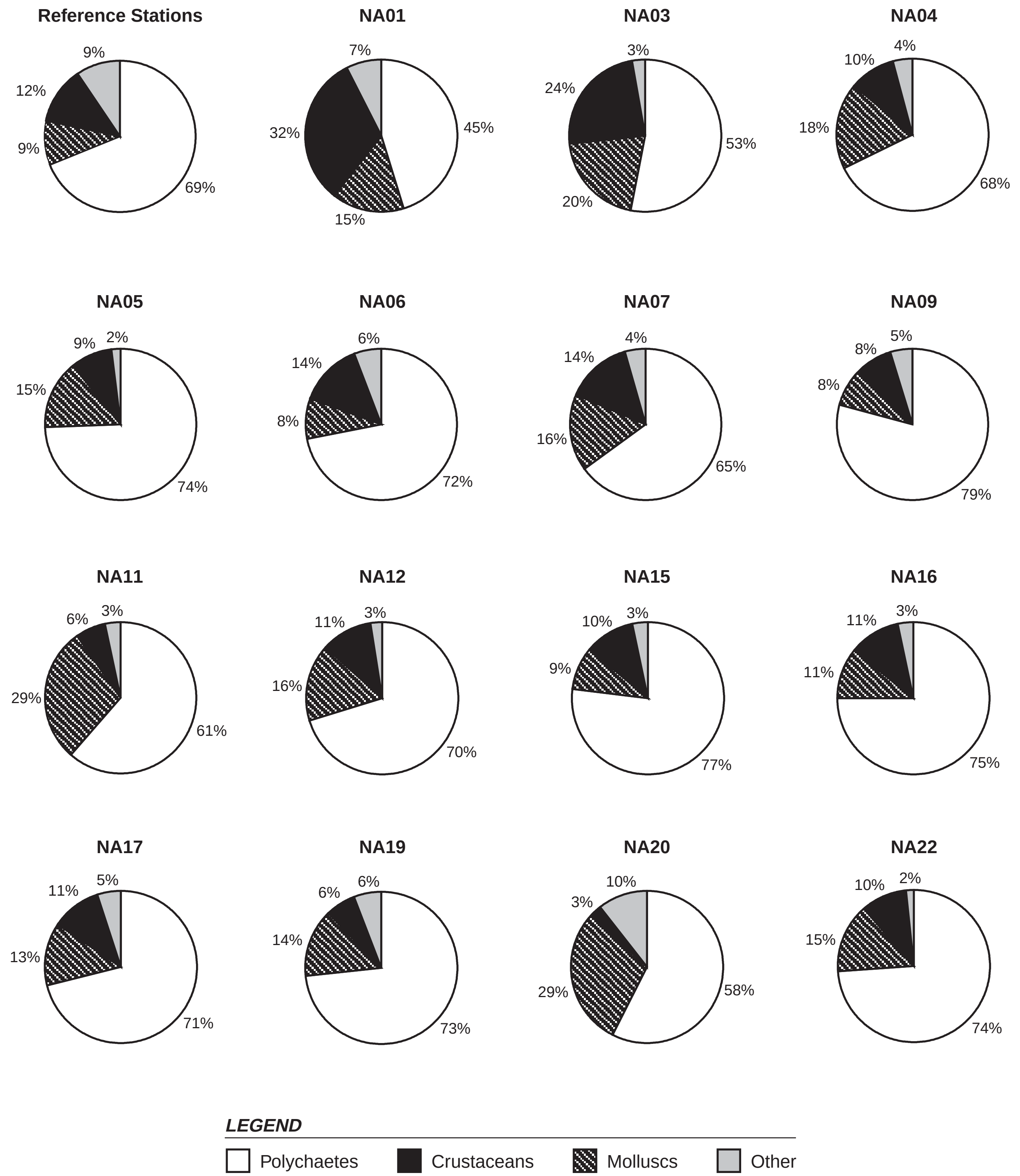


Figure 8-4. Comparison of relative abundances of major benthic taxa at stations in the NASSCO shipyard site and the reference stations

8601718.002 1201 - 8601731.002 1201 10/08/03 WA

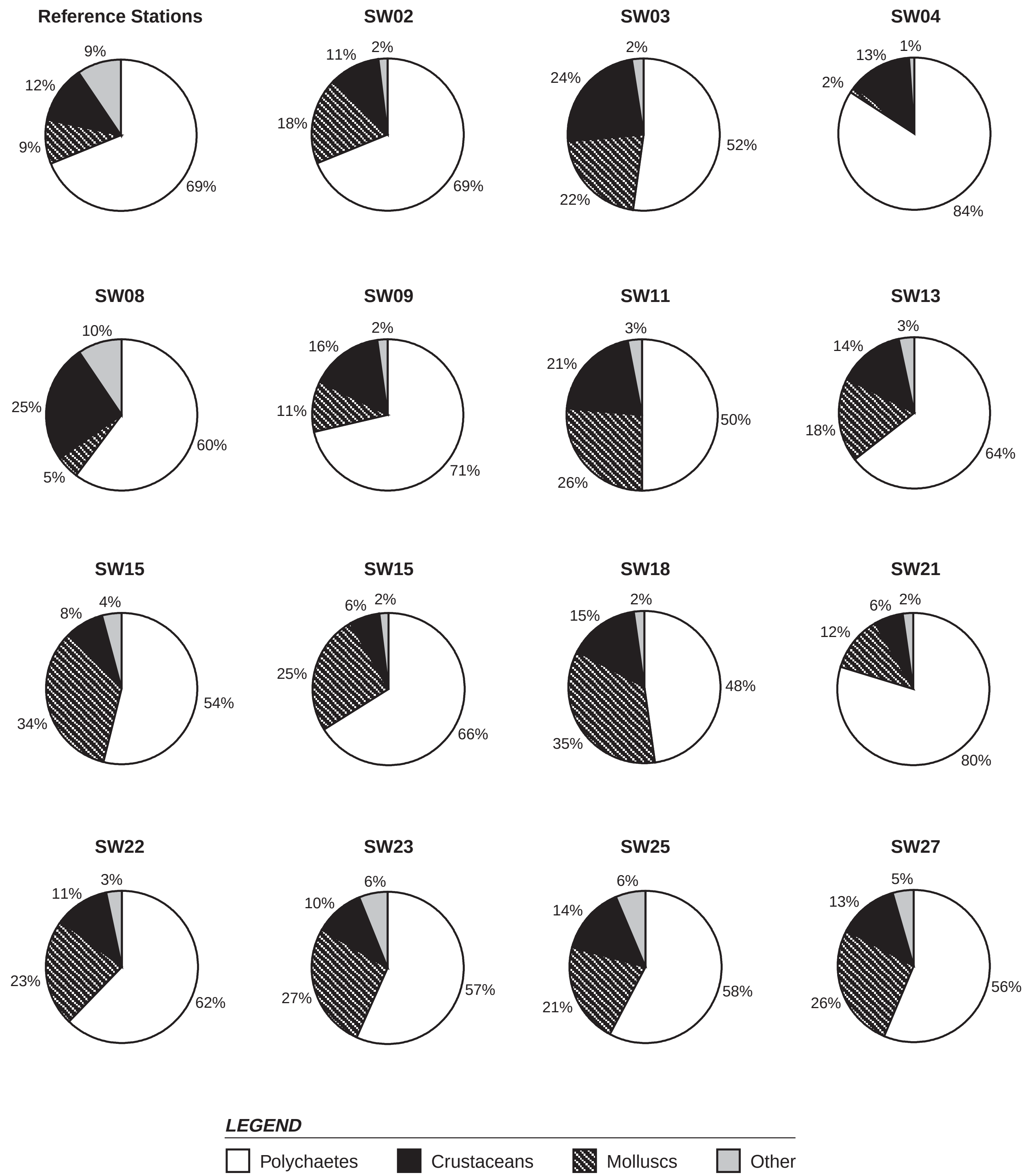


Figure 8-5. Comparison of relative abundances of major benthic taxa at stations in the Southwest Marine shipyard site and the reference stations

8601718.002 1201 - 8601731.002 1201 10/08/03 WA

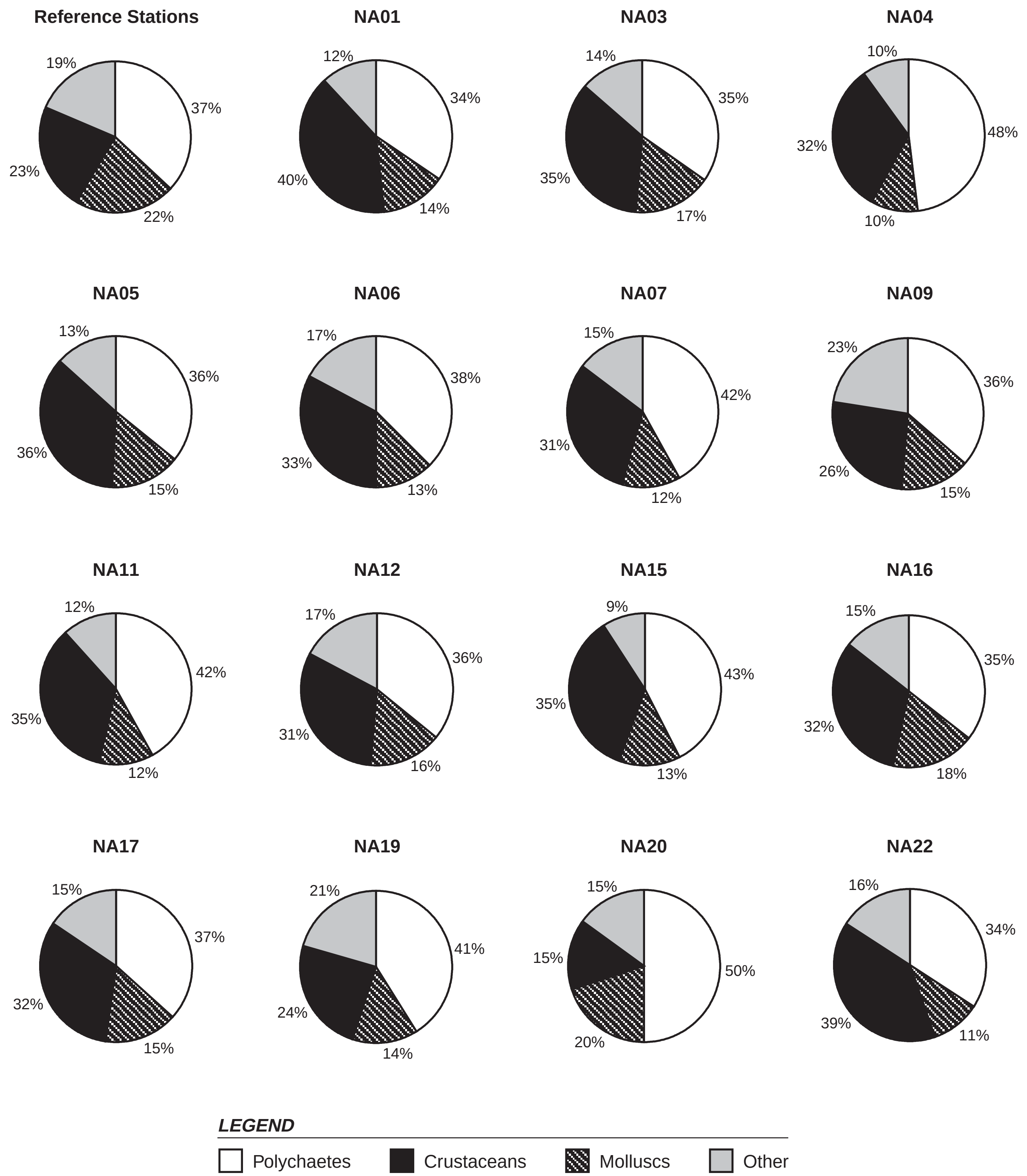


Figure 8-6. Comparison of relative taxa richness at stations in the NASSCO shipyard site and the reference stations

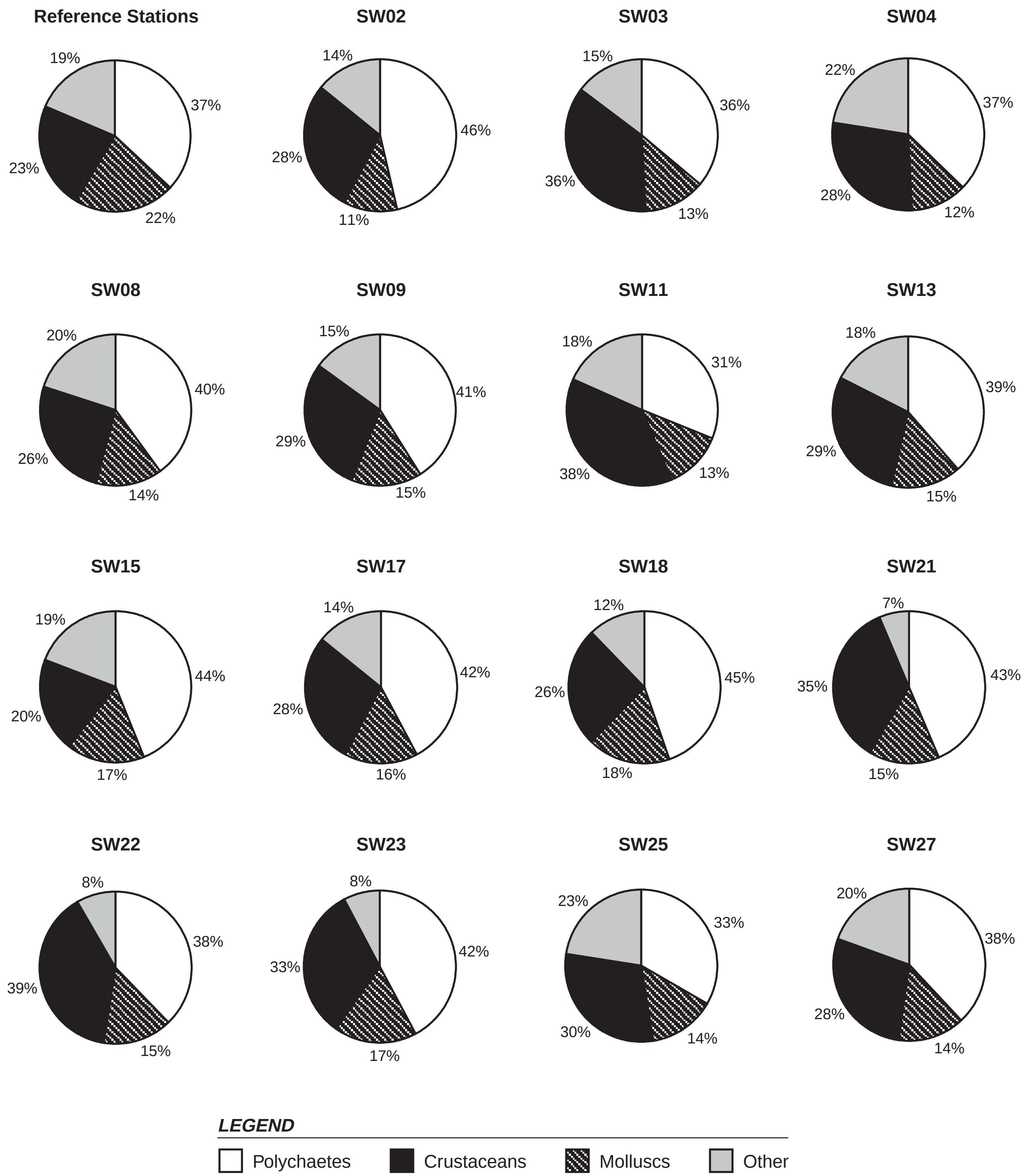


Figure 8-7. Comparison of relative taxa richness at stations in the Southwest Marine shipyard site and the reference stations

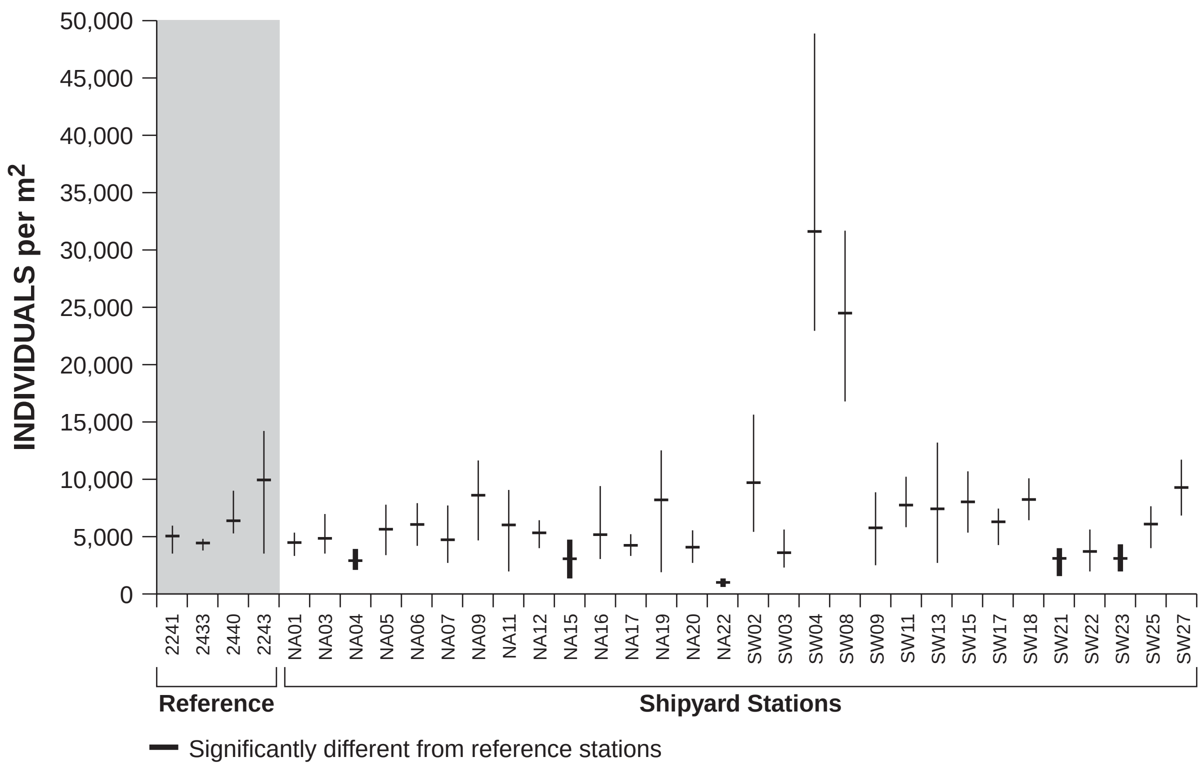


Figure 8-8. Comparison of total abundance of benthic communities among shipyard and reference stations

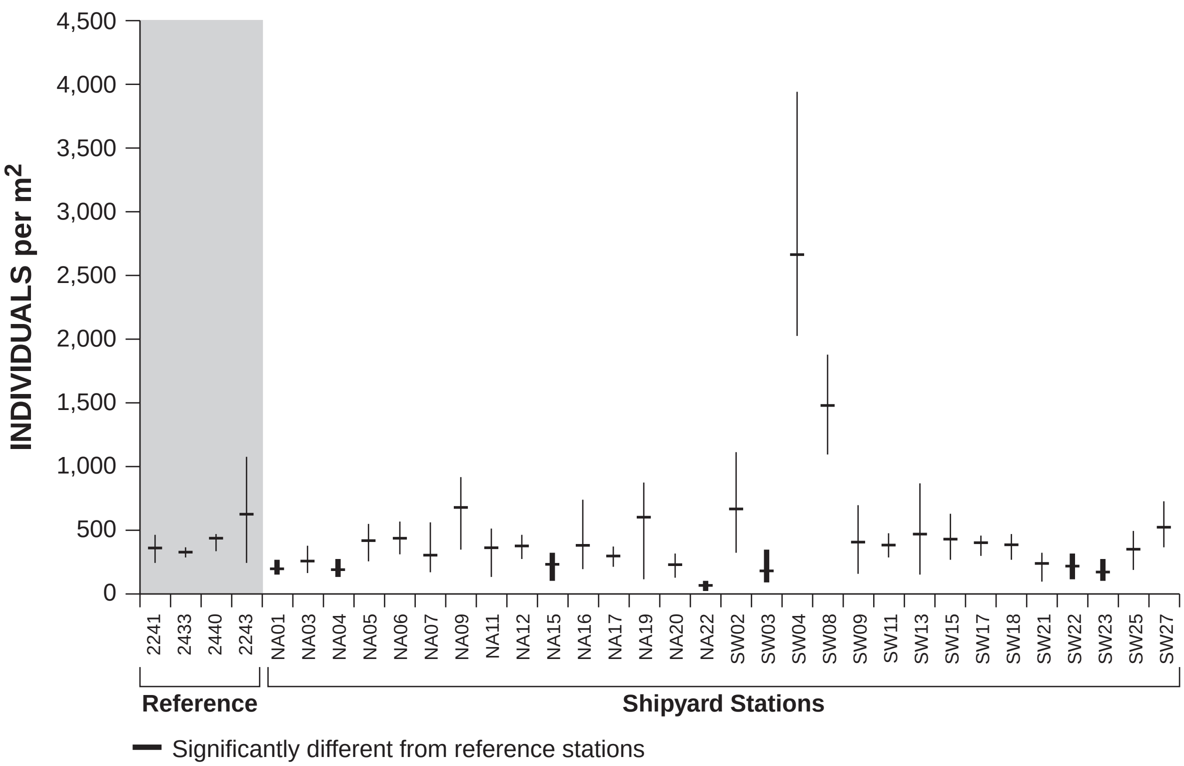


Figure 8-9. Comparison of polychaete abundance among shipyard and reference stations

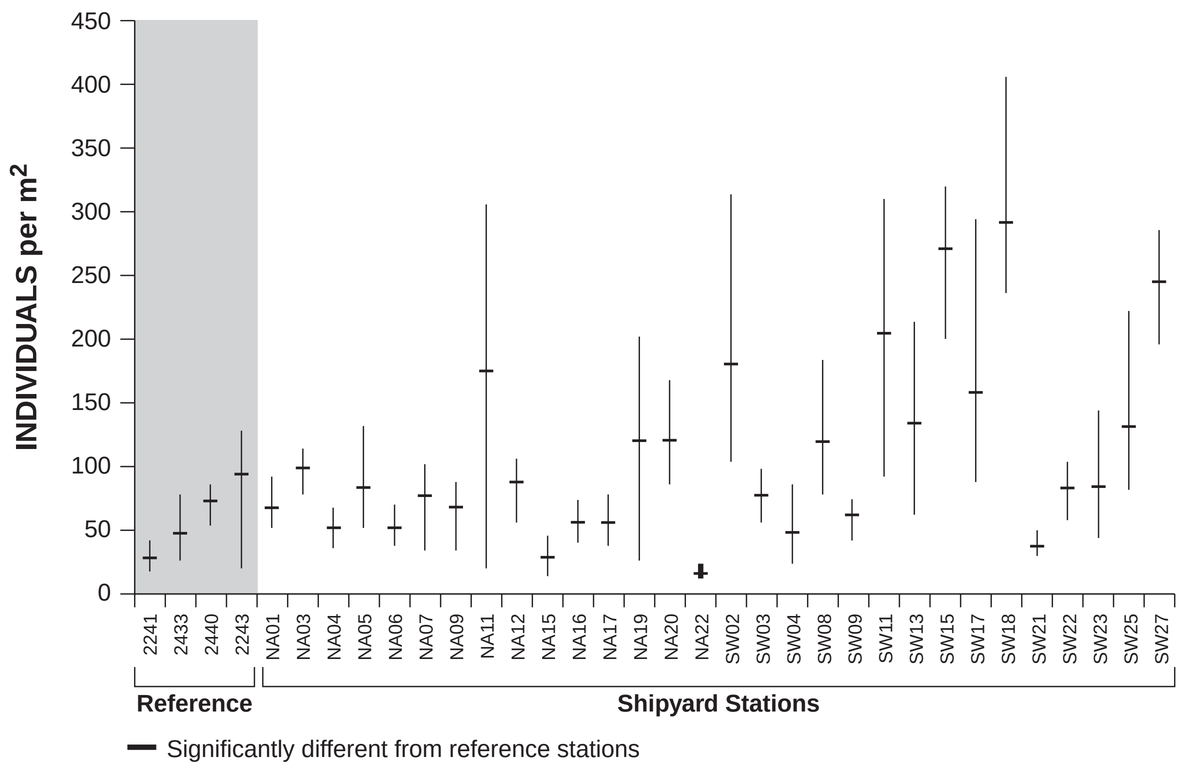


Figure 8-10. Comparison of mollusc abundance among shipyard and reference stations

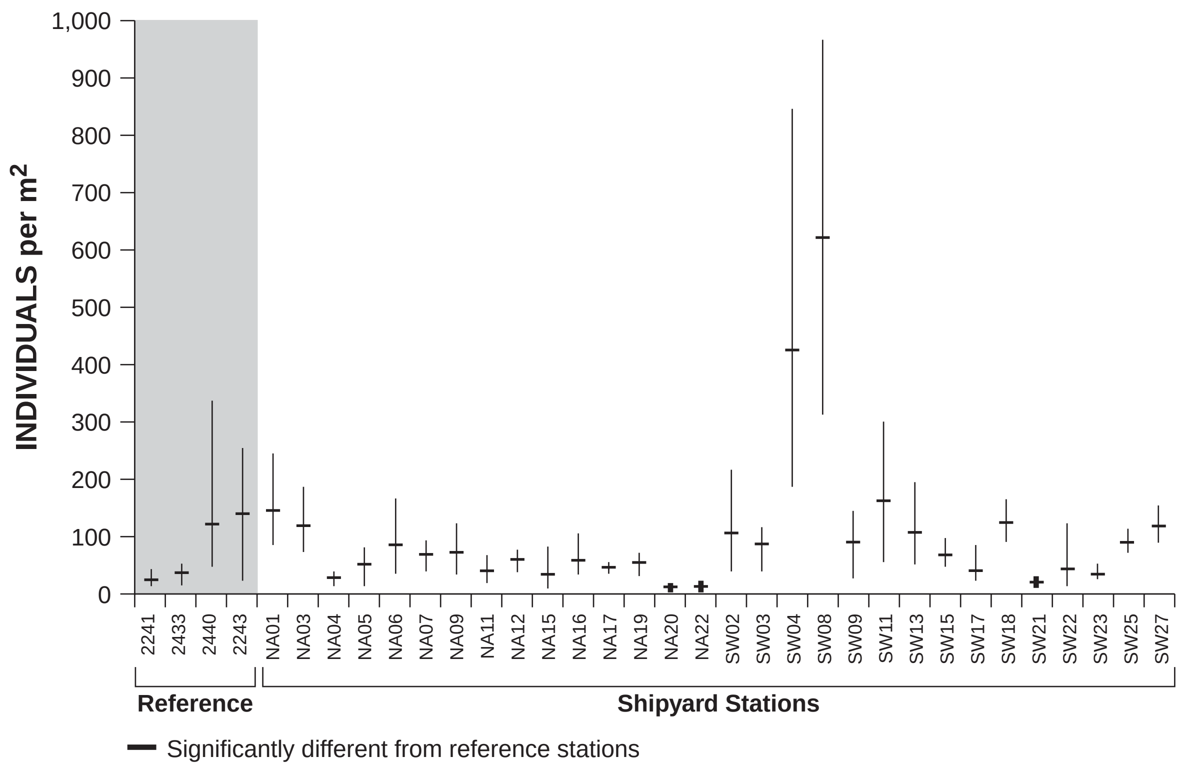


Figure 8-11. Comparison of crustacean abundance among shipyard and reference stations

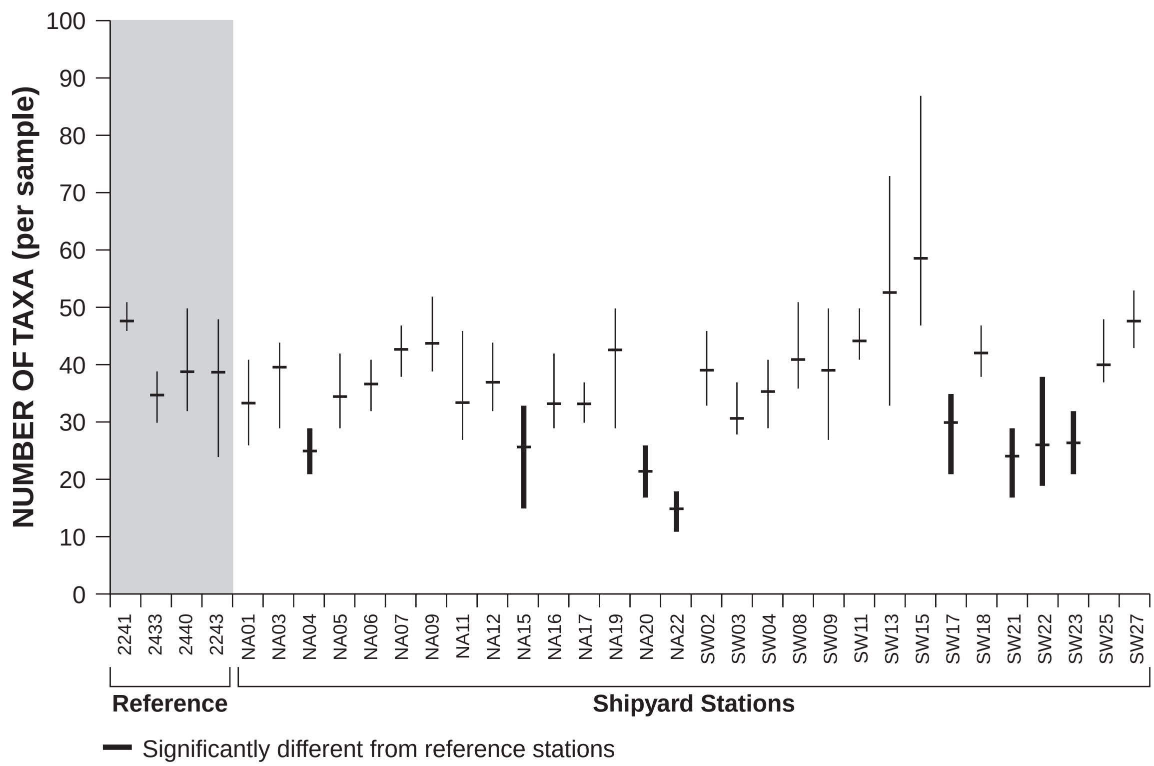


Figure 8-12. Comparison of taxa richness among shipyard and reference stations

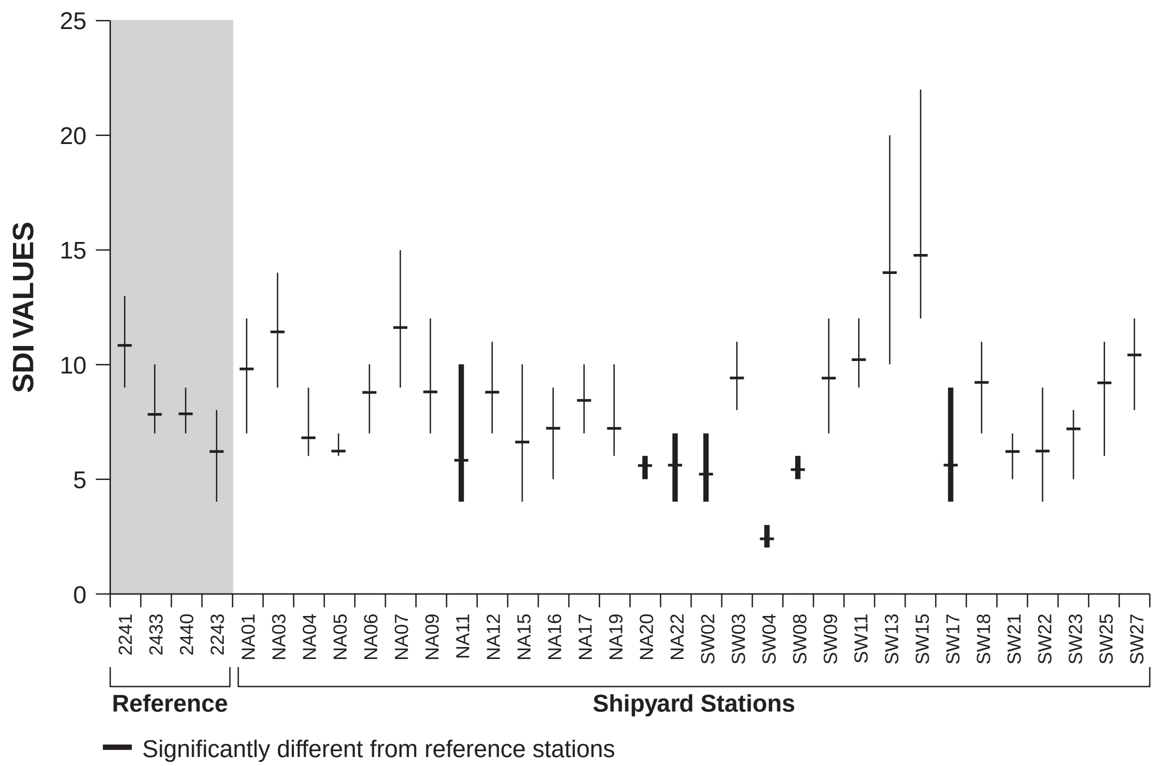


Figure 8-13. Comparison of Swartz' dominance index (SDI) among shipyard and reference stations

8601718.002 1201 - 8601731.002 1201 10/08/03 WA

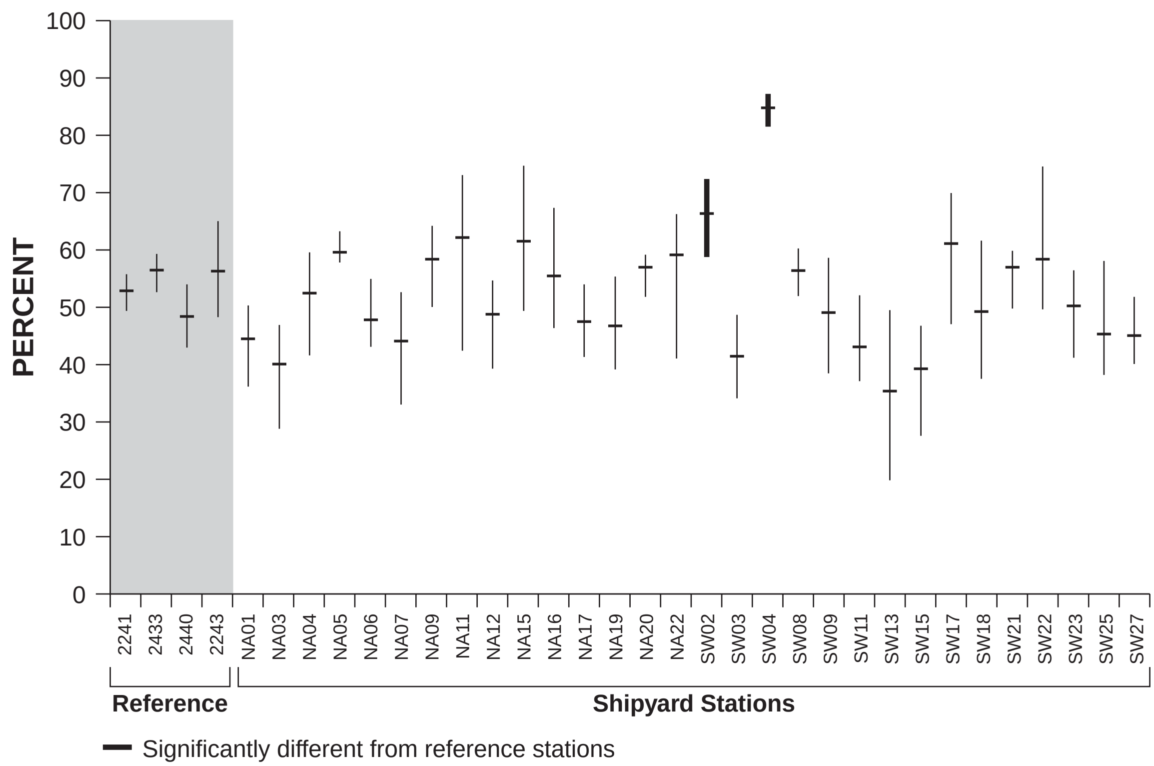


Figure 8-14. Comparison of percent dominance among shipyard and reference stations

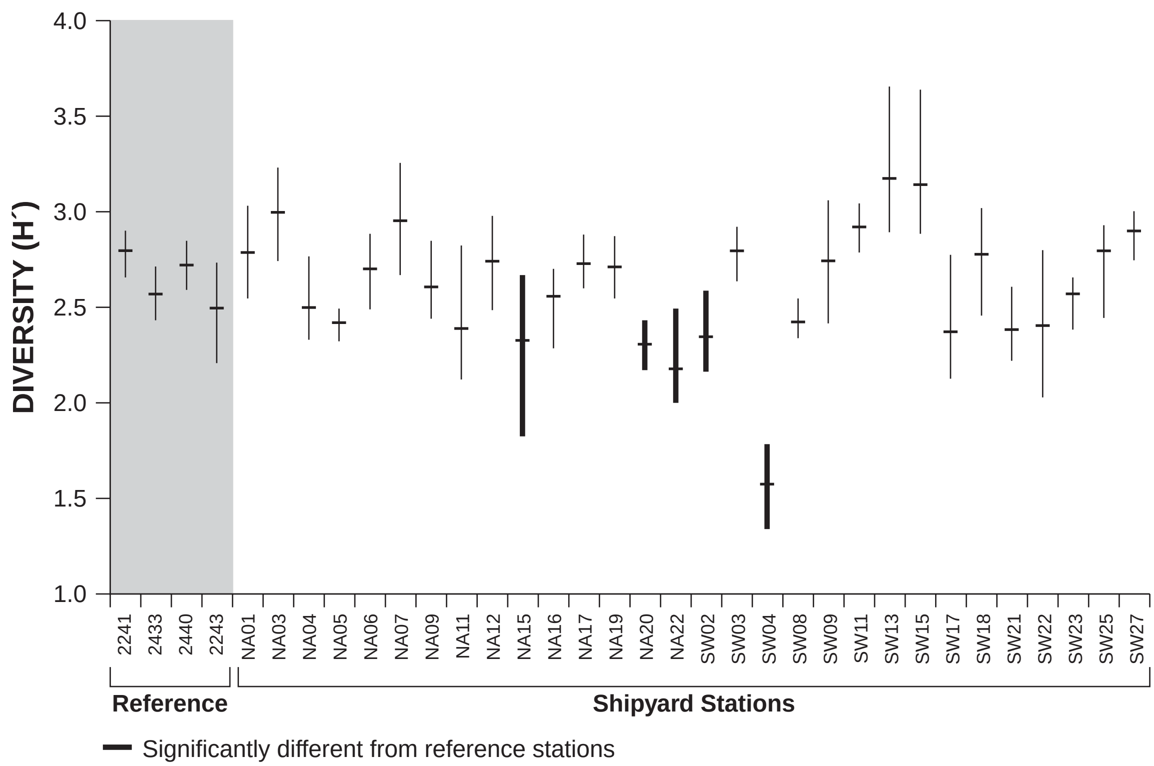


Figure 8-15. Comparison of Shannon-Wiener diversity (H´) among shipyard and reference stations

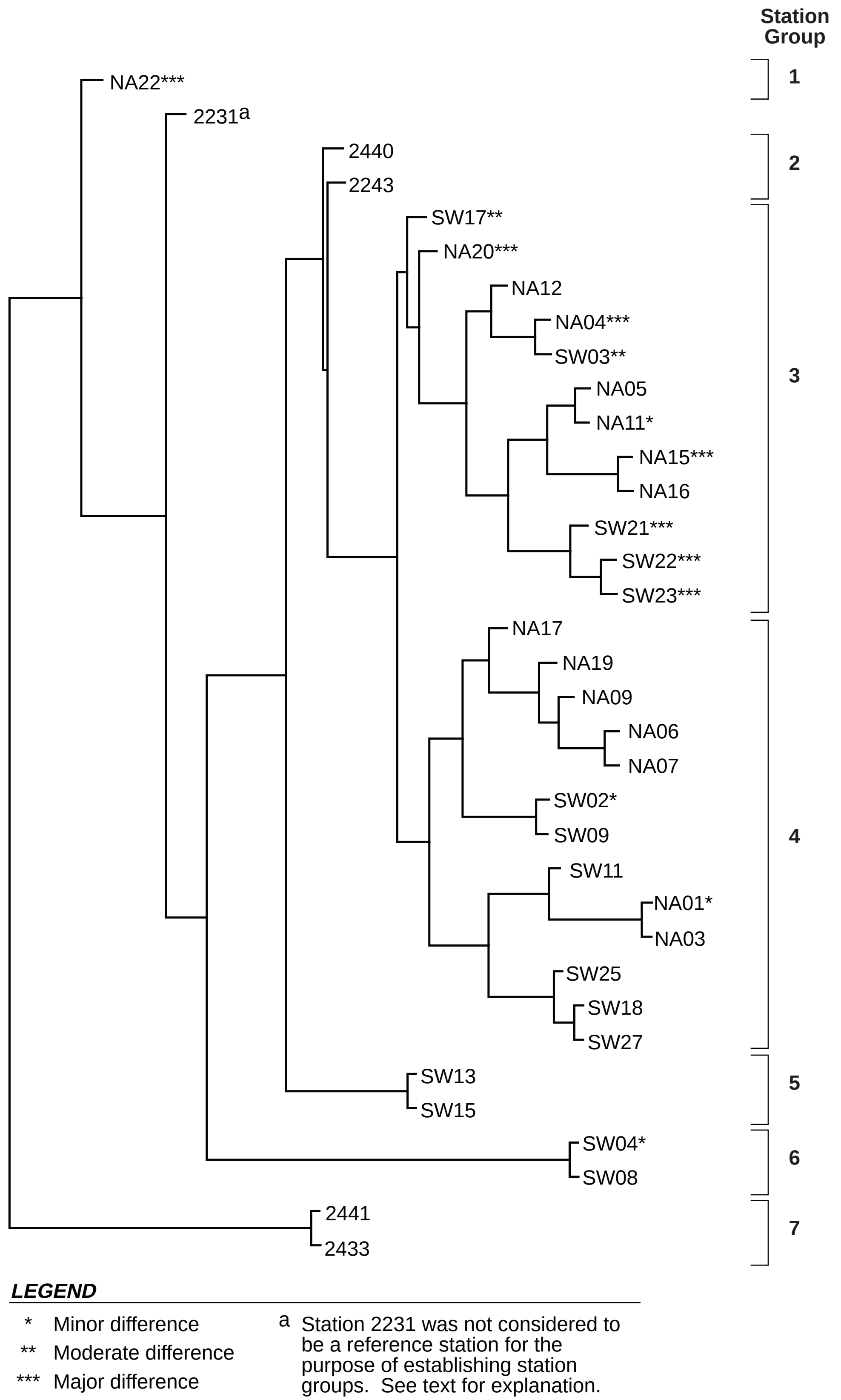


Figure 8-16. Similarity dendrogram for benthic macroinvertebrate communities

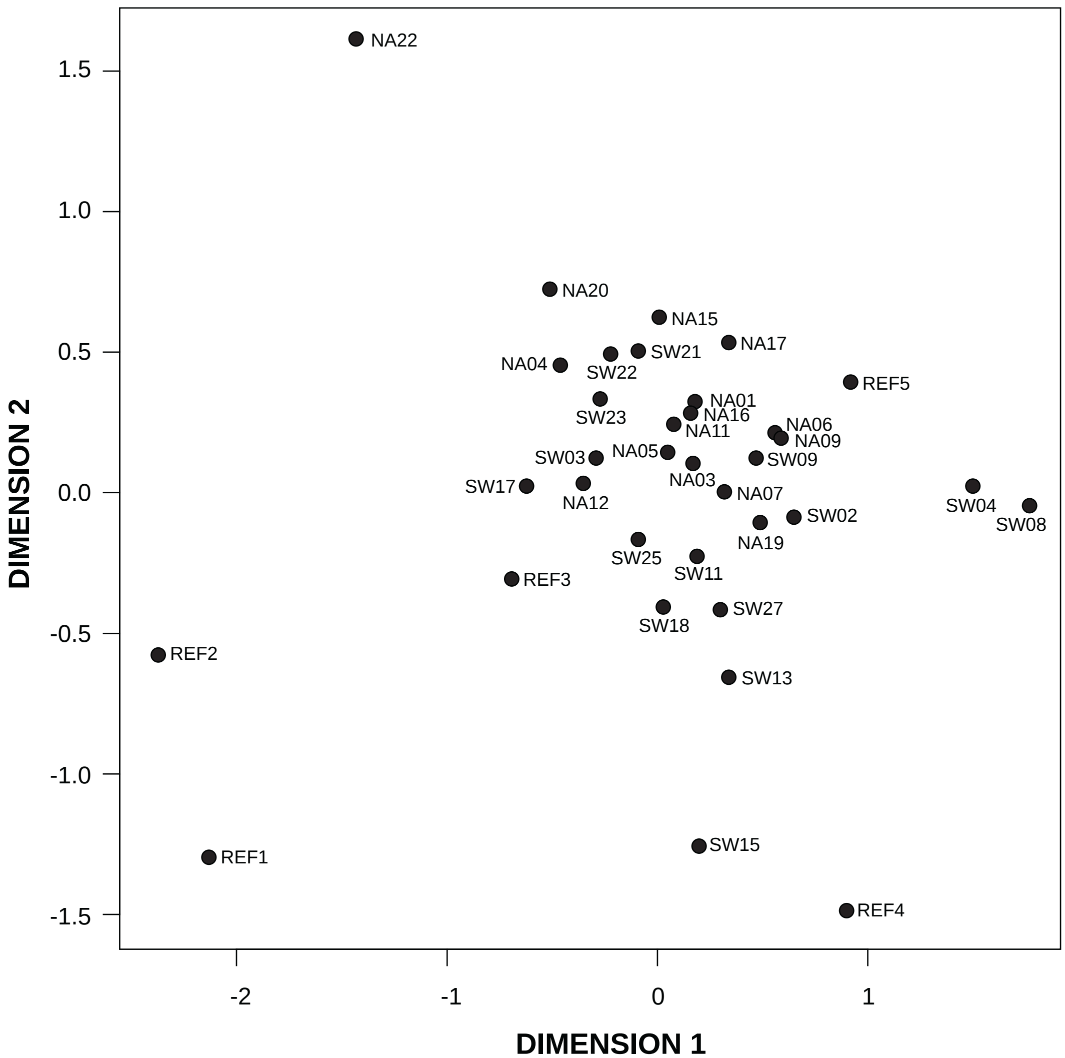


Figure 8-17. Non-metric multidimensional scaling (MDS) plot of benthic macroinvertebrate abundances

8601718.002 1201 - 8601731.002 1201 10/08/03 WA

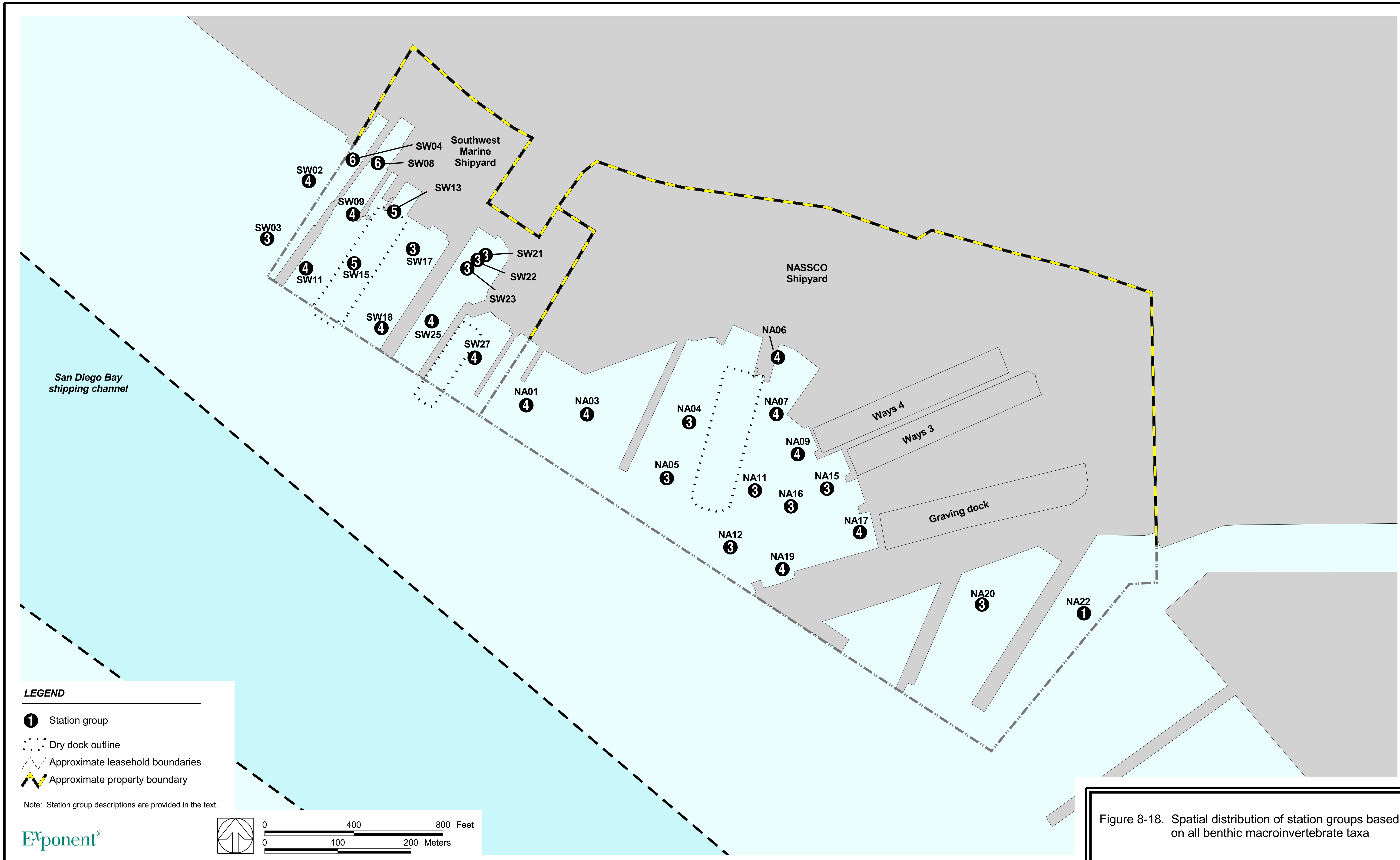


Figure 8-18. Spatial distribution of station groups based on all benthic macroinvertebrate taxa

8601718/1731.002 1201 | Oct 08, 2003 | fig 08-18 macroinvertebrate all view | Fig 08-18 Macroinvertebrate all layout | g:/swm_nassco/projects/shipyards_rpt.apr

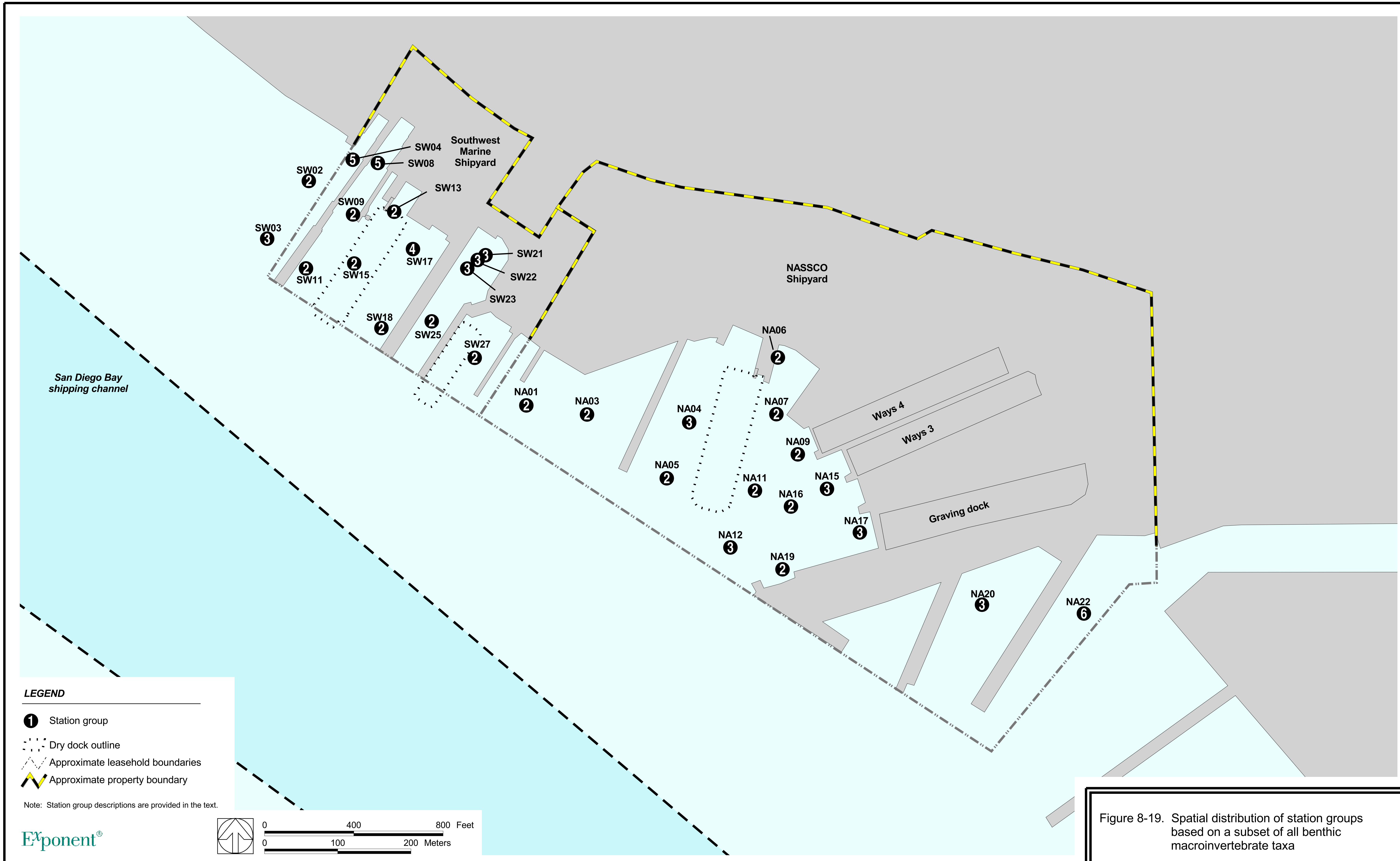


Figure 8-19. Spatial distribution of station groups based on a subset of all benthic macroinvertebrate taxa

8601718/1731.002 1201 | Oct 08, 2003 | fig 08-19 macroinvertebrate subset view | Fig 08-19 Macroinvertebrate subset layout | g:/swm_nassco/projects/shipyards_rpt.apr

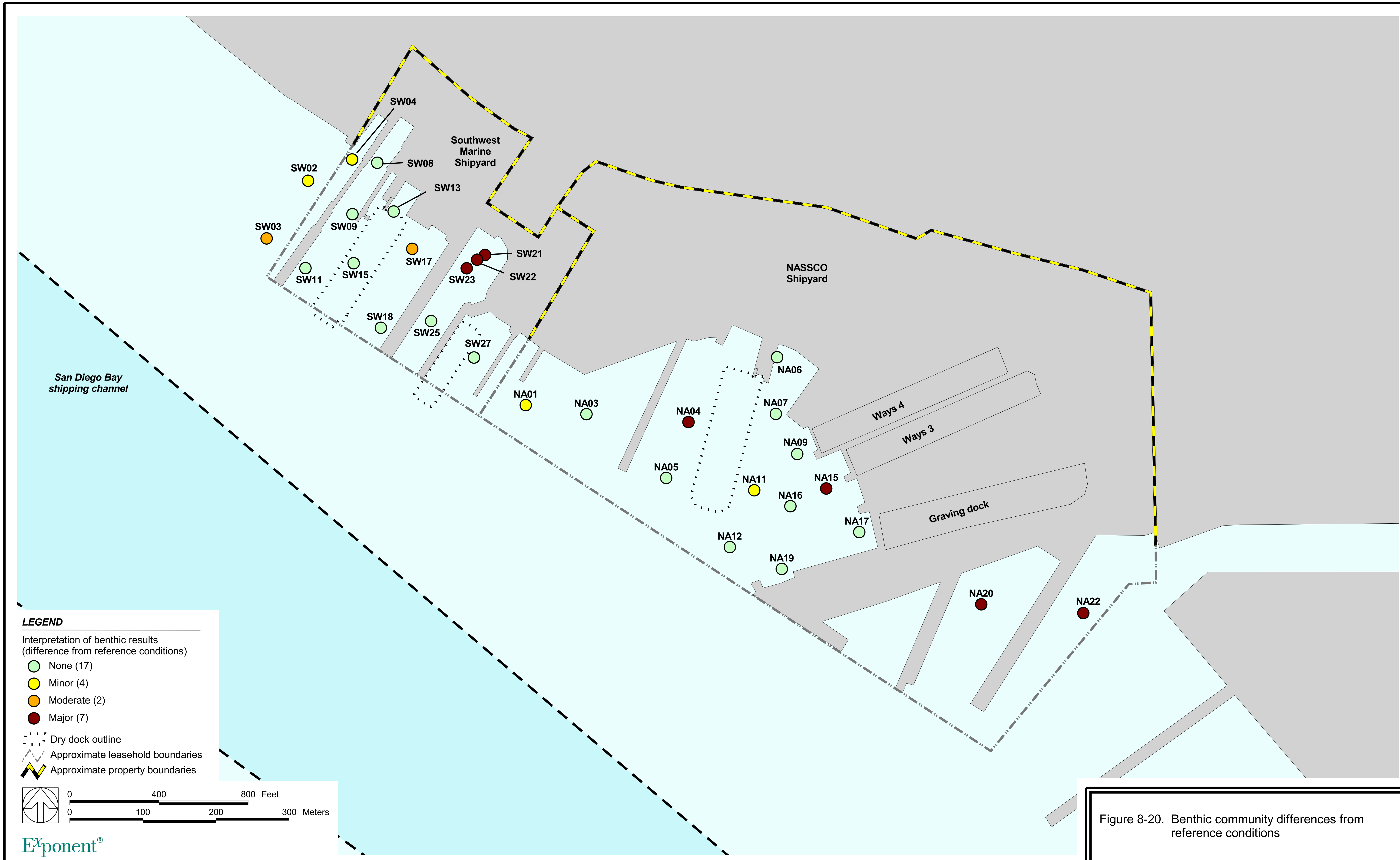


Figure 8-20. Benthic community differences from reference conditions

8601718/8601731.002 1201 | Oct 08, 2003 | fig 08-20 benthic diff view | Fig 08-20 benthic diff layout | g:/swm_nassco/projects/shipyards_rpt.apr

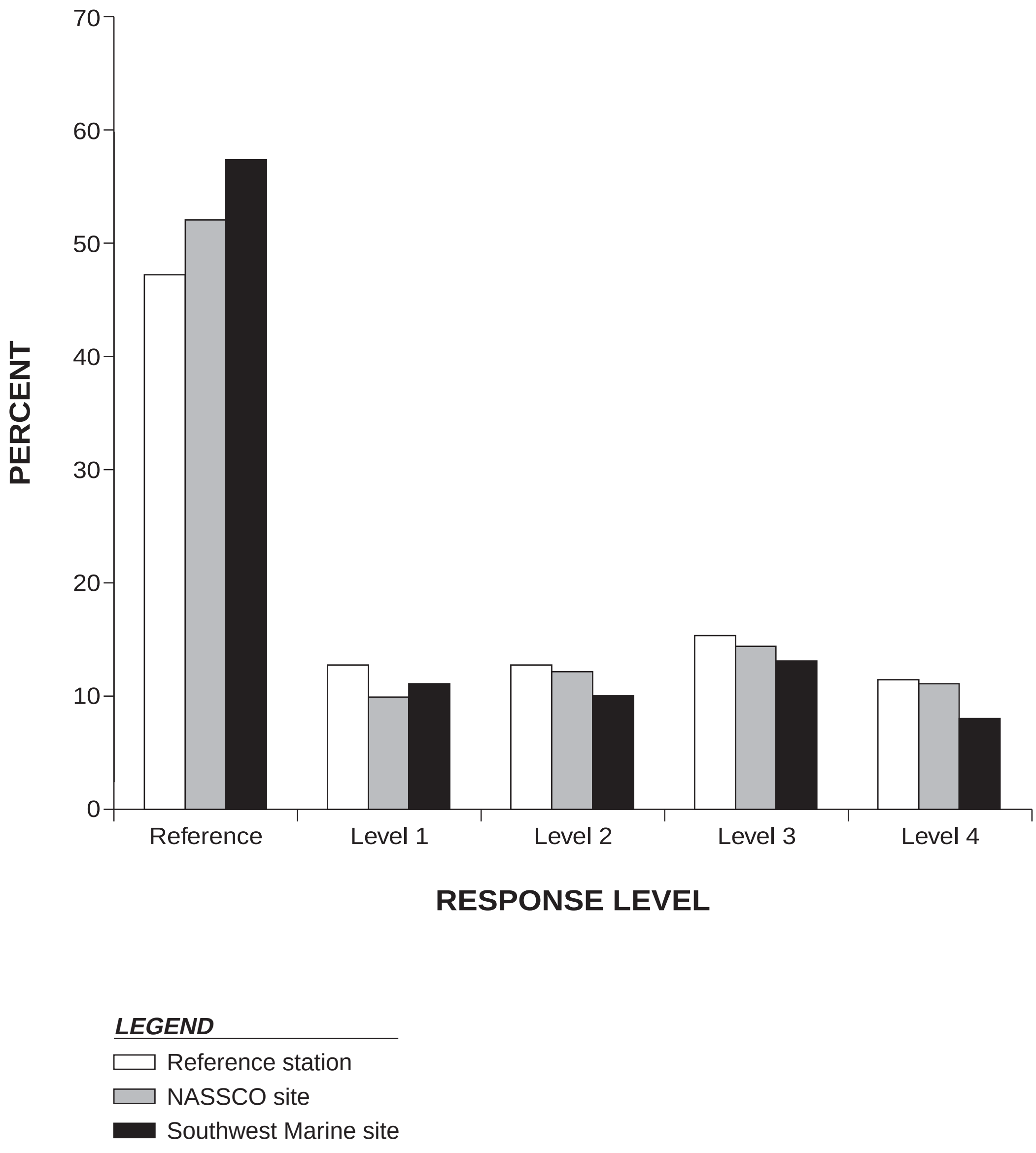


Figure 8-21. Percentage of species from various benthic response levels at the reference stations and shipyard sites

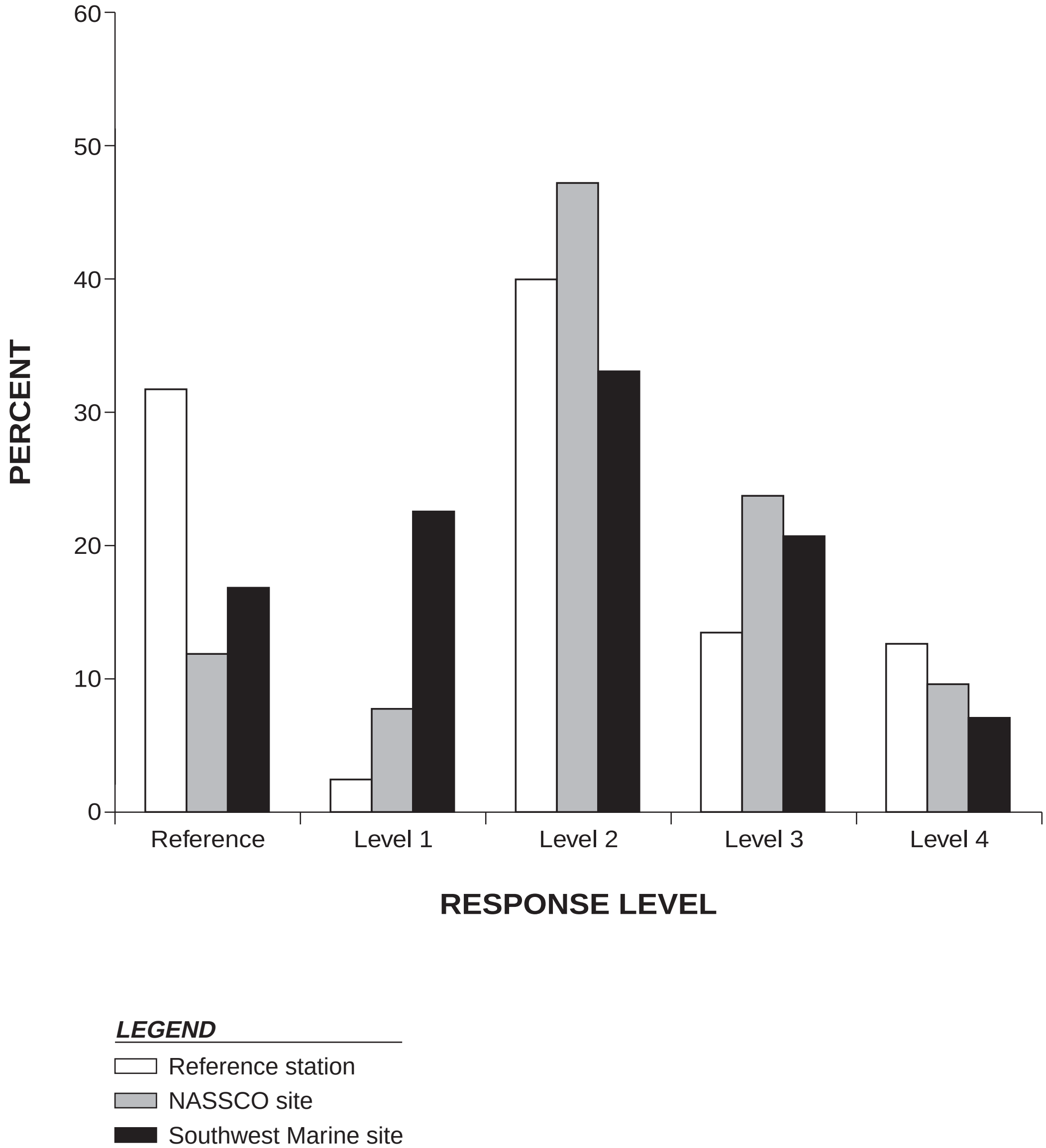


Figure 8-22. Percentage of individuals from various benthic response levels at the reference stations and shipyard sites

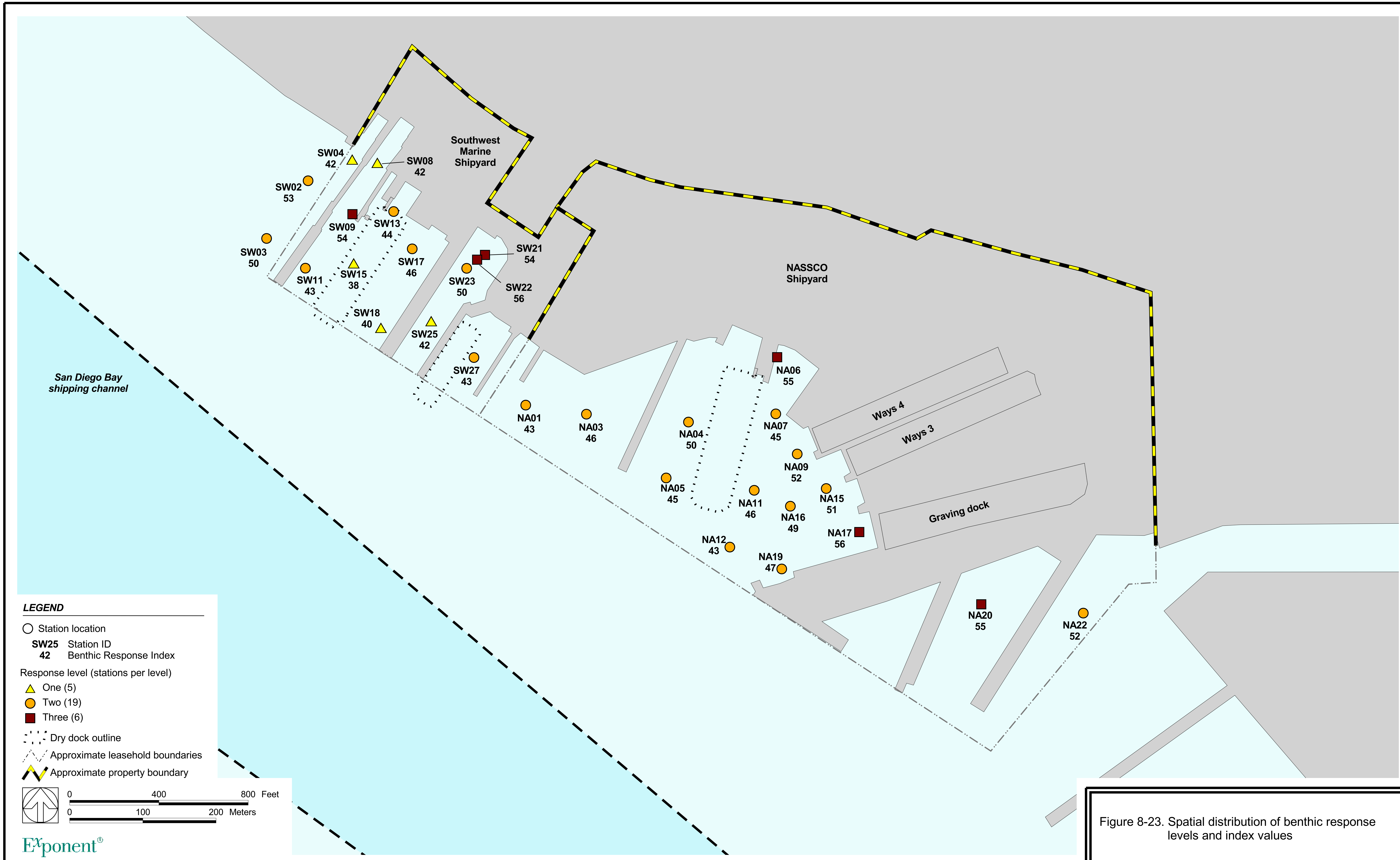
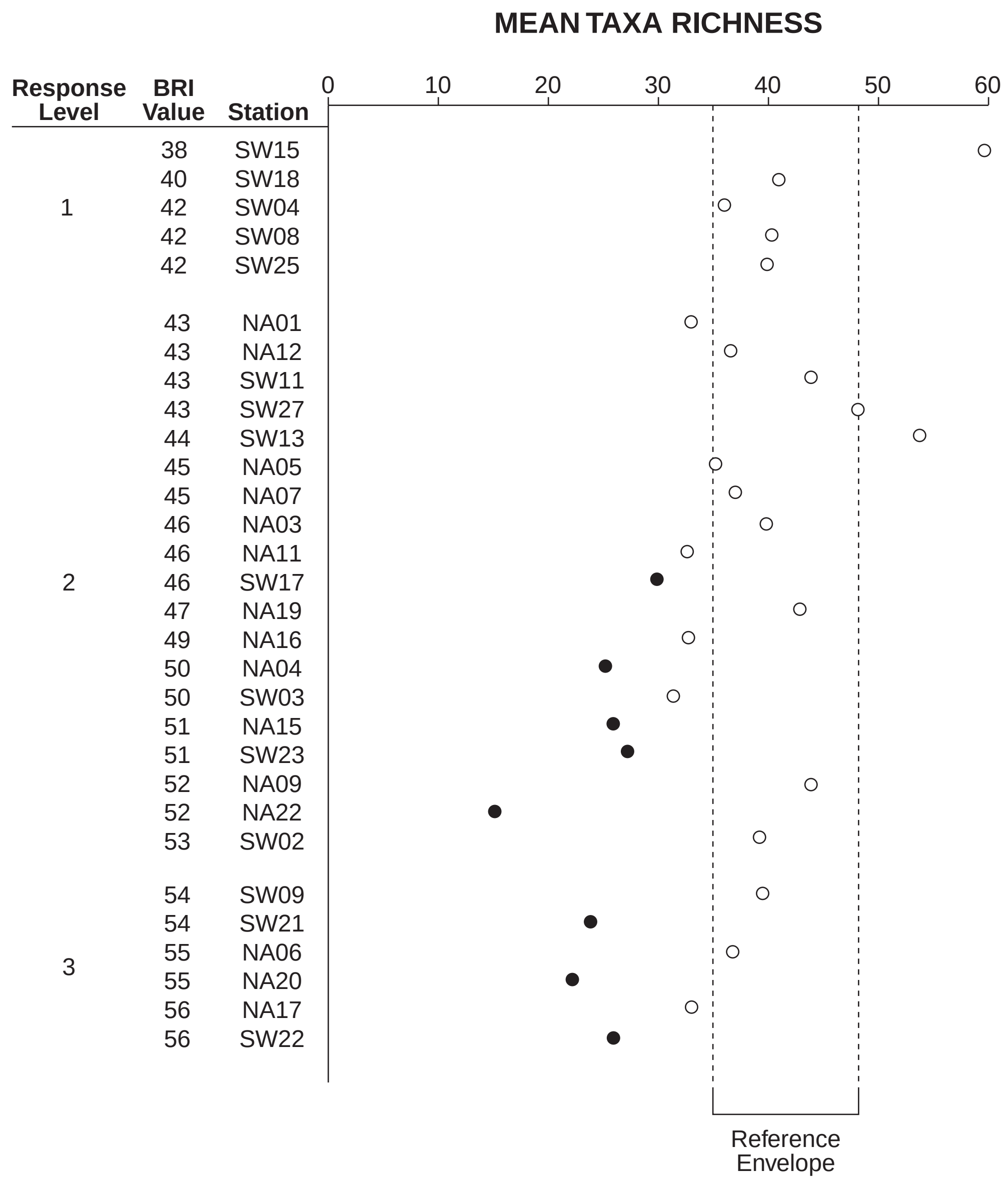


Figure 8-23. Spatial distribution of benthic response levels and index values

8601718/8601731.002 1201 | Oct 08, 2003 | fig 08-23 BRI response view | Fig 08-23 BRI lvls layout | g:/swm_nassco/projects/shipyards_rpt.apr



LEGEND

○ Value is not significantly different (p>0.05) from mean reference value of 40.2 taxa

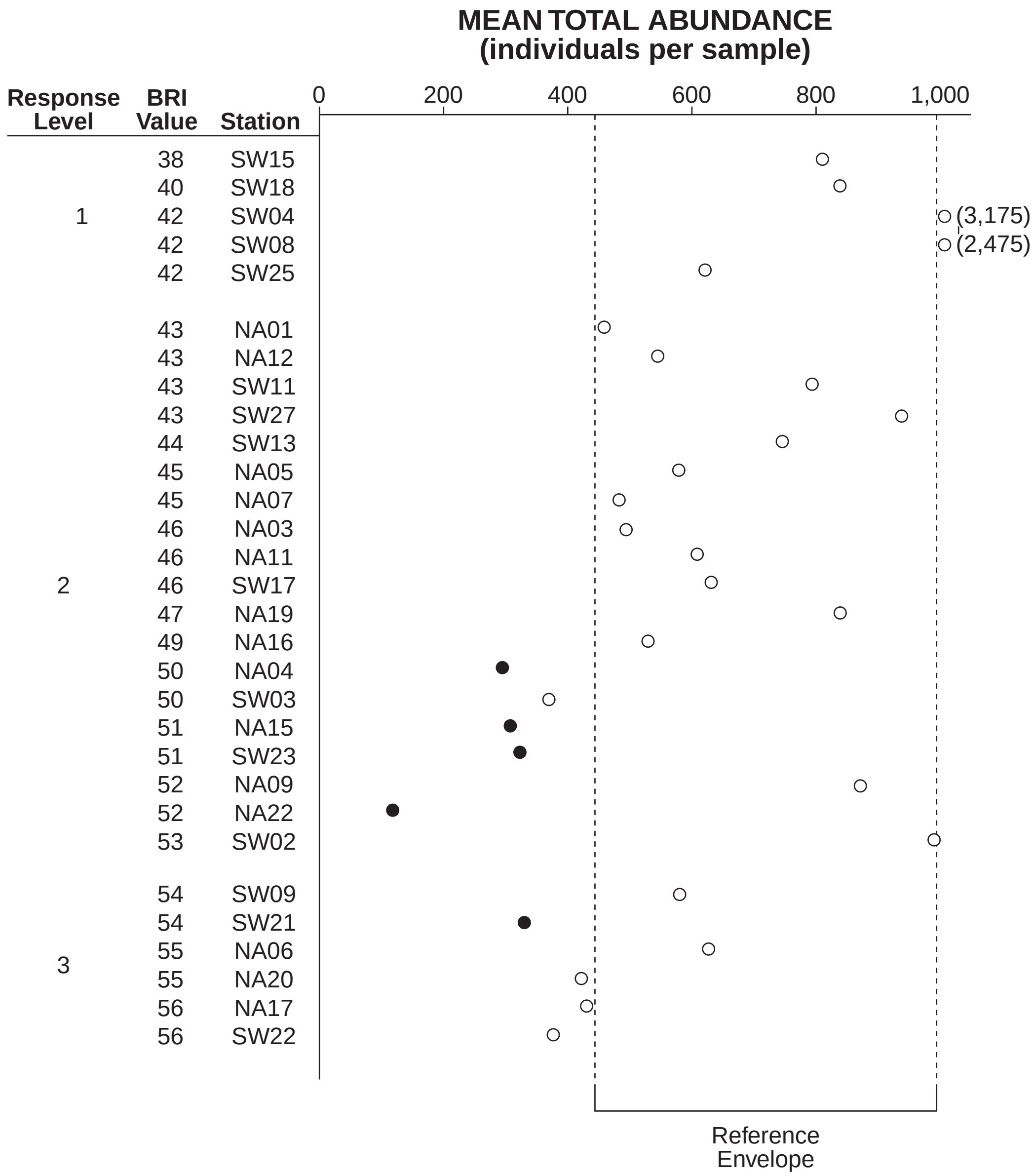
● Value is significantly different ($p \leq 0.05$) from mean reference value

BRI Benthic response index

Note: Correlation between mean taxa richness and BRI values was significant (Spearman $r_S = -0.59$; $p \leq 0.01$)

Figure 8-24. Comparison of mean taxa richness at the shipyard stations with BRI values and reference conditions

8601718.002 1201 - 8601731.002 1201 10/08/03 WA



LEGEND

○ Value is not significantly different (p>0.05) from mean reference value of 643 individuals per sample

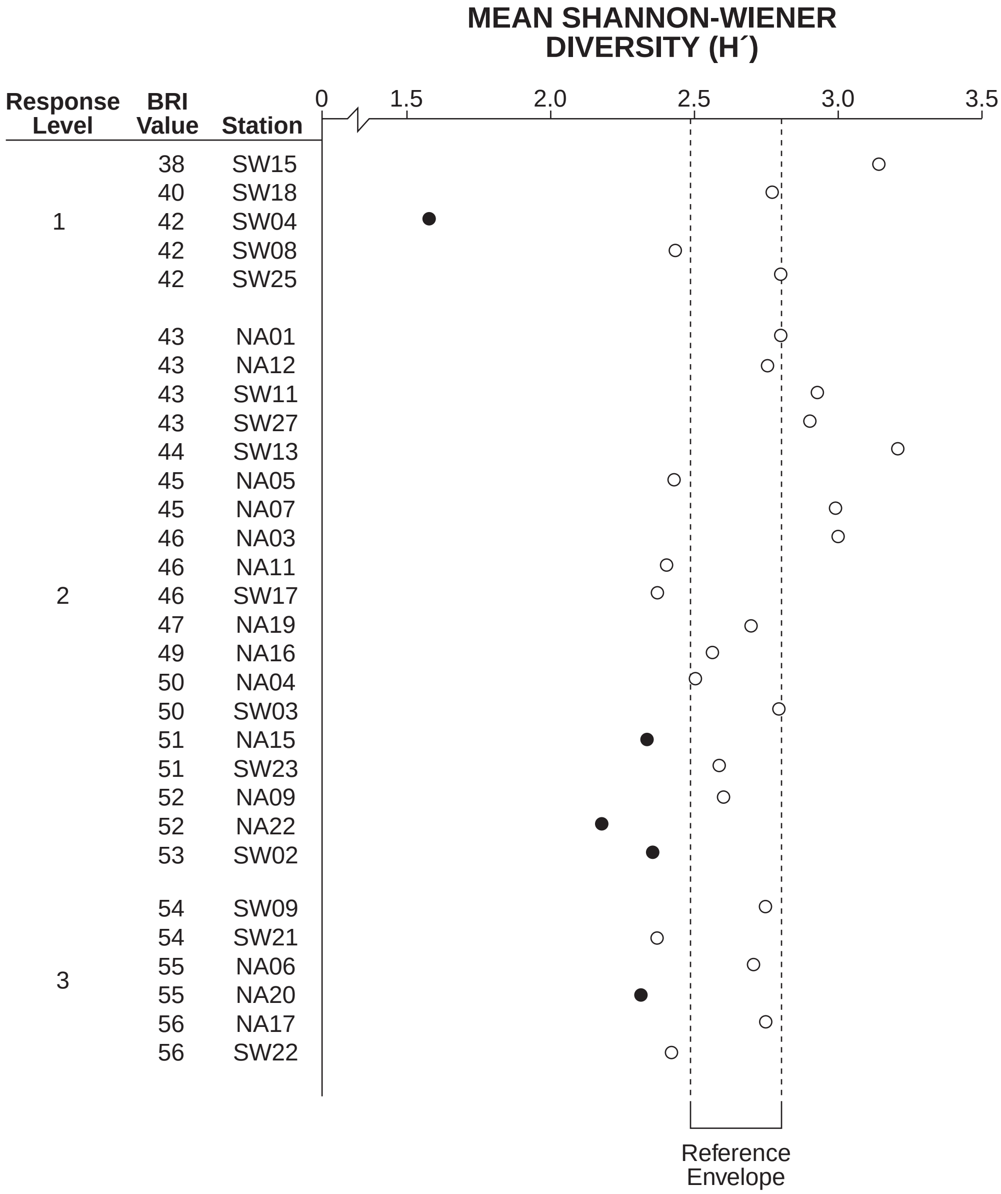
● Value is significantly different ($p \leq 0.05$) from mean reference value

BRI Benthic response index

Note: Correlation between mean total abundance and BRI values was significant (Spearman $r_S = -0.53$; $p \leq 0.01$)

Figure 8-25. Comparison of mean total abundance at the shipyard stations with BRI values and reference conditions

8601718.002 1201 - 8601731.002 1201 10/08/03 WA



LEGEND

- ○ Value is not significantly different ($p>0.05$) from mean reference value of 2.65
- ● Value is significantly different ($p \leq 0.05$) from mean reference value

BRI Benthic response index

Note: Correlation between mean species diversity and BRI values was significant (Spearman $r_S = -0.43$; $p \leq 0.05$)

Figure 8-26. Comparison of mean species diversity at the shipyard stations with BRI values and reference conditions

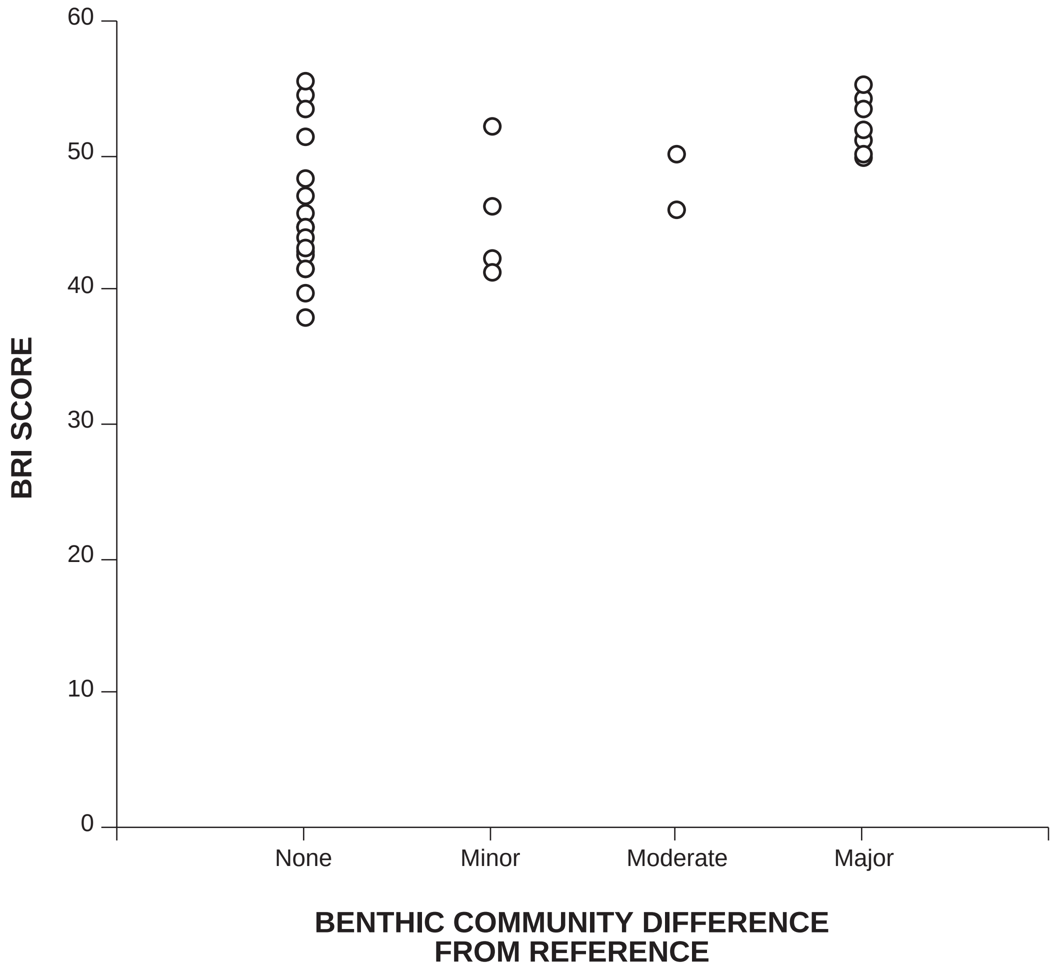


Figure 8-27. Relationship between BRI scores and benthic community alterations

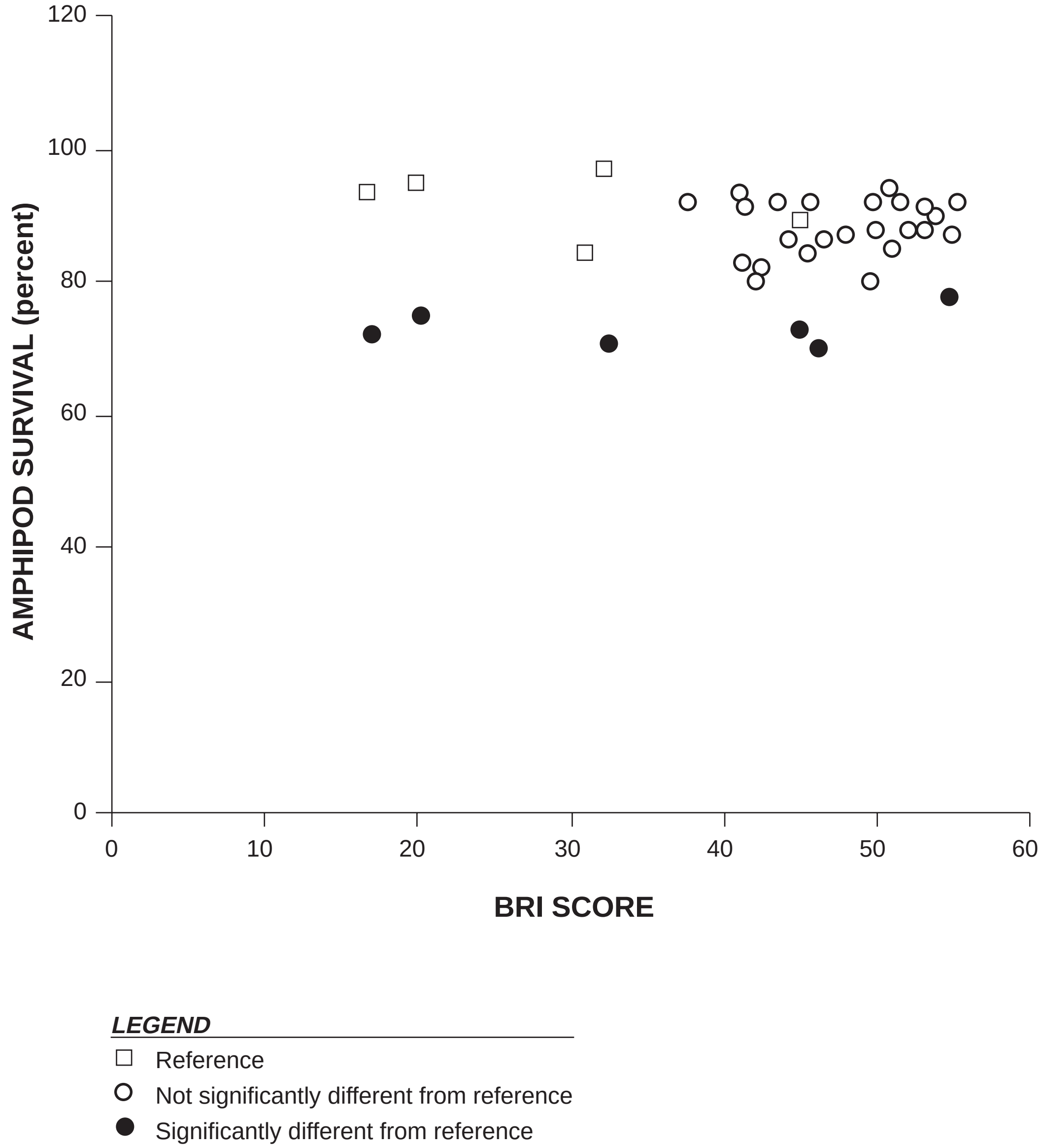


Figure 8-28. Relationship between BRI scores and amphipod survival

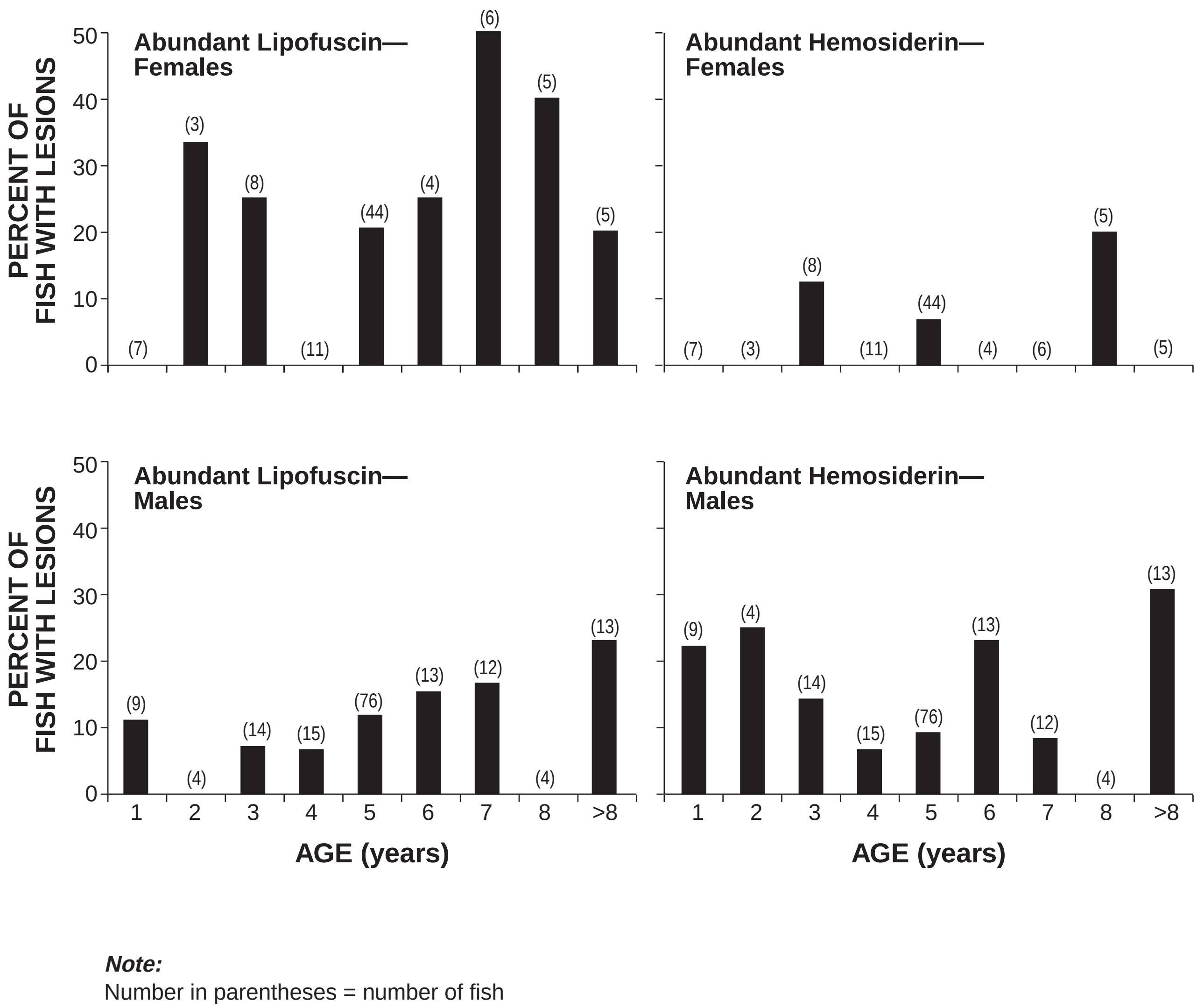


Figure 8-29. Comparison of the prevalences of abundant lipofuscin and abundant hemosiderin with age of spotted sand bass

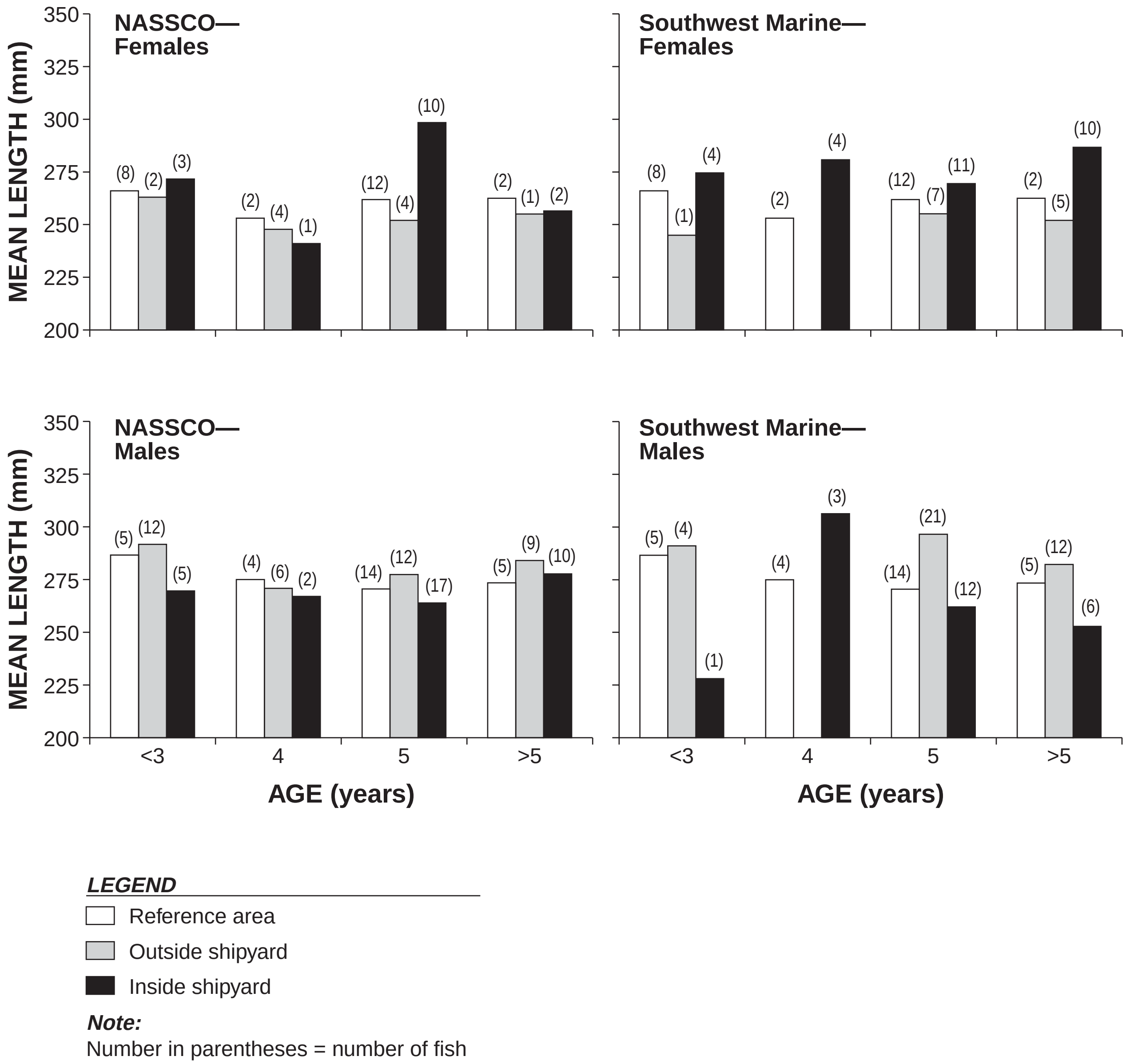


Figure 8-30. Comparison of mean length of spotted sand bass captured at the shipyard locations and the reference area

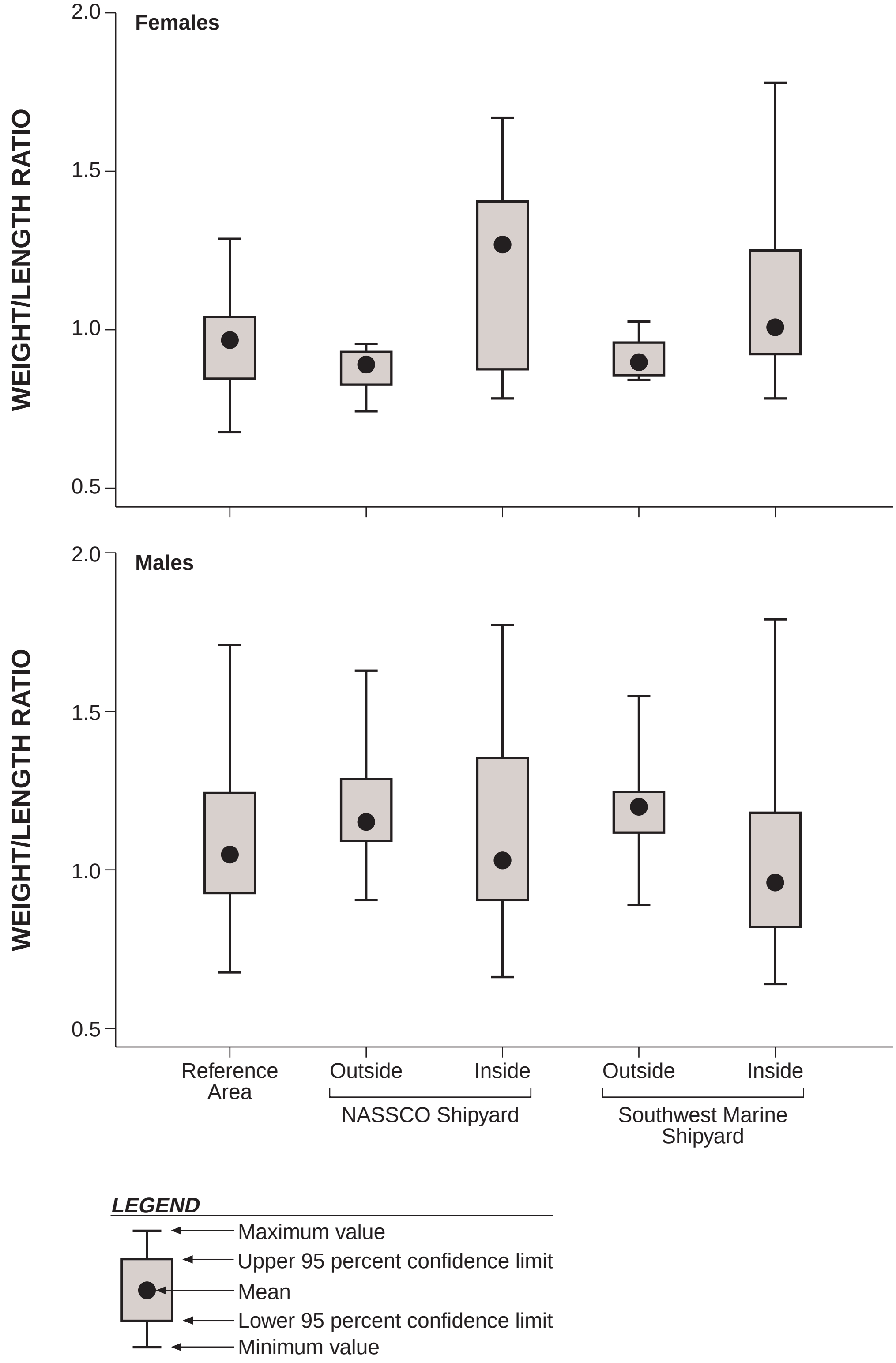


Figure 8-31. Comparison of mean weight/length ratios of spotted sand bass among shipyard locations and the reference area

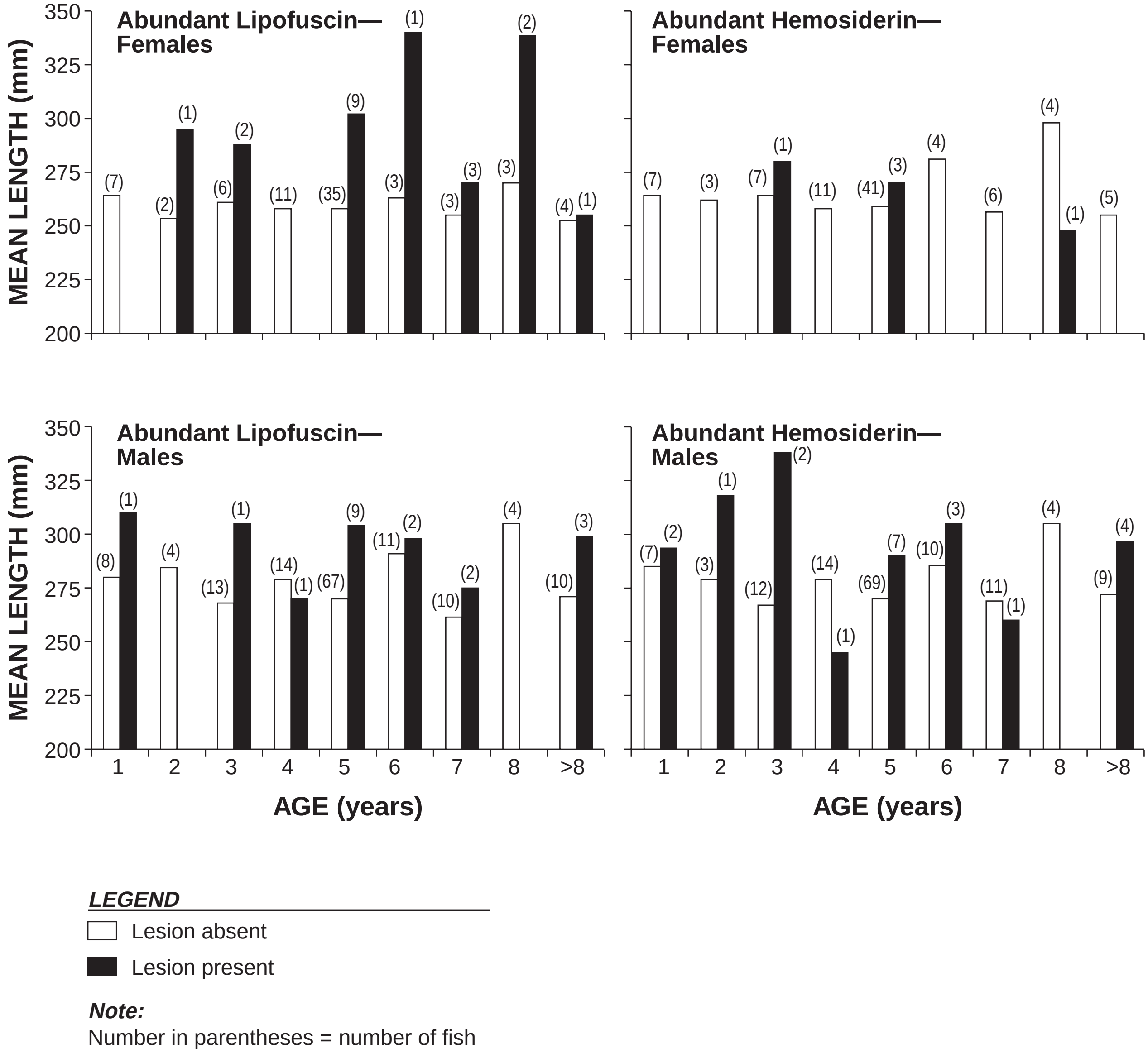


Figure 8-32. Comparison of mean length for spotted sand bass in which abundant lipofuscin and abundant hemosiderin were absent or present

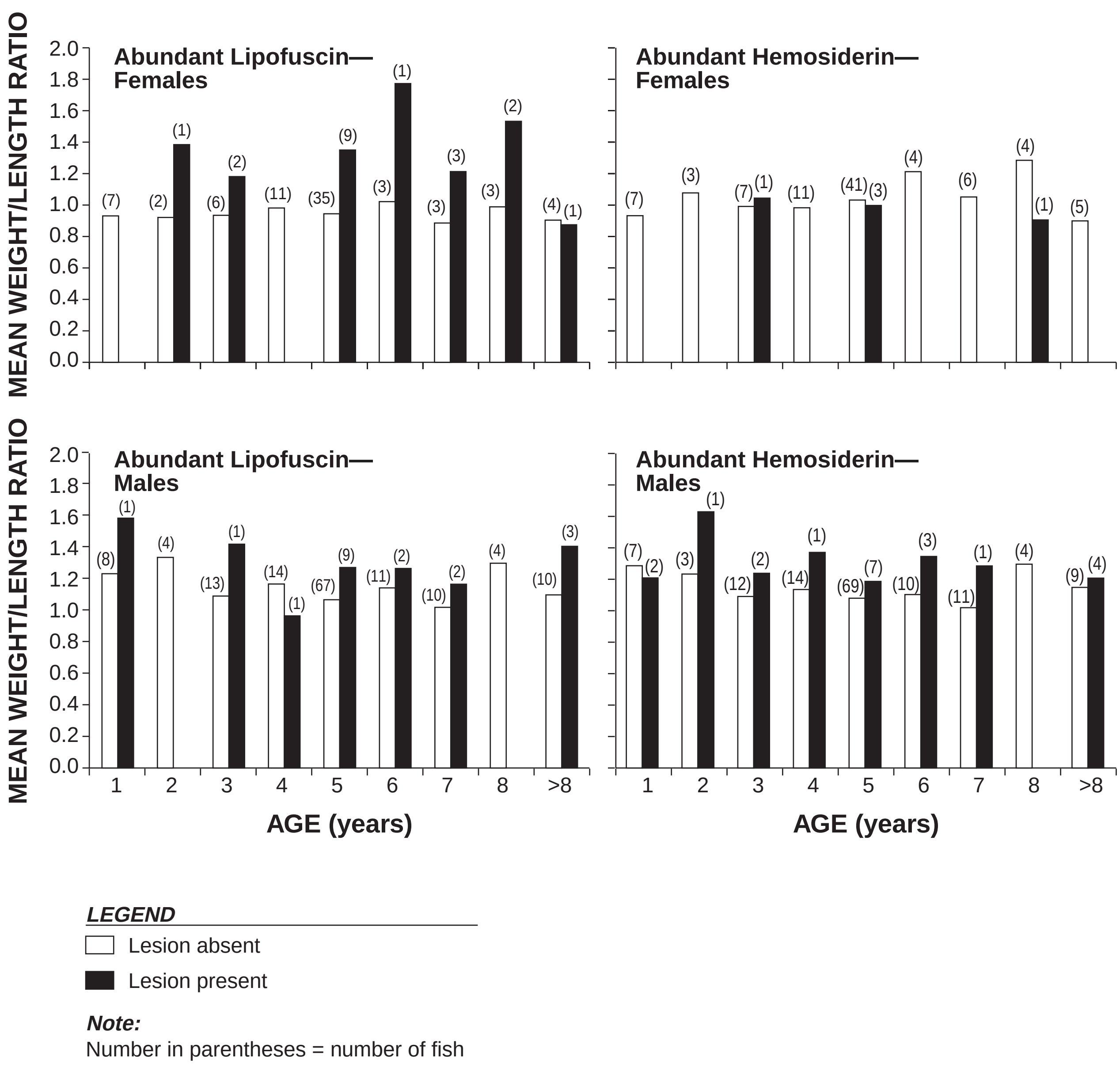


Figure 8-33. Comparison of mean weight/length ratios at age for spotted sand bass in which abundant lipofuscin and abundant hemosiderin were absent or present

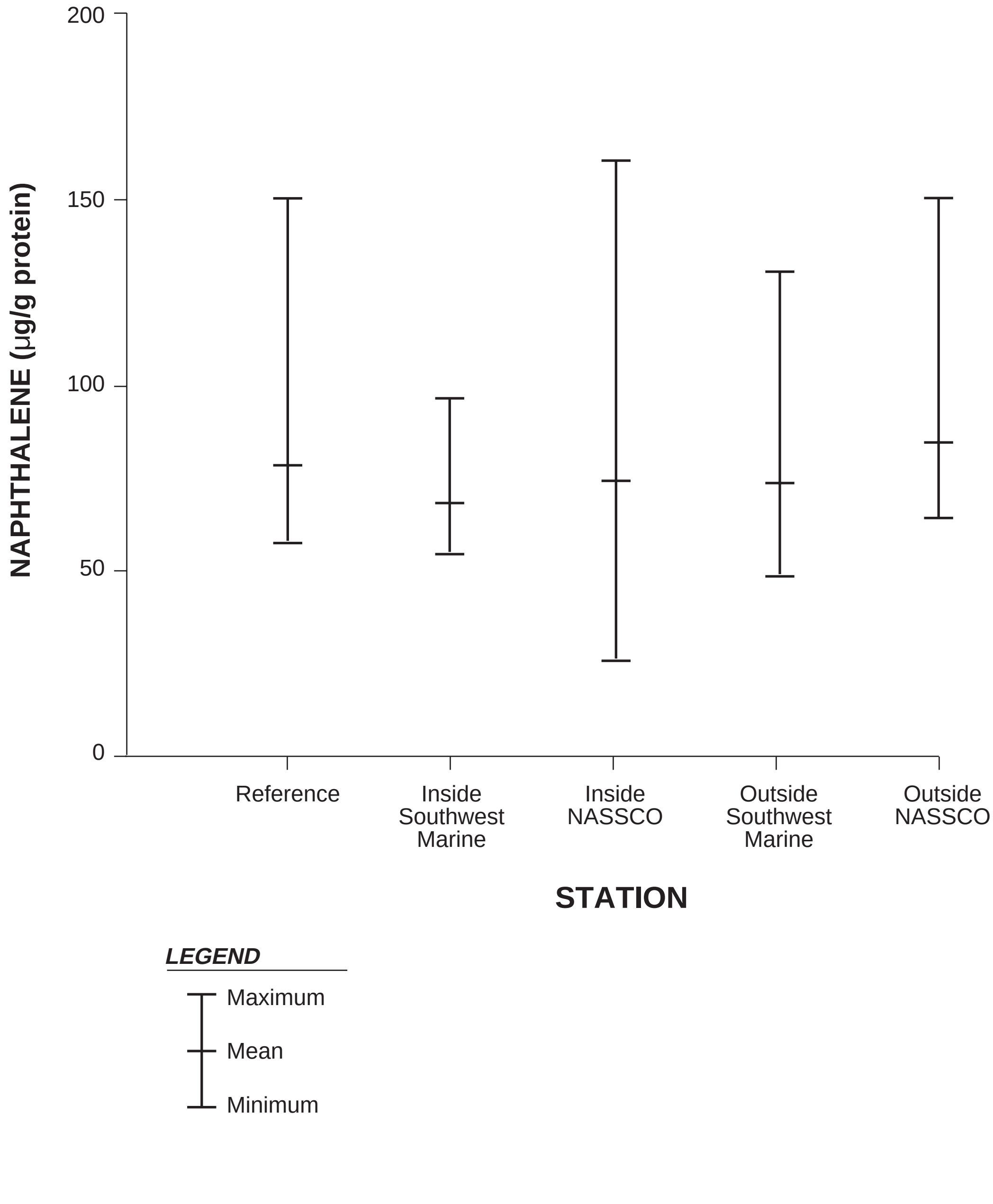


Figure 8-34. Naphthalene breakdown products in fish bile

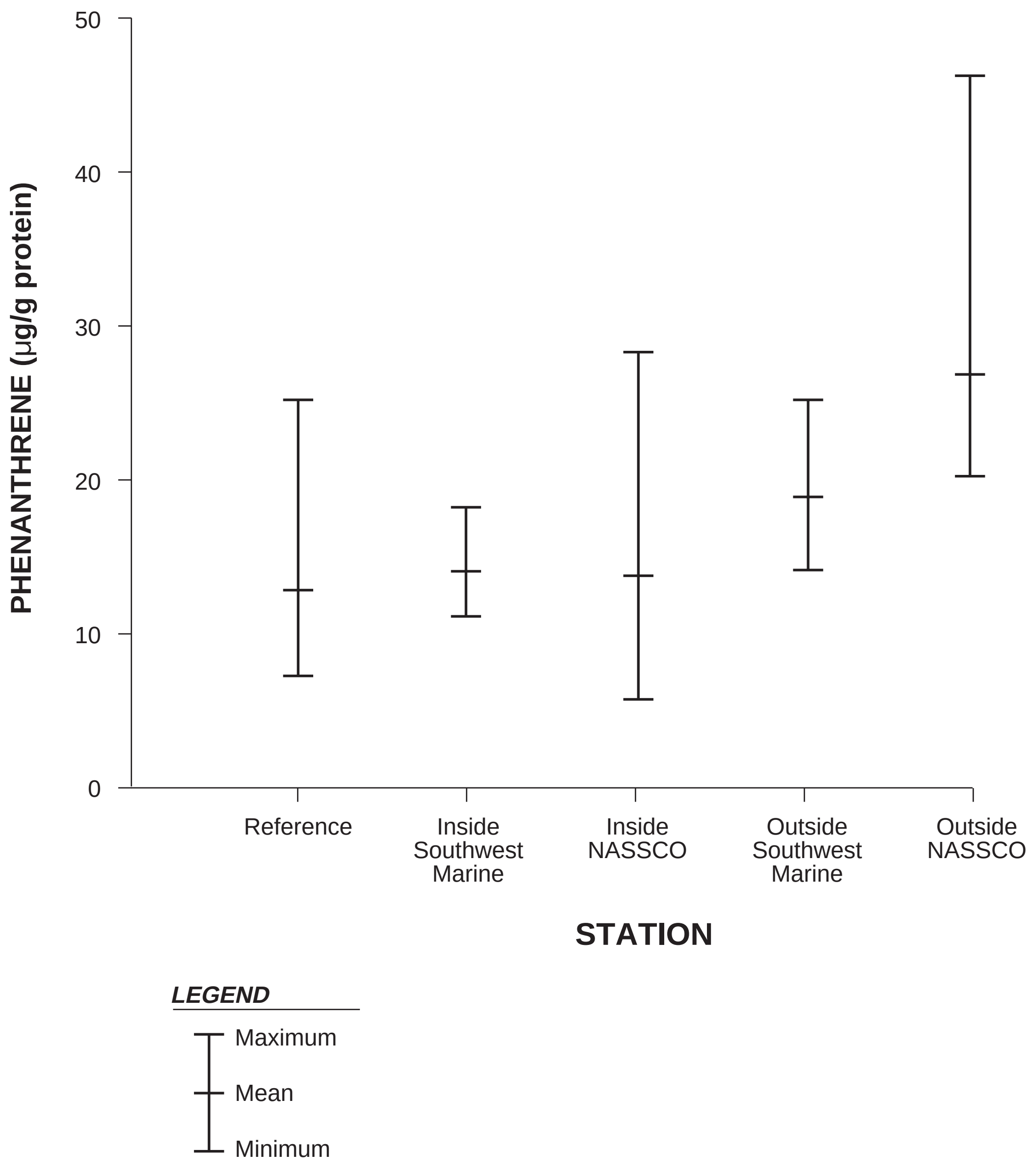


Figure 8-35. Phenanthrene breakdown products in fish bile

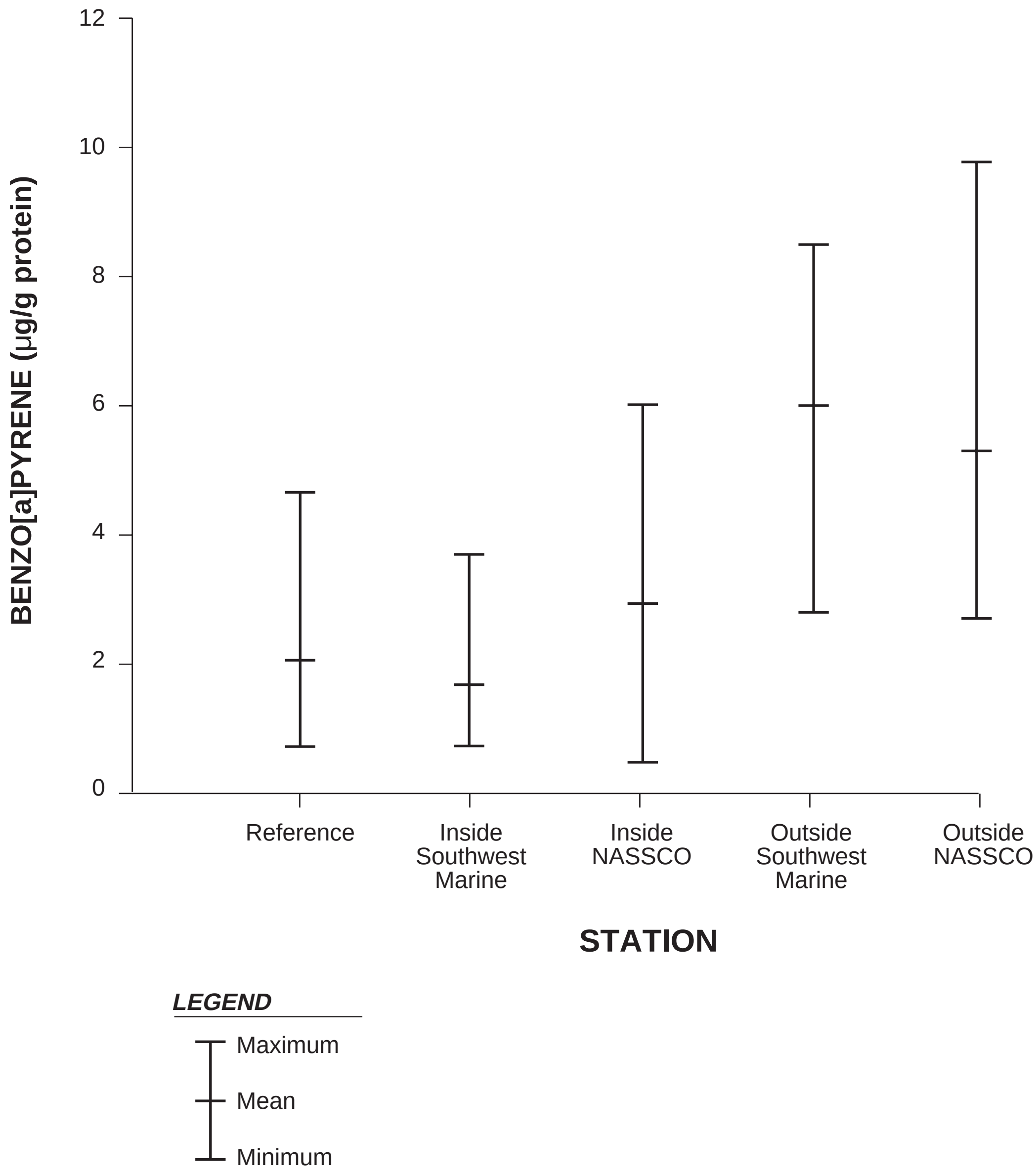


Figure 8-36. Benzo[a]pyrene breakdown products in fish bile

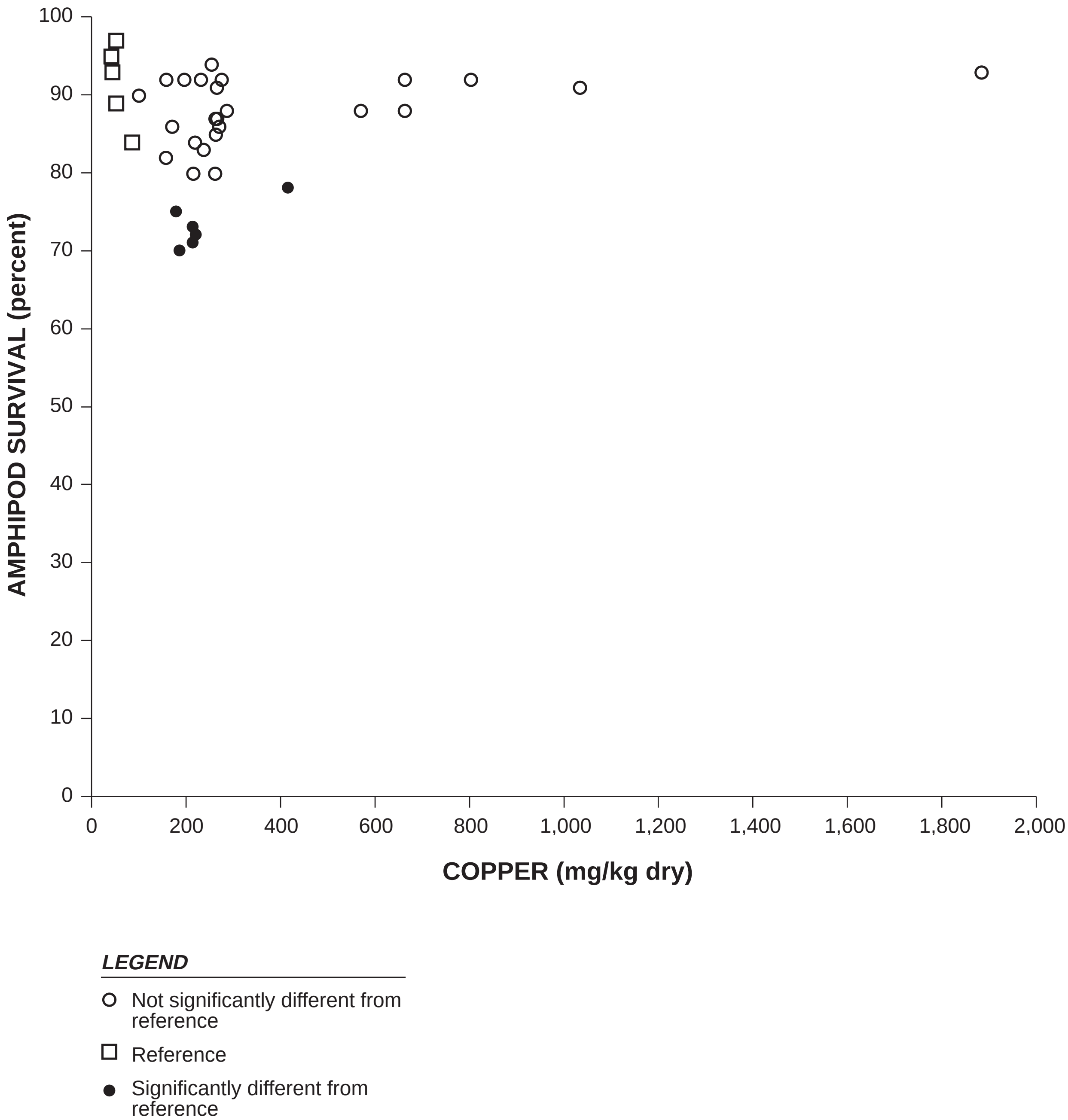


Figure 9-1. Sediment copper concentrations in relation to amphipod survival

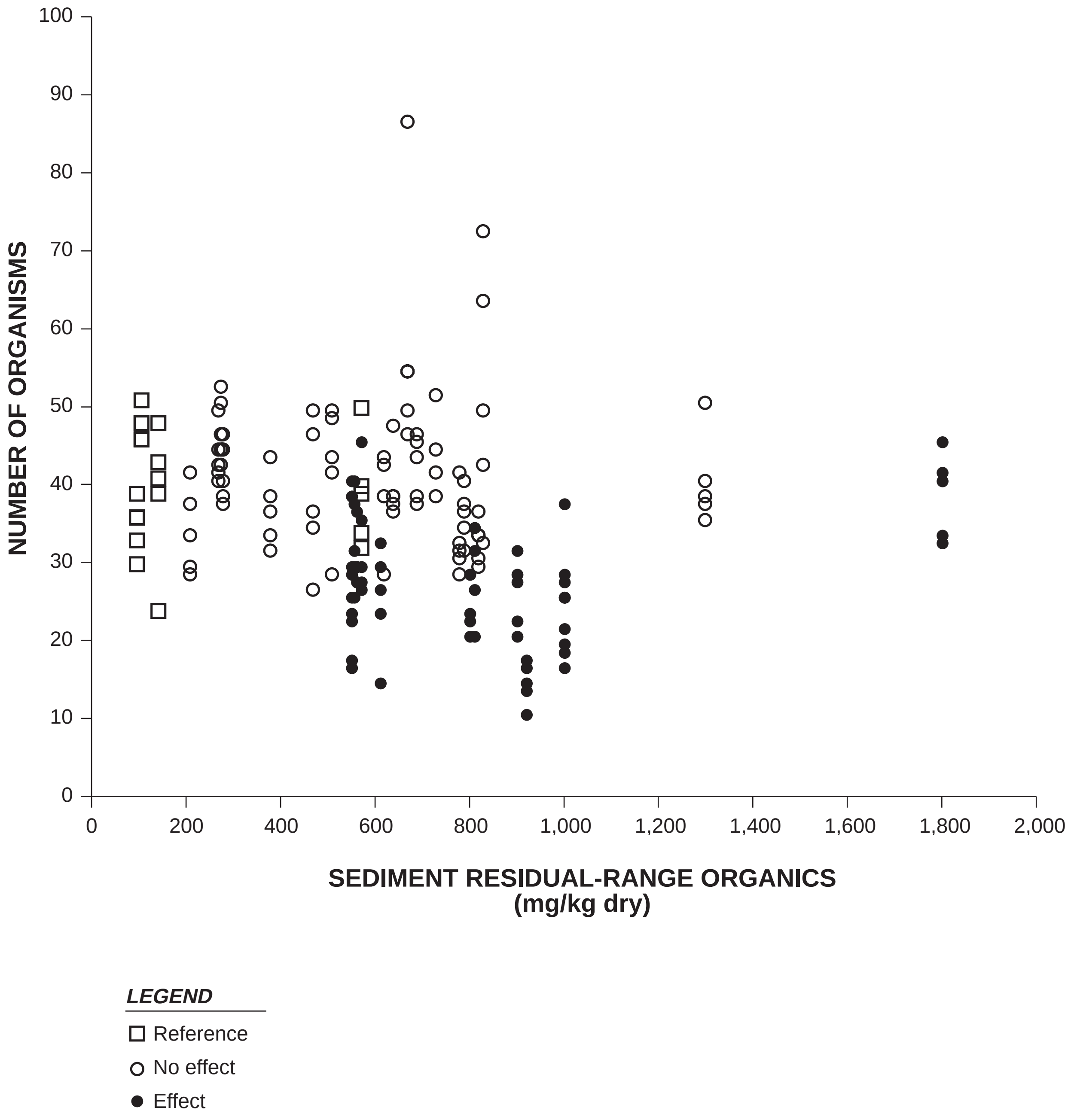


Figure 9-2. Sediment residual-range organics in relation to benthic macroinvertebrate richness

BENTHIC EFFECTS ONLY[b]		VERTICAL GRADIENT OF PERCENT FINES[a]	
		No	Yes
	Yes	5	0
	No	3	6

Note:

Numerical entries represent the number of stations in each category

a The absence of a vertical gradient may indicate physical disturbance of the sediment

b "Yes" indicates that the only type of effect observed was a difference in the benthic macroinvertebrate community.

"No" indicates that either no adverse biological effects were observed or that those effects included amphipod or bivalve toxicity.

Figure 9-3. Relationship between percent fine profiles and benthic macroinvertebrate communities

8601718.002 1201 - 8601731.002 1201 10/08/03 WA

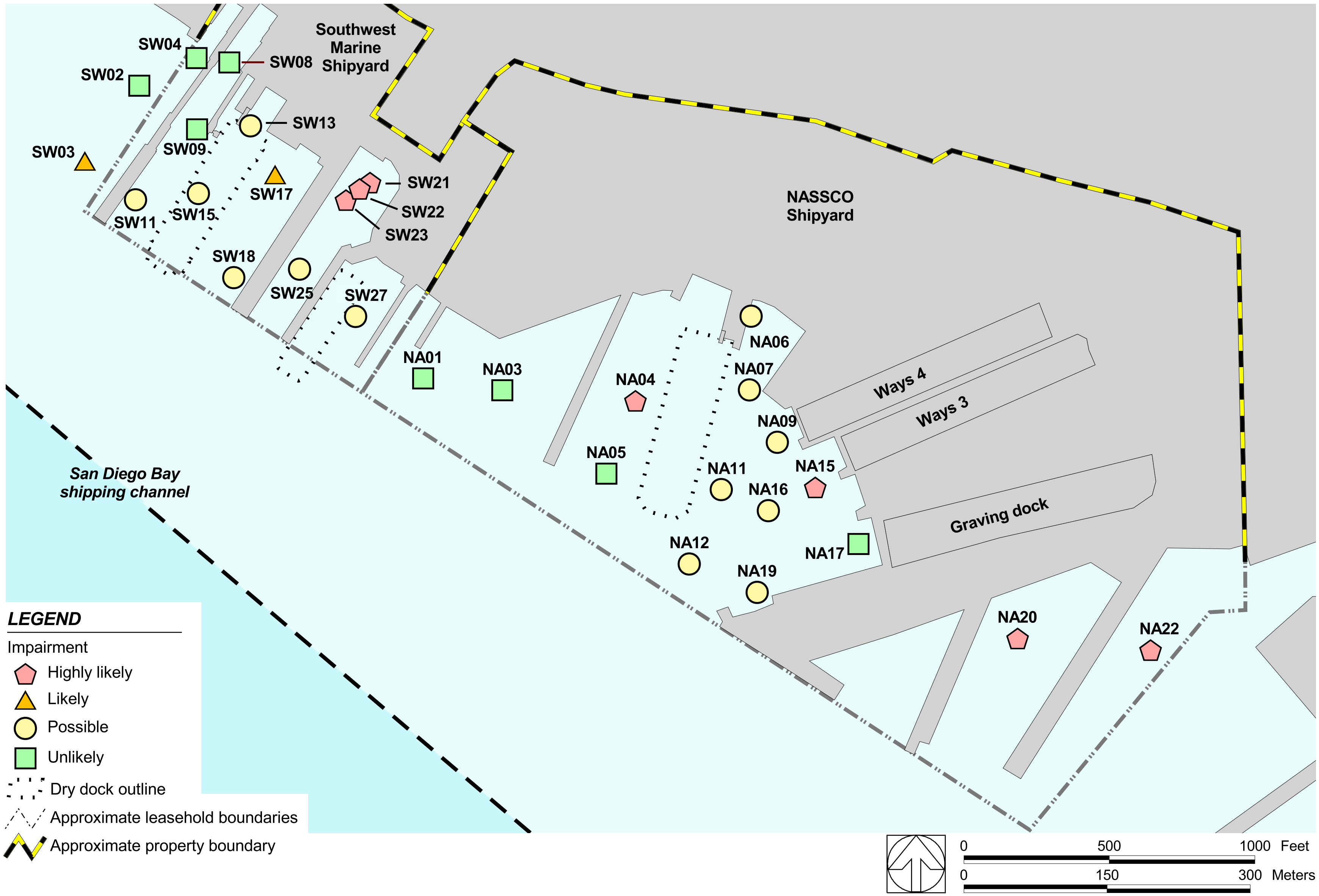


Figure 9-4. Map of potential effects on aquatic life beneficial use

Exponent®

8301718/8601731.002 1201 | Sep 22, 2003 | fig 09-4 impairment view | Fig 09-4 impairment layout | g:/swm_nassco/projects/shipyards_rpt.apr

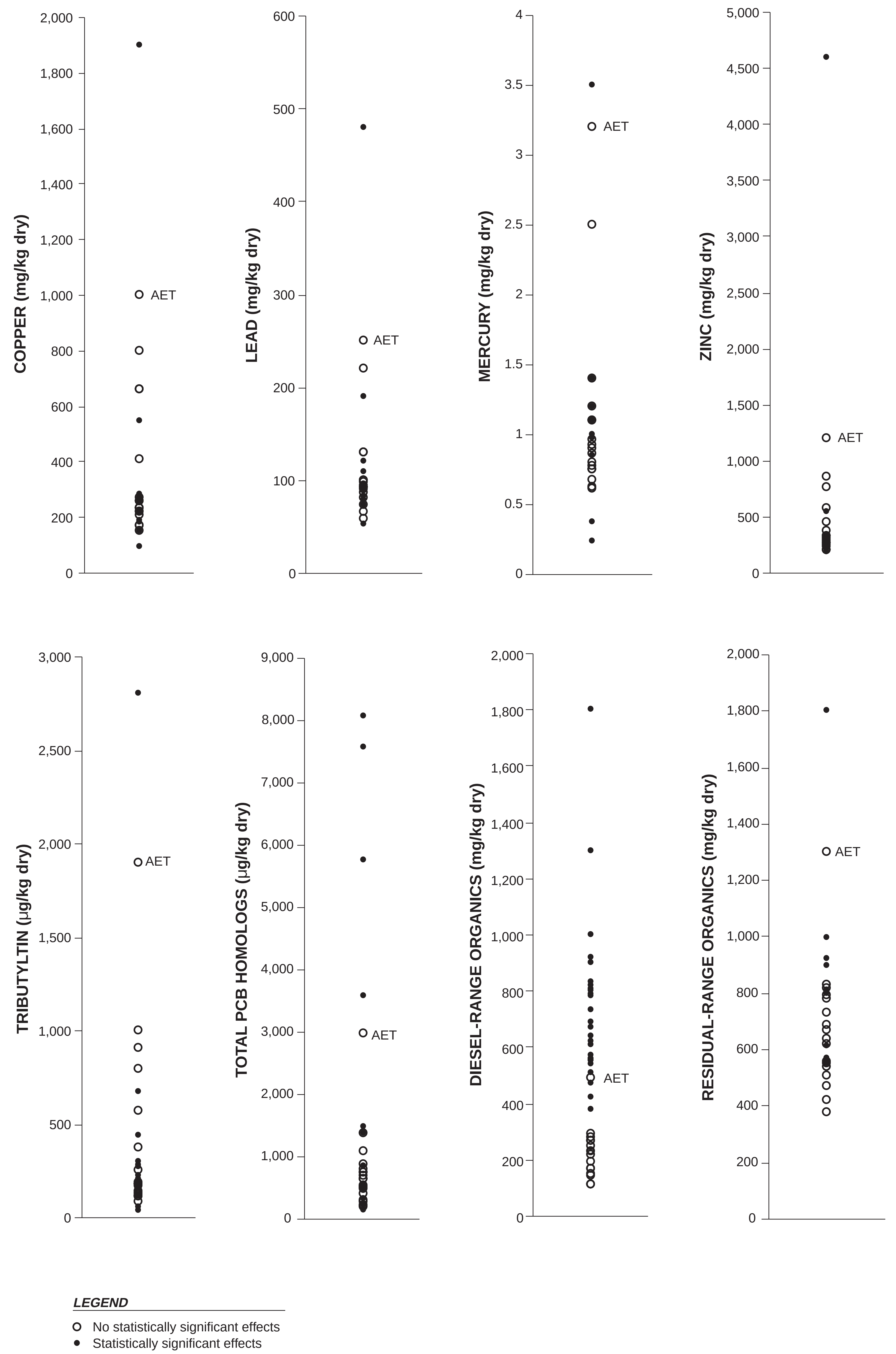


Figure 9-5. Ranges of chemical concentrations for any benthic differences

8601718.002 1201 - 8601731.002 1201 10/08/03 WA

PREDICTED EFFECTS

ACTUAL EFFECTS	Yes	No	Total
Yes	4	5	9
No	2	19	21
Total	6	24	30

Correct predictions

$$\text{Sensitivity} = \frac{4}{9} = 44\%$$

$$\text{Specificity} = \frac{19}{21} = 90\%$$

$$\text{Efficiency} = \frac{4}{6} = 67\%$$

$$\text{Reliability} = \frac{4 + 19}{30} = 77\%$$

Figure 9-6. Illustration of reliability and related calculations

LAET | **95% UPL of Final Reference Pool**

A Detailed

LAET — BENEFICIAL USE IMPAIRMENT vs. EXCEEDS

	EXCEEDS Yes	EXCEEDS No
Highly likely	1	6
Likely	0	2
Possible	0	13
Unlikely	2	6

95% UPL of Final Reference Pool — BENEFICIAL USE IMPAIRMENT vs. EXCEEDS

	EXCEEDS Yes	EXCEEDS No
Highly likely	7	0
Likely	2	0
Possible	13	0
Unlikely	8	0

B Grouped for performance evaluation

LAET — BENEFICIAL USE IMPAIRMENT vs. EXCEEDS

	EXCEEDS Yes	EXCEEDS No
Likely or highly likely	1	8
Possible or unlikely	2	19

Sensitivity: 11%
Specificity: 90%
Efficiency: 33%
Overall reliability: 67%

95% UPL of Final Reference Pool — BENEFICIAL USE IMPAIRMENT vs. EXCEEDS

	EXCEEDS Yes	EXCEEDS No
Likely or highly likely	9	0
Possible or unlikely	21	0

Sensitivity: 100%
Specificity: 0%
Efficiency: 30%
Overall reliability: 30%

C Modified to consider non-shipyard effects*

LAET — BENEFICIAL USE IMPAIRMENT vs. EXCEEDS

	EXCEEDS Yes	EXCEEDS No
Likely or highly likely	1	6
Possible or unlikely	2	21

Sensitivity: 14%
Specificity: 91%
Efficiency: 33%
Overall reliability: 73%

95% UPL of Final Reference Pool — BENEFICIAL USE IMPAIRMENT vs. EXCEEDS

	EXCEEDS Yes	EXCEEDS No
Likely or highly likely	7	0
Possible or unlikely	23	0

Sensitivity: 100%
Specificity: 0%
Efficiency: 23%
Overall reliability: 23%

* Effects at NA20 attributed to physical disturbance, and effects at NA22 attributed to Chollas Creek and the city storm drain

Figure 12-1. Performance of candidate cleanup levels

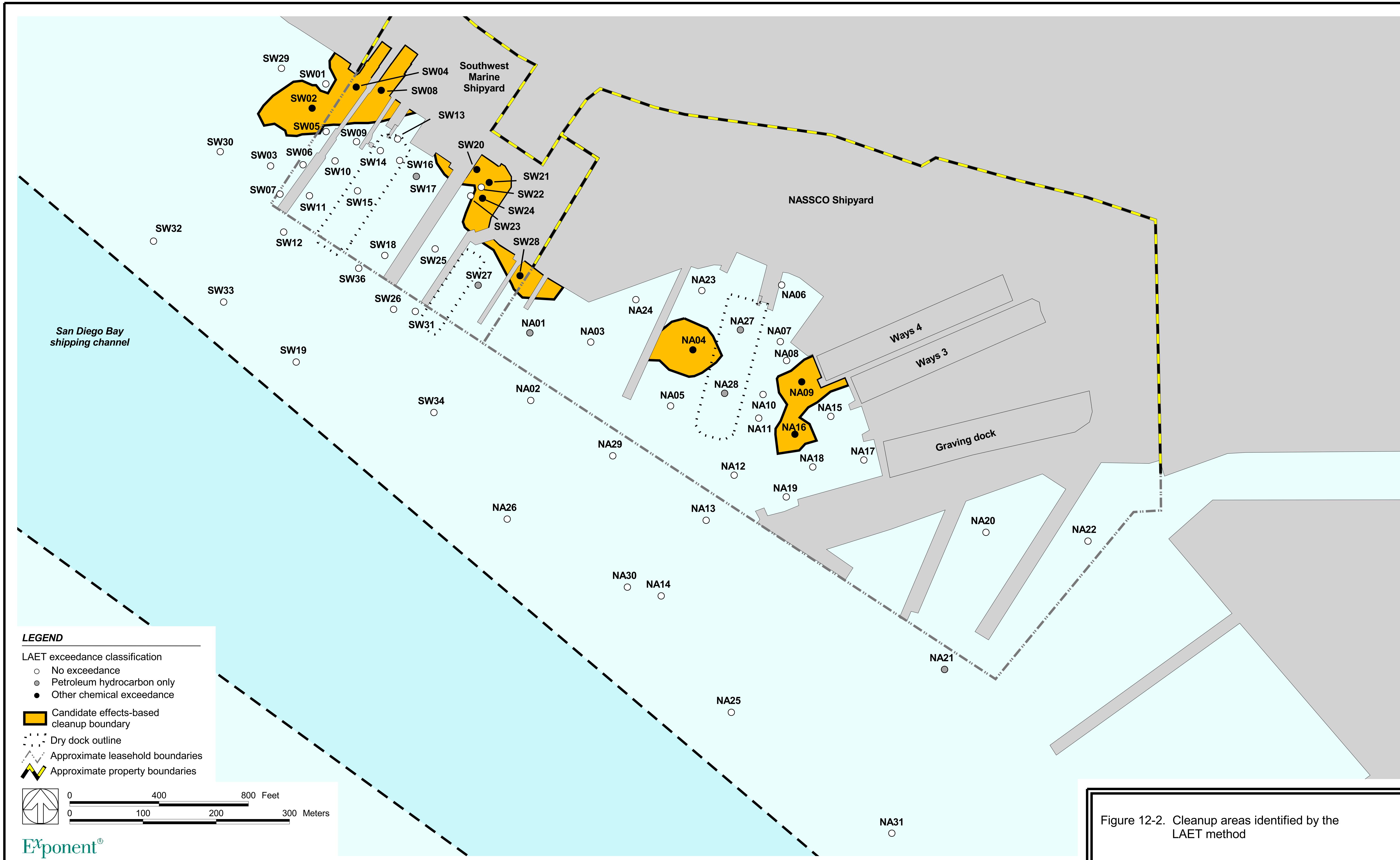
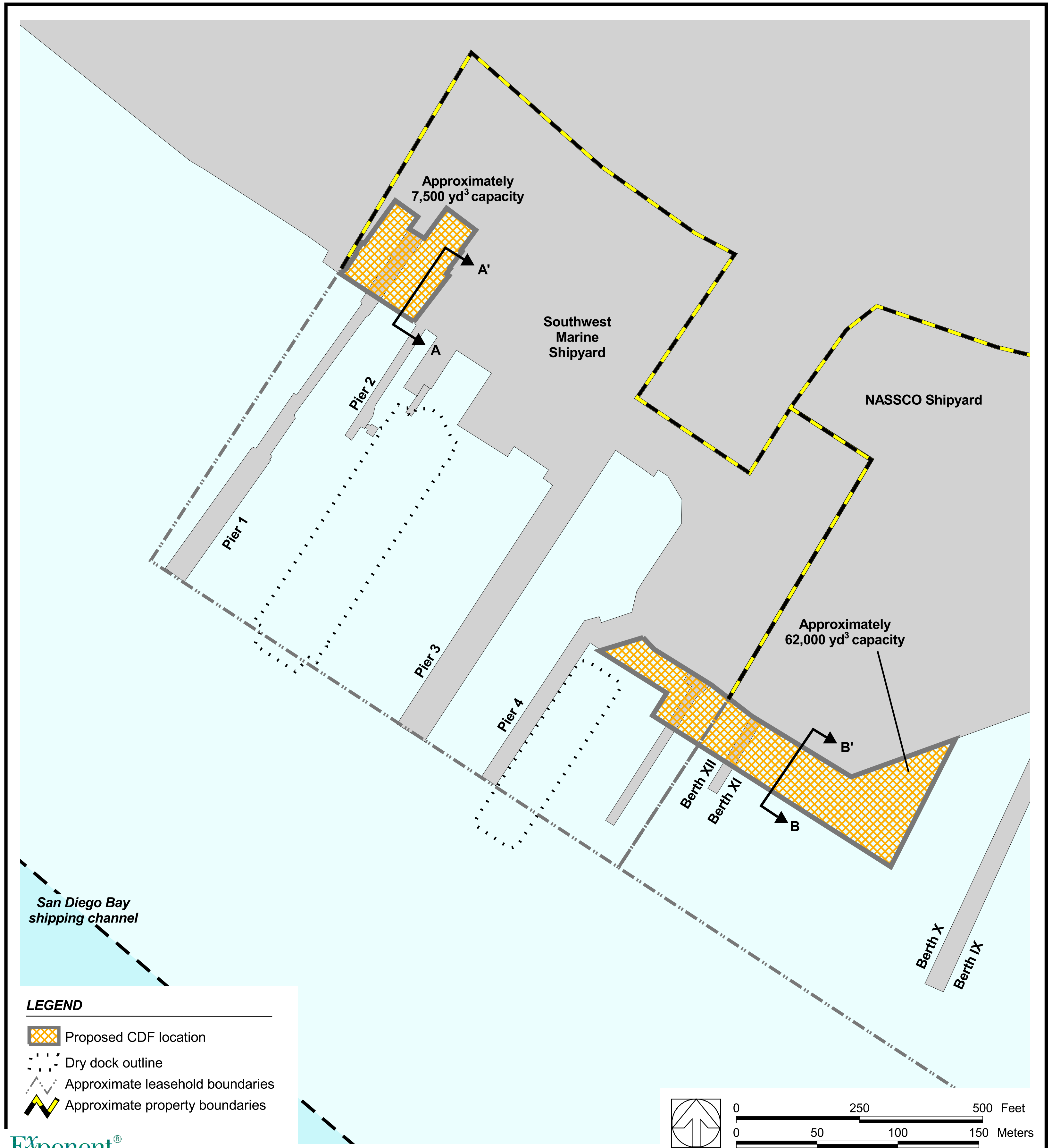


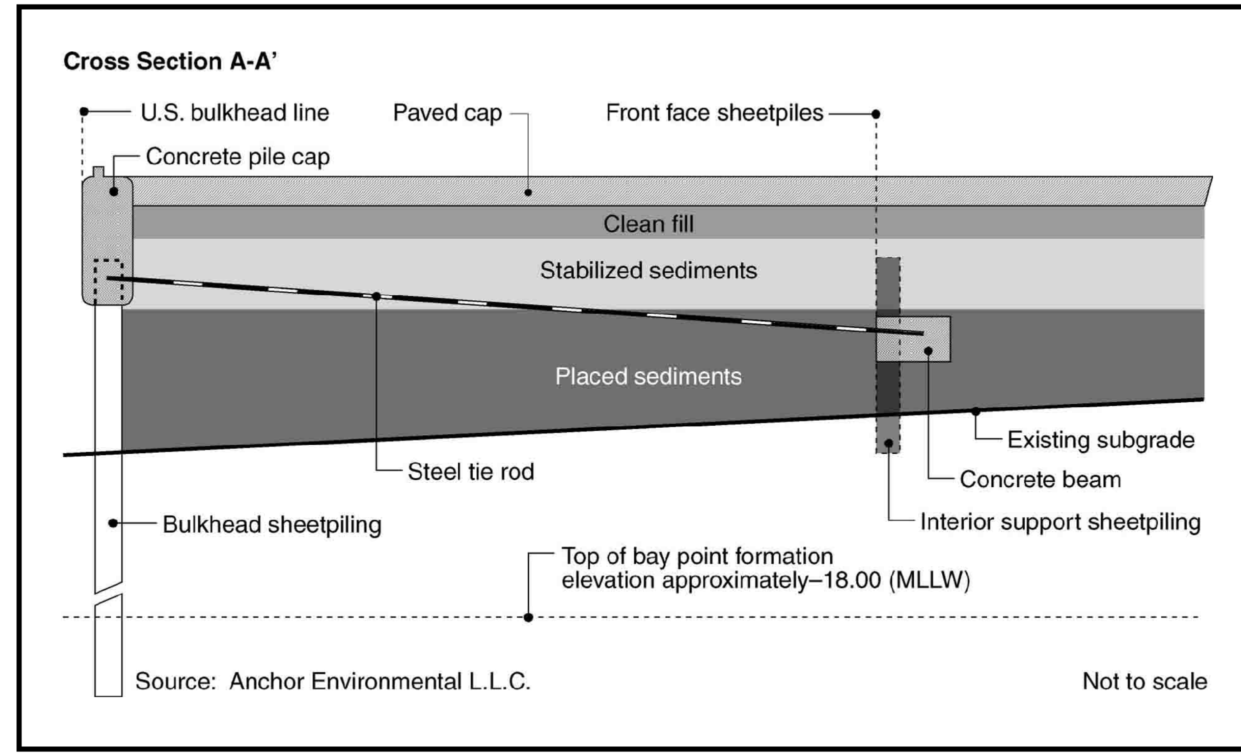
Figure 12-2. Cleanup areas identified by the LAET method

8601718/8601731.002 1201 | Oct 08, 2003 | fig 12-2 LAET exceed view | Fig 12-2 laet exceed layout | g:/swm_nassco/projects/shipyards_rpt.apr

Figure 16-1. Regional map of offsite landfill disposal location

Exponent®





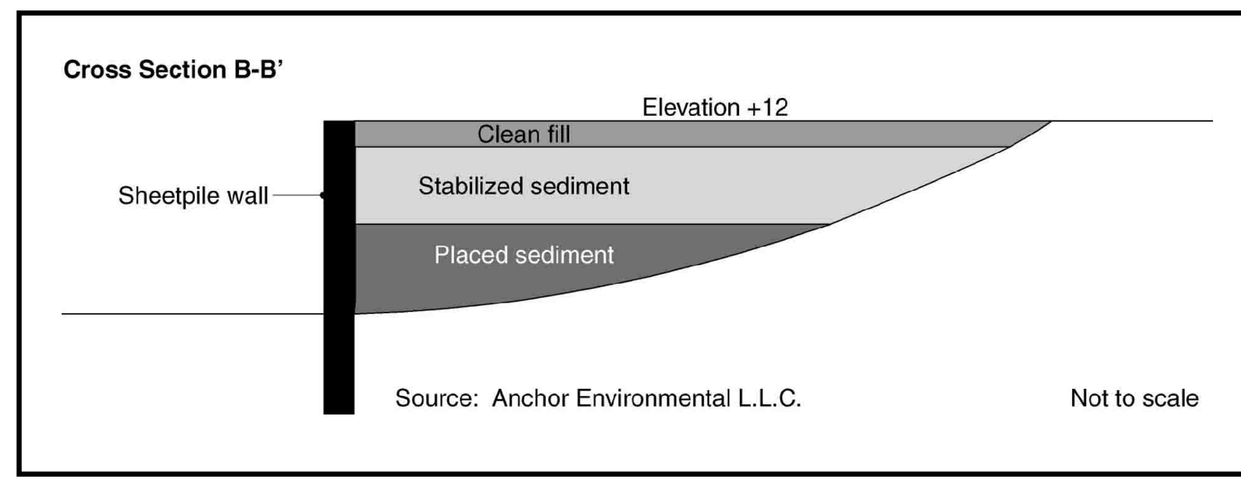


Figure 16-2. Location and conceptual design details of CDFs

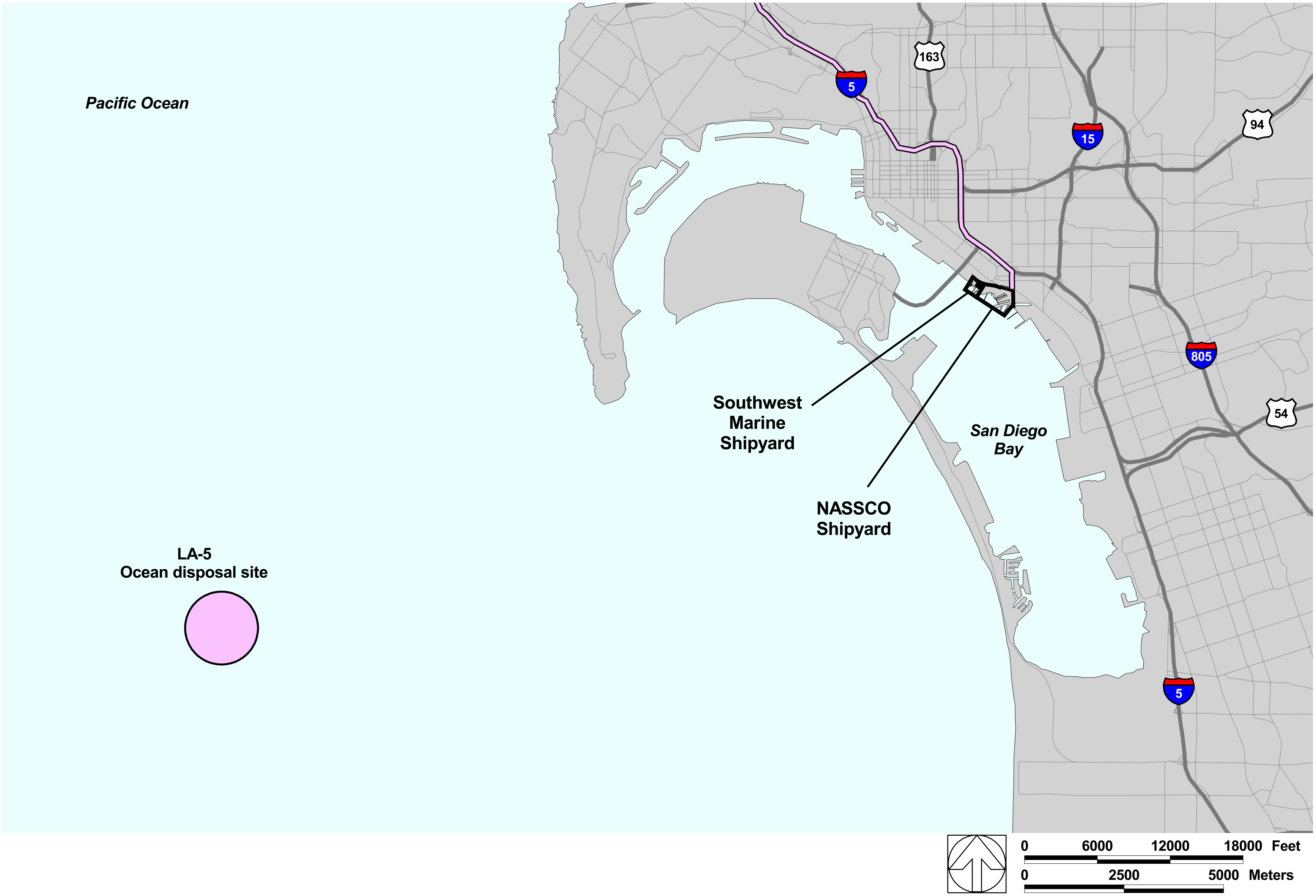


Figure 16-3. LA-5 offsite ocean disposal location

Exponent®

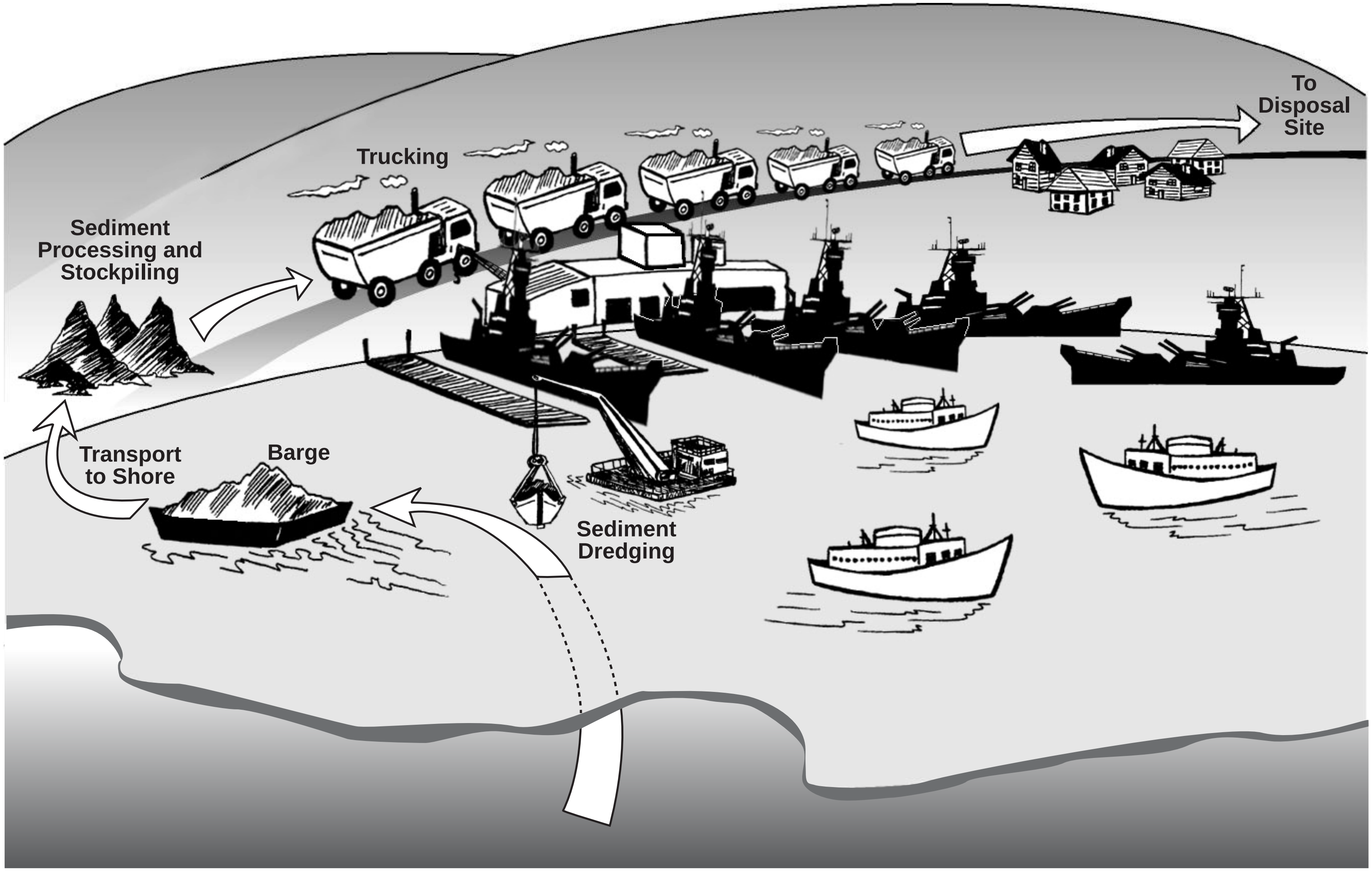


Figure 17-1. Conceptual model of Alternative B1

8601718.002 1201 - 8601731.002 1201 10/08/03 WA

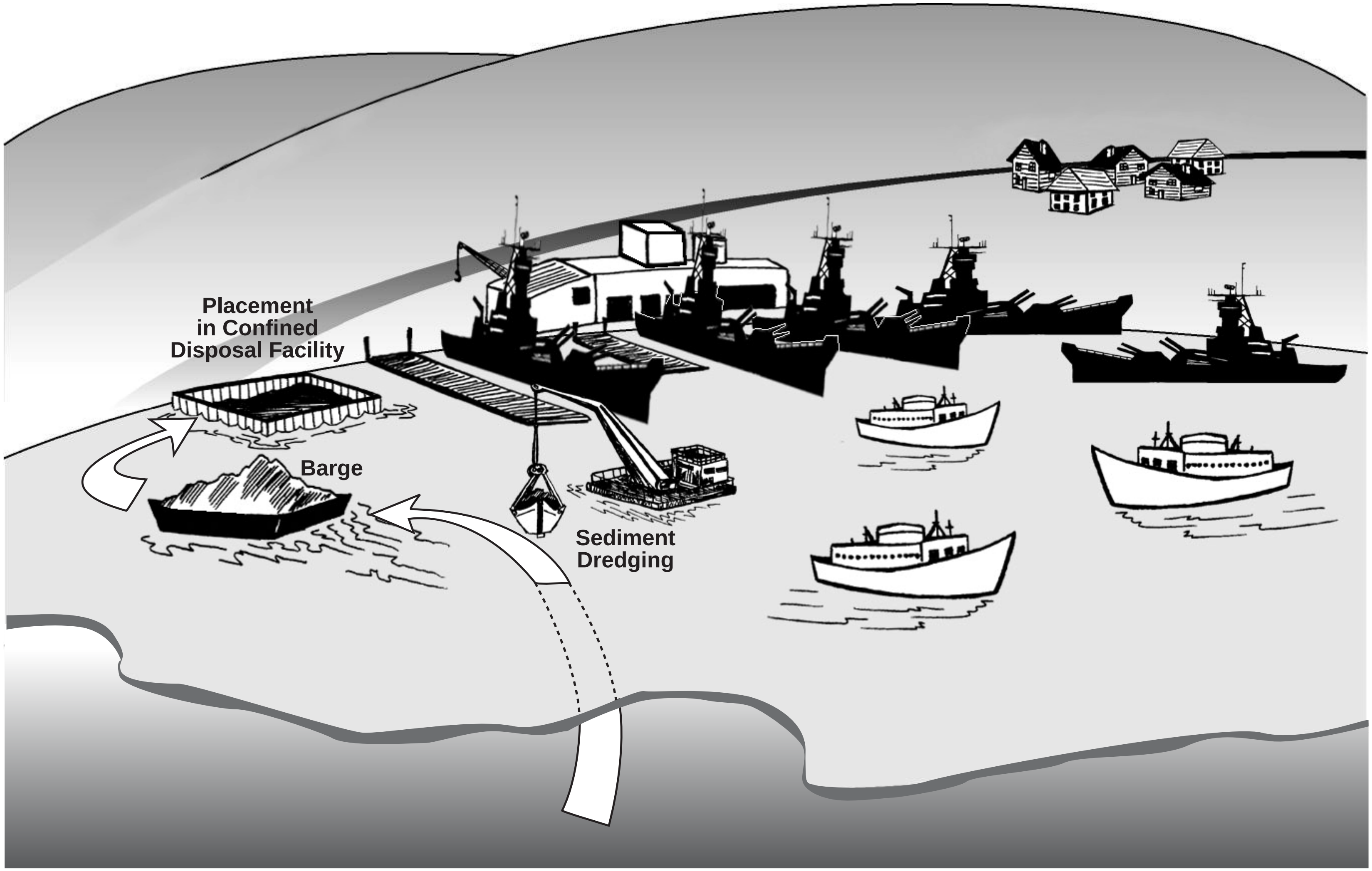


Figure 17-2. Conceptual model of Alternative B2

8601718.002 1201 - 8601731.002 1201 10/08/03 WA

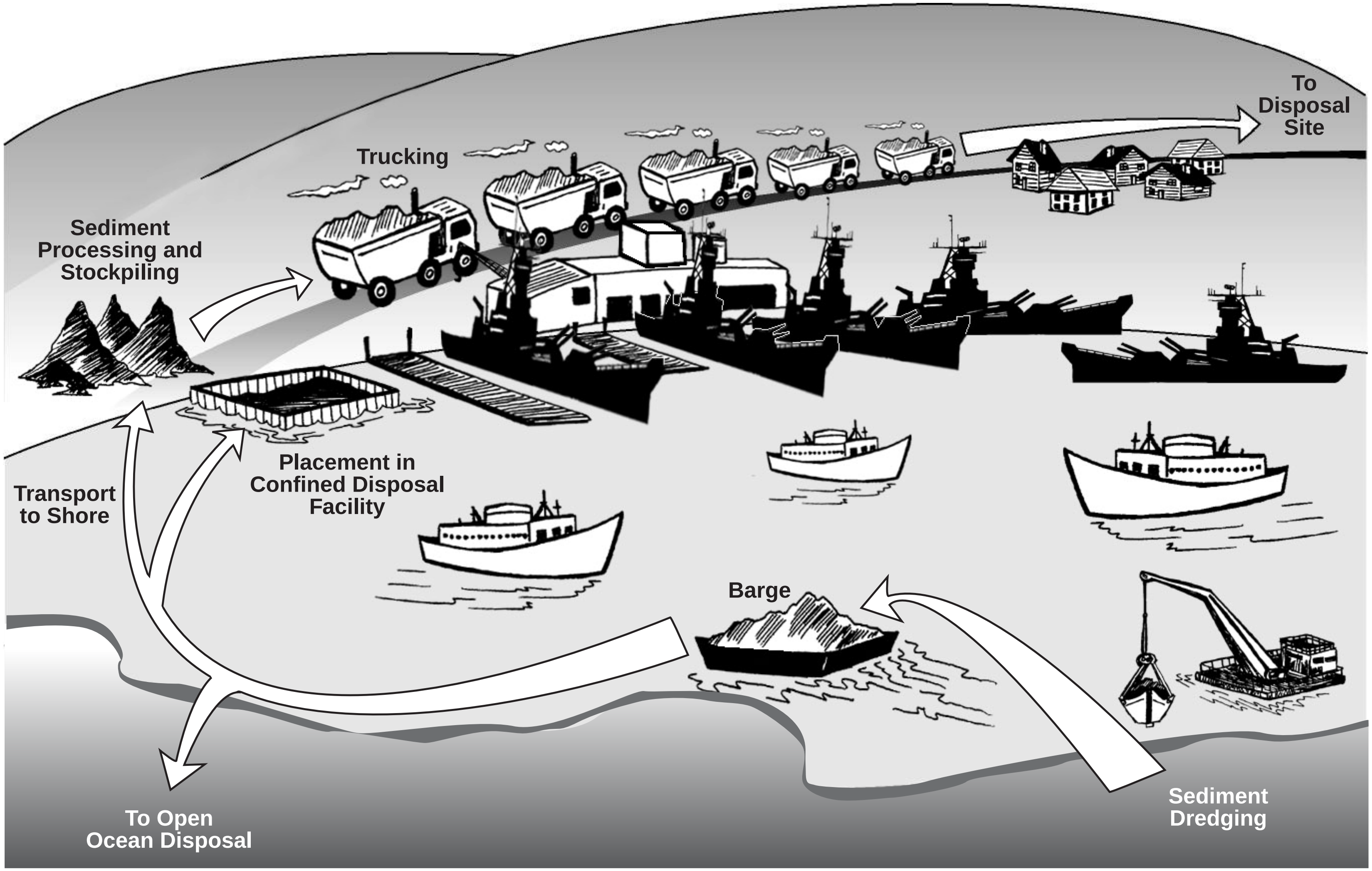


Figure 17-3. Conceptual model of Alternative C

8601718.002 1201 - 8601731.002 1201 10/08/03 WA

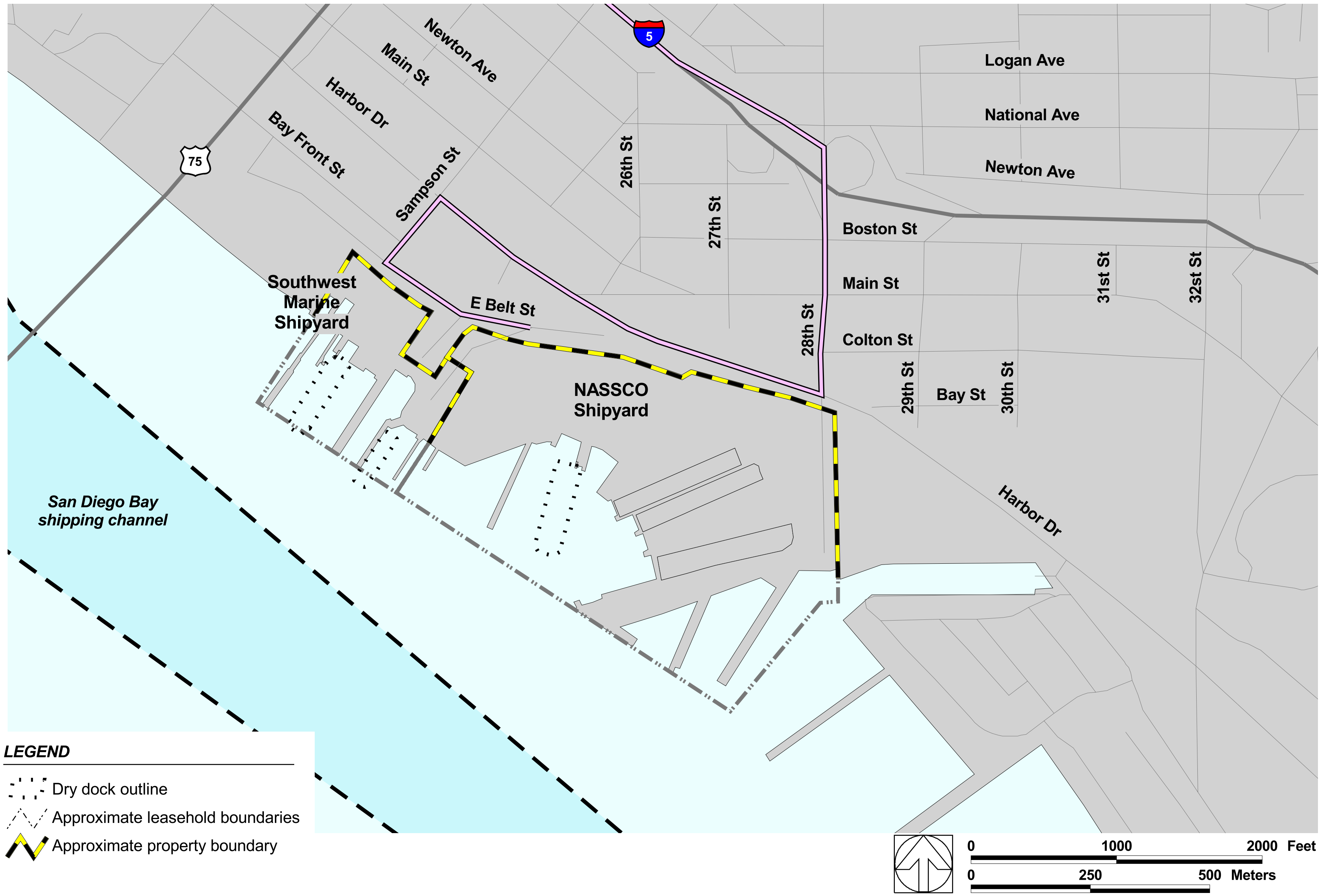


Figure 18-1. Local street network offsite disposal route

Exponent®

8601718/1731.002 1201 | Oct 07, 2003 | fig 18-1 street view | Fig 18-1 street layout | g:\swm_nassco\projects\shipyards_rpt.apr

Tables

Table 2-1. Analytes measured in Phase 1 and Phase 2 samples

Chemical Analyses	Phase 1		Phase 2						
	Sediment Chemistry	*Macoma* (bioaccum-ulation) Tissue	Pore Water	Surface Sediment Associated with Pore Water Samples	Surface Sediment (Bight '98 Stations)	Subsurface Sediment Cores	Additional Surface Sediment	Fish and Invertebrate Tissue	Eelgrass
Conventional Wet Chemistry									
Total organic carbon	X			X	X	X	X		
Grain size distribution (sand, silt, clay)	X			X	X	X	X		
Solids	X	X		X	X	X	X	X	X
Lipids		X						X	
Dissolved organic carbon			X						
Metals									
Arsenic	X	X	X	X	X	X	X	X	X
Cadmium	X	X	X	X	X	X	X	X	X
Chromium	X	X	X	X	X	X	X	X	X
Hexavalent chromium			X	X					
Copper	X	X	X	X	X	X	X	X	X
Lead	X	X	X	X	X	X	X	X	X
Mercury	X	X	X	X	X	X	X	X	X
Nickel	X	X	X	X	X	X	X	X	X
Selenium	X	X	X	X	X	X	X	X	X
Silver	X	X	X	X	X	X	X	X	X
Zinc	X	X	X	X	X	X	X	X	X
Organometallic Compounds									
Butyltin	X	X	X	X	X	X	X	X	X
Dibutyltin	X	X	X	X	X	X	X	X	X
Tributyltin	X	X	X	X	X	X	X	X	X
Tetrabutyltin	X	X	X	X	X	X	X	X	X
Polycyclic Aromatic Hydrocarbons									
Naphthalene	X	X	X	X	X	X	X	X[a]	X
2-Methylnaphthalene	X	X	X	X	X	X	X	X[a]	X
Acenaphthylene	X	X	X	X	X	X	X	X[a]	X
Acenaphthene	X	X	X	X	X	X	X	X[a]	X
Fluorene	X	X	X	X	X	X	X	X[a]	X

Table 2-1. (cont.)

	Phase 1		Phase 2						
Chemical Analyses	Sediment Chemistry	*Macoma* (bioaccum-ulation) Tissue	Pore Water	Surface Sediment Associated with Pore Water Samples	Surface Sediment (Bight '98 Stations)	Subsurface Sediment Cores	Additional Surface Sediment	Fish and Invertebrate Tissue	Eelgrass
Phenanthrene	X	X	X	X	X	X	X	X[a]	X
Anthracene	X	X	X	X	X	X	X	X[a]	X
Fluoranthene	X	X	X	X	X	X	X	X[a]	X
Pyrene	X	X	X	X	X	X	X	X[a]	X
Benz[a]anthracene	X	X	X	X	X	X	X	X[a]	X
Chrysene	X	X	X	X	X	X	X	X[a]	X
Benzo[b]fluoranthene	X	X	X	X	X	X	X	X[a]	X
Benzo[k]fluoranthene	X	X	X	X	X	X	X	X[a]	X
Benz[a]pyrene	X	X	X	X	X	X	X	X[a]	X
Indeno[1,2,3-cd]pyrene	X	X	X	X	X	X	X	X[a]	X
Dibenz[a,h]anthracene	X	X	X	X	X	X	X	X[a]	X
Benzo[ghi]perylene	X	X	X	X	X	X	X	X[a]	X
Alkylated PAH compounds[b]			X	X		X			
Pesticides									
Organophosphate pesticides							X[c]		
Organochlorine pesticides							X[c]		
Petroleum Hydrocarbons									
Gasoline-range organics	X				X	X	X		
Diesel-range organics	X				X	X	X		
Residual-range organics	X				X	X	X		
Polychlorinated Biphenyls									
Selected polychlorinated biphenyl congeners[d]	X	X			X	X	X	X	X
Aroclor® 1016	X	X	X	X	X	X	X	X	X
Aroclor® 1221	X	X	X	X	X	X	X	X	X
Aroclor® 1232	X	X	X	X	X	X	X	X	X
Aroclor® 1242	X	X	X	X	X	X	X	X	X

Table 2-1. (cont.)

Chemical Analyses	Phase 1: Sediment Chemistry	Phase 1: *Macoma* (bioaccum-ulation) Tissue	Phase 2: Pore Water	Phase 2: Surface Sediment Associated with Pore Water Samples	Phase 2: Surface Sediment (Bight '98 Stations)	Phase 2: Subsurface Sediment Cores	Phase 2: Additional Surface Sediment	Phase 2: Fish and Invertebrate Tissue	Phase 2: Eelgrass
Aroclor® 1248	X	X	X	X	X	X	X	X	X
Aroclor® 1254	X	X	X	X	X	X	X	X	X
Aroclor® 1260	X	X	X	X	X	X	X	X	X
Aroclor® 1268	X	X	X	X	X	X	X	X	X
Polychlorinated Terphenyls									
Aroclor® 5032	X	X			X	X	X		
Aroclor® 5442	X	X			X	X	X		
Aroclor® 5460	X	X			X	X	X		

Note: PAH - polycyclic aromatic hydrocarbon

[a] Only invertebrate tissue was analyzed for PAH.

[b] Alkylated PAH was measured in the upper horizon of some cores and in pore water and associated sediment.

[c] Pesticides were measured in selected surface sediment samples.

[d] IUPAC congeners 18, 28, 37, 44, 49, 52, 66, 70, 74, 77, 81, 87, 99, 101, 105, 110, 114, 118, 119, 123, 126, 128, 138, 149, 151, 153, 156, 157, 158, 167, 168, 169, 170, 177, 180, 183, 187, 189, 194, 201, and 206, and total homologs for each chlorination level.

Table 2-2. Summary of analyses by station

Station	Coordinates[a] Latitude	Longitude	Phase 1 Triad Analyses[b]	Phase 1 Additional Surface Sediment	Phase 1 Bioaccum-ulation	Phase 2 Core for Chemical Analysis	Phase 2 Pore Water	Phase 2 Additional Surface Sediment	Phase 2 Core for Engineering Properties
NASSCO									
NA01	3616867.150000	486618.000000	X			X			
NA02	3616775.020000	486619.220000		X		X			
NA03	3616854.678703	486700.993722	X						
NA04	3616843.990000	486840.440000	X			X	X	X[c]	
NA05	3616767.512513	486809.931465	X						
NA06	3616932.510000	486961.610000	X		X	X	X		X
NA07	3616855.259861	486959.722777	X[d]						
NA08	3616829.389691	486968.273321		X					
NA09	3616800.390000	486988.960000	X			X			X
NA10	3616783.096101	486936.176432		X					
NA11	3616750.797778	486930.303333	X		X			X[c]	
NA12	3616672.986217	486896.831631	X		X				
NA13	3616611.410000	486858.480000		X		X	X		X
NA14	3616508.047784	486797.087827		X					
NA15	3616753.183215	487028.646327	X						
NA16	3616728.900000	486979.600000	X			X	X		
NA17	3616693.610000	487073.710000	X			X	X		X
NA18	3616684.027819	487004.073697		X					
NA19	3616643.220000	486967.900000	X			X			
NA20	3616594.920000	487240.400000	X		X	X			
NA21	3616407.690000	487183.990000		X		X			
NA22	3616582.832500	487379.712500	X					X[c]	
NA23	3616925.030000	486852.600000				X		X	
NA24	3616912.580000	486762.720000				X		X	X
NA25	3616349.260000	486892.940000				X		X	
NA26	3616612.940000	486587.140000				X		X	
NA27	3616871.251559	486905.328588						X	
NA28	3616784.712792	486883.693896						X	
NA29	3616699.320000	486731.150000				X		X	
NA30	3616520.060000	486751.000000				X		X	
NA31	3616184.210000	487111.930000				X		X	

Table 2-2. (cont.)

Station	Coordinates[a]		Phase 1			Phase 2			
	Latitude	Longitude	Triad Analyses[b]	Additional Surface Sediment	Bioaccum-ulation	Core for Chemical Analysis	Pore Water	Additional Surface Sediment	Core for Engineering Properties
Southwest Marine									
SW01	3617206.990000	486339.470000		X		X	X		X
SW02	3617173.880000	486320.790000	X			X	X		
SW03	3617095.051914	486264.049842	X						
SW04	3617202.830000	486380.920000	X[d]		X	X	X	X[c]	
SW05	3617141.991289	486339.873319		X					
SW06	3617096.656107	486308.430201		X					
SW07	3617056.615892	486276.873082		X					
SW08	3617198.370000	486415.190000	X		X	X	X		
SW09	3617128.147179	486381.270040	X						
SW10	3617101.970000	486352.020000		X		X			X
SW11	3617054.405921	486317.050697	X						
SW12	3617004.710000	486281.940000		X		X	X		
SW13	3617131.839371	486437.518825	X		X				
SW14	3617115.959411	486413.953396		X					
SW15	3617061.139224	486382.842764	X						
SW16	3617102.528070	486440.262208		X					
SW17	3617080.840000	486463.100000	X			X			X
SW18	3616972.897179	486420.053694	X						
SW19	3616827.460000	486299.010000		X		X			
SW20	3617090.190000	486545.510000		X		X			
SW21	3617072.473283	486562.393409	X		X				
SW22	3617065.955876	486551.644511	X						
SW23	3617054.105245	486537.339936	X						
SW24	3617050.990000	486553.400000		X		X	X		X
SW25	3616981.930000	486488.740000	X			X	X		
SW26	3616899.257878	486431.954162		X					
SW27	3616932.220000	486547.400000	X			X			
SW28	3616945.190000	486604.420000		X	X	X	X		
SW29	3617228.400000	486278.860000				X		X	
SW30	3617114.480000	486195.450000				X		X	X
SW31	3616896.510000	486461.560000				X		X	X
SW32	3616992.440000	486104.400000				X		X	
SW33	3616909.220000	486200.080000				X		X	
SW34	3616758.500000	486487.120000				X		X	
SW36	3616955.330000	486384.480000				X		X	

Table 2-2. (cont.)

Station	Coordinates[a] Latitude	Longitude	Phase 1 Triad Analyses[b]	Additional Surface Sediment	Bioaccum-ulation	Phase 2 Core for Chemical Analysis	Pore Water	Additional Surface Sediment	Core for Engineering Properties
Reference									
2229	3619035.560536	483501.910215						X	
2230	3618324.650116	483255.473513						X	
2231	3617448.642000	485325.876000	X		X		X	X	
2240	3614441.124194	485552.428884						X	
2241	3614741.868181	487203.077910						X	
2243	3614105.548000	486625.544000	X		X		X	X	
2244	3613571.802548	487639.180461						X	
2265	3616251.802897	486847.215393						X	
2433	3620528.253988	480397.853986	X		X		X	X	
2435	3619330.202811	479108.531823						X	
2440	3620092.082000	483620.208000	X		X		X	X	
2441	3617113.053991	477860.015961	X		X		X	X	

Note: PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl
TBT - tributyltin

[a] Universal Transverse Mercator Zone 18, North American Datum 1983.

[b] Surface sediment chemistry; amphipod, echinoderm, and bivalve toxicity tests; and benthic macroinvertebrates.

[c] Organophosphate pesticide analysis only.

[d] Includes serial dilution toxicity test.

Table 2-3. Relative effort of sediment investigations

			Number of Stations										
Geographic Location	Acres	Study	Amphipod Bioassay	Sediment Chemistry	Benthic Community	Echinoderm Bioassay	Bivalve Bioassay	Bioaccum-ulation Test	Lobster Tissue Chemistry	Mussel Tissue Chemistry	Fish Tissue Chemistry	Fish Histo-pathology	Fish Bile
NASSCO and Southwest Marine[a]	143	Exponent	30	66	30	30	30	10	2	2	4	4	4
Chollas and Paleta Creeks TMDL[b]		Navy 2001	17	31	31	17							
San Diego Bay	11,231	Bight '98	46	46	46								
San Diego Bay	11,231	BPTCP		158	22								
Ventura Harbor	154	Bight '98	1	1									
Channel Islands Harbor	148	Bight '98	4	4									
Marina del Rey	417	Bight '98	7	7									
San Pedro Bay	12,444	Bight '98											
Anaheim Bay	604	Bight '98	3	3									
Newport Bay	1,202	Bight '98	11	11									
Mission Bay	2,315	Bight '98	3	3									
Mission Bay	2,315	BPTCP			3								
Dana Point Harbor	170	Bight '98	3	3									
Los Angeles Harbor	7,000	Bight '98	36	36									

Note: BPTCP - Bay Protection and Toxic Cleanup Program
TMDL - total maximum daily load

[a] Includes areas out to the ship channel; counts of samples do not include reference areas.

Table 2-4. Study elements and beneficial uses

Study Element	Benefical Uses		
	Aquatic Life	Aquatic-Dependent Wildlife	Human Health
Amphipod survival toxicity test	X		
Echinoderm fertilization toxicity test	X		
Bivalve development toxicity test	X		
Benthic macroinvertebrate community assessment	X		
Eelgrass distribution and chemical analyses	X	X	
Lobster tissue chemical analyses			X
Mussel tissue chemical analyses		X	
Fish tissue chemical analyses		X	X
Fish histopathology		X	
Fish bile analyses		X	

Table 3-1. Final reference pool samples

Investigation	Station
Bight '98	2231
Bight '98	2233
Bight '98	2238
Bight '98	2240
Bight '98	2241
Bight '98	2242
Bight '98	2243
Bight '98	2244
Bight '98	2247
Bight '98	2252
Bight '98	2256
Bight '98	2257
Bight '98	2265
Bight '98	2433
Bight '98	2435
Bight '98	2436
Bight '98	2440
U.S. Navy 2001 Chollas/Paleta	2238[a]
U.S. Navy 2001 Chollas/Paleta	2433
Shipyards Phase 1	2441
Shipyards Phase 1	2433
Shipyards Phase 1	2243[a]

[a]Chemistry and toxicity data only.

Table 4-1. Existence of vertical gradients of grain size

Station	Vertical Gradient of Grain Size?
NA01	No
NA02	Yes
NA04	No
NA06	Yes
NA09	No
NA13	Yes
NA16	No
NA17	Yes
NA19	Yes
NA20	No
NA21	No[a]
NA23	Yes
NA24	No[a]
NA25	Yes
NA26	Yes
NA29	Yes
NA30	Yes
NA31	Yes
SW01	No[a]
SW02	No
SW04	No
SW08	Yes
SW10	No[a]
SW12	Yes
SW17	Yes
SW19	Yes
SW20	No
SW24	No
SW25	No
SW27	Yes
SW28	No
SW29	Yes
SW30	No
SW31	Yes
SW33	Yes
SW34	Yes
SW36	Yes

[a]A decrease in percent fines appears in the lowest horizons of the core, but not in the upper horizons.

Table 4-2. Sediment cleanup levels and other benchmarks

Chemical[a]	Units	Final Reference Pool Sediment 95%UPL
Arsenic	mg/kg	9
Cadmium	mg/kg	0.29
Chromium	mg/kg	57
Copper	mg/kg	120
Lead	mg/kg	48
Mercury	mg/kg	0.56
Nickel	mg/kg	17
Selenium	mg/kg	0.72
Silver	mg/kg	1.0
Zinc	mg/kg	210
Dibutyltin	µg/kg	15
Monobutyltin	µg/kg	6.9
Tributyltin	µg/kg	5.1
Tetrabutyltin	µg/kg	1.7
HPAH	µg/kg	340
Total PCB congeners	µg/kg	36
Polychlorinated terphenyls	µg/kg	170

Note: 95%UPL - 95 percent upper prediction limit for a single future observation
HPAH - high-molecular-weight polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

[a] All values are reported on a dry weight basis.

Table 4-3. Molar concentrations of AVS and SEM

Station	SEM (mmol/kg)								
	Cadmium	Copper	Lead	Mercury	Nickel	Zinc	Sum of Metals	AVS (mmol/kg)	Excess Metals (mmol/kg)
NASSCO									
NA01	0.00089	3.32	0.373	0.0009	0.141	4.18	8.01	2.78	5.2
NA01	0.00089	3.40	0.392	0.0008	0.141	4.13	8.07	2.90	5.2
NA02	0.00089	2.52	0.325	0.0010	0.131	3.70	6.68	0.75	5.9
NA03	0.00089	3.48	0.397	0.0009	0.138	4.25	8.27	3.68	4.6
NA04	0.00089	3.79	0.396	0.0009	0.170	4.62	8.98	0.72	8.3
NA05	0.00178	2.33	0.260	0.0005	0.097	3.06	5.75	0.14 *J*	5.6
NA06	0.00445	5.08	0.462	0.0009	0.158 *J*	4.94	10.65	6.49	4.2
NA07	0.00445	3.87	0.447	0.0012	0.152 *J*	4.88	9.36	2.59 *J*	6.8
NA07	0.00445	4.09	0.437	0.0011	0.155 *J*	4.64	9.32	3.34 *J*	6.0
NA08	0.00356	4.15	0.349	0.0004	0.143	5.05	9.70	1.68 *J*	8.0
NA09	0.00356	4.06	0.352	0.0002	0.148	5.14	9.70	19.25 *J*	-9.5
NA10	0.00178	2.11	0.216	0.0004	0.087	2.71	5.12	0.32 *J*	4.8
NA11	0.00089 *U*	2.75	0.294	0.0009	0.305 *J*	3.72	7.07	0.56	6.5
NA12	0.00089	2.39	0.251	0.0008	0.121	3.30	6.07	0.56	5.5
NA13	0.00089	2.60	0.344	0.0006	0.148	4.21	7.30	3.06	4.2
NA14	0.00178	1.76	0.243	0.0003	0.099	2.92	5.03	1.83 *J*	3.2
NA15	0.00089	4.04	0.356	0.0008	0.143	4.83	9.38	0.84	8.5
NA16	0.00089	3.92	0.375	0.0011	0.146	4.57	9.02	3.87	5.1
NA17	0.00178	8.42	0.512	0.0004	0.170	11.82	20.93	8.86	12.1
NA18	0.00267	3.64	0.399	0.0005	0.126	6.55	10.71	0.83 *J*	9.9
NA19	0.00178	4.47	0.410	0.0007	0.146	7.79	12.82	3.93	8.9
NA20	0.00267	1.42	0.251	0.0001	0.072	3.15	4.90	3.84	1.1
NA21	0.00356	2.19	0.307	0.0001	0.109	3.53	6.14	2.99 *J*	3.1
NA22	0.00356	1.86	0.281	0.0000 *U*	0.080	3.29	5.51	5.52 *J*	-0.01
Southwest Marine									
SW01	0.00445	5.51	0.502	0.0005	0.434 *J*		6.45	1.78	4.7
SW02	0.02224	6.00	0.666	0.0002	1.044 *J*	8.61	16.34	4.52	11.8
SW02	0.02758	5.27	0.681	0.0001	0.598 *J*	7.69	14.27	10.11	4.2
SW03	0.00712	1.48	0.285	0.0023	0.196 *J*	3.26	5.23	4.71	0.5
SW04	0.02669	16.68	2.548	0.0002	0.368 *J*	94.84	114.47	1.08	113.4
SW05	0.00534	3.10	0.362	0.0005	0.157 *J*	5.05	8.67	5.90	2.8
SW06	0.00623	2.30	0.294	0.0005	0.184 *J*	4.47	7.25	4.87	2.4
SW07	0.00089 *U*	2.34	0.247	0.0007	0.806 *J*	3.07	6.47	3.21	3.3
SW08	0.01157	13.63	1.139	0.0012	0.216 *J*	12.83	27.83	48.97 *J*	-21.1
SW09	0.00712	6.77	0.753	0.0004	0.175 *J*		7.70	4.99	2.7

Table 4-3. (cont.)

Station	SEM (mmol/kg) Cadmium	Copper	Lead	Mercury	Nickel	Zinc	Sum of metals	AVS (mmol/kg)	Excess Metals (mmol/kg)
SW10	0.00623	2.25	0.318	0.0002	0.136 *J*	5.98	8.69	18.93	-10.2
SW11	0.00267	2.86	0.298	0.0002	0.135	3.84	7.14	6.55 *J*	0.6
SW12	0.00267	1.94	0.262	0.0008	0.100 *J*	2.72	5.02	0.56	4.5
SW13	0.00801	19.36	0.460	0.0003	0.330 *J*	15.91	36.07	6.74	29.3
SW14	0.00445	4.85	0.410	0.0009	0.153 *J*	4.79	10.20	14.69	-4.5
SW15	0.00623	3.59	0.401	0.0011	0.162 *J*	4.54	8.70	14.85	-6.1
SW16	0.00979	7.93	0.507	0.0005	0.479 *J*	6.82	15.75	8.64	7.1
SW17	0.00356	5.15	0.414	0.0004	0.608	5.48	11.65	1.32 *J*	10.3
SW18	0.00356	3.54	0.391	0.0003	1.589	4.74	10.27	0.72 *J*	9.6
SW19	0.00267	1.43	0.240	0.0006	0.104 *J*	2.36	4.13	1.55	2.6
SW20	0.00445	4.15	0.444	0.0002	0.124	6.79	11.52	11.63 *J*	-0.1
SW21	0.00445	3.93	0.460	0.0010	0.445 *J*	4.83	9.68	0.55	9.1
SW22	0.00267	4.64	0.407	0.0003	0.136	5.17	10.36	1.93 *J*	8.4
SW23	0.00356	4.25	0.391	0.0003	0.133	5.06	9.84	4.93 *J*	4.9
SW24	0.00534	3.82	0.396	0.0007	0.143 *J*	4.38	8.74	1.86 *J*	6.9
SW25	0.00356	3.97	0.371	0.0002	0.448	5.11	9.90	2.97 *J*	6.9
SW26	0.00267	1.75	0.247	0.0010	0.095 *J*	2.34	4.43	0.08 *J*	4.4
SW27	0.00267	3.34	0.347	0.0002	0.129	4.28	8.10	5.99 *J*	2.1
SW28	0.00178	4.22	0.492	0.0003	0.141	5.19	10.04	13.97	-3.9
Reference									
2441	0.00178	0.49	0.052	0.0000	0.087	1.14	1.77	5.52	-3.8
2433	0.00178	0.53	0.067	0.0001	0.066	1.27	1.94	0.25	1.7
2440	0.00267	0.67	0.291	0.0000 *U*	0.058	1.59	2.61	2.19 *J*	0.4
2231	0.00267	1.26	0.208	0.0011	0.099 *J*	2.02	3.59	0.02 *UJ*	3.6
2243	0.00089	0.65	0.084	0.0001	0.045	1.39	2.17	0.27 *J*	1.9

Note: All measurements are on a dry weight basis.

AVS - acid-volatile sulfide
J - estimated
SEM - simultaneously extracted metals
U - undetected at the quantitation limit shown

Table 4-4. Stations selected for microprobe analyses

Station	Rationale
NA19	Relatively high concentrations of chromium, copper, lead, and zinc, relatively low concentrations of organics, and toxicity to bivalves
SW02	A high concentration of chromium (and organics), but no toxicity
SW04	High concentrations of copper, lead, and zinc (and organics) but no toxicity
SW27	Relatively high concentrations of chromium, copper, lead, and zinc, relatively low concentrations of organics, and toxicity to amphipods and bivalves

Table 4-5. Extended analyte list for PAHs

LPAH	HPAH	Additional Aromatic Hydrocarbons
Naphthalene[a]	Fluoranthene[a]	Dibenzothiophene
1-Methylnaphthalene	Pyrene[a]	C1-Dibenzothiophenes
2-Methylnaphthalene	C1-Fluoranthenes/pyrenes	C2-Dibenzothiophenes
C2-Naphthalenes	Benz[a]anthracene[a]	C3-Dibenzothiophenes
C3-Naphthalenes	Chrysene[a]	
C4-Naphthalenes	C1-Benz[a]anthracenes/chrysenes	
Acenaphthylene[a]	C2-Benz[a]anthracenes/chrysenes	
Acenaphthene[a]	C3-Benz[a]anthracenes/chrysenes	
Fluorene[a]	C4-Benz[a]anthracenes/chrysenes	
C1-Fluorenes	Benzo[b]fluoranthene[a]	
C2-Fluorenes	Benzo[k]fluoranthene[a]	
C3-Fluorenes	Perylene	
Phenanthrene[a]	Benzo[a]pyrene[a]	
Anthracene[a]	Benzo[e]pyrene	
C1-Phenanthrenes/anthracenes	Indeno-[1,2,3-cd]pyrene[a]	
C2-Phenanthrenes/anthracenes	Dibenz[a,h]anthracene[a]	
C3-Phenanthrenes/anthracenes	Benzo[ghi]perylene[a]	
C4-Phenanthrenes/anthracenes		

Note: EPA - U.S. Environmental Protection Agency
HPAH - high-molecular-weight polycyclic aromatic hydrocarbon
LPAH - low-molecular-weight polycyclic aromatic hydrocarbon

[a] EPA priority pollutant PAH.

Table 4-6. LPAHs as fraction of total PAHs in sediment samples

Sample Number	Survey Station	Date	Two- and Three-Ring PAHs[a] (μg/kg dry weight)	Total PAHs[b] (μg/kg dry weight)	Percentage of Two- and Three-Ring PAHs
Sediment Core Samples (0–2 ft)					
SD0065	NA19	09/03/02	310	5,167	6.0
SD0006	SW24	08/13/02	6,673	34,160	19.5
SD0033	SW08	08/28/02	1,894	35,644	5.3
Surface Sediment Samples (0–2 cm)					
SD0168	2231	11/06/02	79	1,117	7.1
SD0176	2243	11/07/02	51	342	14.9
SD0167	2433	11/06/02	82	1,013	8.1
SD0175	2440	11/07/02	179	2,135	8.4
SD0166A	2441	11/06/02	96	1,005	9.5
SD0179	NA01	11/07/02	635	11,115	5.7
SD0181	NA06	11/07/02	387	6,420	6.0
SD0183	NA13	11/08/02	216	4,286	5.0
SD0182	NA16	11/07/02	292	5,595	5.2
SD0184	NA17	11/08/02	204	3,911	5.2
SD0171	SW01	11/06/02	709	9,800	7.2
SD0169	SW01	11/06/02	731	9,725	7.5
SD0172	SW02	11/06/02	16,961	38,451	44.1
SD0170	SW04	11/06/02	2,678	28,908	9.3
SD0178	SW08	11/07/02	4,019	47,079	8.5
SD0173	SW24	11/06/02	2,868	82,818	3.5
SD0174	SW25	11/06/02	507	10,171	5.0
SD0177	SW28	11/07/02	2,446	27,076	9.0
SD0180	SW36	11/07/02	500	7,675	6.5

Note: Results for undetected analytes are included at one-half the reported quantitation limit.

LPAH - low-molecular-weight polycyclic aromatic hydrocarbon
PAH - polycyclic aromatic hydrocarbon

[a] Two- and three-ring PAHs include the priority pollutant LPAHs and C1-, C2-, and C3-alkylated naphthalenes; C1-phenanthrenes and 3,6-dimethylphenanthrene; and biphenyl as included by Zeng and Vista (1997, Table 1).

[b] Total PAHs include the priority pollutant PAHs; the additional two- and three-ring PAHs listed in footnote a; and benzo[e]pyrene and perylene as included by Zeng and Vista (1997, Table 1). The PAHs 2,3-benzofluorene and 9,10-diphenylanthracene were included by Zeng and Vista (1997), but were not included as analytes in this study and are not included in the total.

Table 5-1. California water quality criteria[a]

Compound	Concentration (μg/L)
Arsenic	36
Cadmium	9.3
Chromium(VI)	50
Copper	3.1
Lead	8.1
Nickel	8.2
Selenium	71
Zinc	81
Polychlorinated biphenyls	0.03

[a]Criterion continuous concentrations for marine waters. Concentrations are for dissolved forms of chemicals.

Table 5-2. Relationships between pore water and sediment

Chemical	Pore Water (pw) Units	Sediment (sed) Units	Proportional Relationship	SW02 Outlier		Prediction Equation	R-square
Arsenic	μg/L	mg/kg	no	no		no relationship	
Copper	μg/L	mg/kg	no	yes	pw =	$10.9 + 0.0343 \times \text{sed}$	0.73
Lead	μg/L	mg/kg	no	yes	pw =	$3.63 + 0.0450 \times \text{sed}$	0.75
Mercury	ng/L	mg/kg	no	yes	pw =	$[4.19 + 5.18 \times (\text{sed})^{1/2}]^2$	0.63
Nickel	μg/L	mg/kg	no	yes		no relationship	
Silver	μg/L	mg/kg	no	yes		no relationship	
Zinc	μg/L	mg/kg	no	yes	pw =	$10.6 + 1.049 \times (\text{sed})^{1/2}$	0.61
TBT	μg/L	μg/kg	yes	no	pw =	$[0.0676 + 0.0107 \times (\text{sed})^{1/2}]^2$	0.79
PCB homologs	ng/L	ng/g	no	yes	pw =	$[2.65 + 0.374 \times (\text{sed})^{1/2}]^2$	0.85

Note: PCB - polychlorinated biphenyl
TBT - tributyltin

Table 6-1. Amphipod survival results

Station	Batch	Amphipod Survival (percent) Replicate 1	Replicate 2	Replicate 3	Replicate 4	Replicate 5
Reference						
2441	640-2	95	100	95	85	100
2433	640-3	95	90	95	90	95
2440	640-3	95	100	95	100	95
2231	640-2	85	80	90	90	75
2243	640-3	90	90	95	95	75
NASSCO						
NA01	640-2	70	85	95	80	70
NA03	640-2	95	100	70	90	65
NA04	640-2	55	85	90	85	85
NA05	640-3	85	80	80	95	90
NA06	640-2	80	85	60	95	70
NA07	640-1	75	85	55	70	80
NA09	640-3	80	90	90	80	85
NA11	640-2	60	75	75	70	70
NA12	640-2	75	75	95	80	85
NA15	640-3	95	90	100	100	85
NA16	640-3	90	90	85	90	80
NA17	640-3	85	95	95	90	95
NA19	640-3	70	95	100	85	80
NA20	640-2	100	90	90	90	80
NA22	640-3	95	75	95	100	95
Southwest Marine						
SW02	640-2	95	90	90	75	90
SW03	640-2	95	85	95	85	100
SW04	640-1	75	95	100	100	95
SW08	640-2	95	95	85	90	90
SW09	640-2	85	95	85	100	75
SW11	640-3	70	85	75	70	75
SW13	640-2	85	90	95	95	95
SW15	640-2	100	90	90	80	100
SW17	640-3	85	90	95	95	95
SW18	640-3	75	95	40	80	70
SW21	640-2	85	90	90	95	95
SW22	640-3	85	90	90	85	85
SW23	640-3	80	100	90	85	85
SW25	640-3	90	80	85	70	90
SW27	640-3	60	65	95	65	70

Table 6-2. Echinoderm fertilization results

Station	Batch	Echinoderm Fertilization (percent)				
		Replicate 1	Replicate 2	Replicate 3	Replicate 4	Replicate 5
Reference						
2441	Batch 2	80	80	82	90	83
2433	Batch 2	83	69	73	68	71
2440	Batch 3	79	82	81	83	78
2231	Batch 1	96	92	92	99	87
2243	Batch 3	67	69	69	65	76
NASSCO						
NA01	Batch 2	78	77	84	75	80
NA03	Batch 2	78	84	74	80	70
NA04	Batch 2	80	77	85	79	82
NA05	Batch 3	75	74	63	78	67
NA06	Batch 1	99	94	97	99	93
NA07	Batch 1	99	93	91	95	97
NA09	Batch 3	69	70	76	73	83
NA11	Batch 1	93	95	97	93	91
NA12	Batch 2	86	86	85	72	82
NA15	Batch 2	81	86	78	81	78
NA16	Batch 2	76	85	70	80	73
NA17	Batch 2	77	83	82	81	80
NA19	Batch 2	63	74	57	65	70
NA20	Batch 2	66	81	72	70	72
NA22	Batch 3	83	84	80	85	83
Southwest Marine						
SW02	Batch 1	95	96	97	97	94
SW03	Batch 1	96	95	94	96	98
SW04	Batch 3	85	79	79	82	82
SW08	Batch 1	94	94	95	97	98
SW09	Batch 1	94	92	92	95	92
SW11	Batch 3	76	62	66	69	63
SW13	Batch 1	91	93	93	92	93
SW15	Batch 1	94	100	96	97	92
SW17	Batch 3	70	72	72	73	72
SW18	Batch 3	67	60	55	66	62
SW21	Batch 1	96	95	95	96	94
SW22	Batch 3	74	85	77	76	79
SW23	Batch 3	82	80	76	83	82
SW25	Batch 3	74	78	82	71	80
SW27	Batch 3	72	66	67	71	63

Table 6-3. Bivalve normality results

Station	Batch	Bivalve Combined Survival and Normality (percent)				
		Replicate 1	Replicate 2	Replicate 3	Replicate 4	Replicate 5
Reference						
2441	Batch 2	69	77	60	64	59
2433	Batch 2	24	58	66	39	47
2440	Batch 2	61	71	66	64	88
2231	Batch 1	88	86	80	77	80
2243	Batch 2	62	24	75	8	79
NASSCO						
NA01	Batch 2	44	6	10	80	77
NA03	Batch 2	85	90	67	84	90
NA04	Batch 2	60	77	83	80	71
NA05	Batch 2	92	79	82	80	84
NA06	Batch 1	62	38	65	91	86
NA07	Batch 1	81	82	93	57	91
NA09	Batch 2	5	0	1	0	0
NA11	Batch 1	90	84	84	35	79
NA12	Batch 2	65	0	0	0	2
NA15	Batch 2	75	89	74	88	84
NA16	Batch 2	1	12	0	0	3
NA17	Batch 2	66	80	77	47	79
NA19	Batch 2	0	0	0	0	8
NA20	Batch 1	71	65	65	81	89
NA22	Batch 2	0	2	0	7	0
Southwest Marine						
SW02	Batch 1	90	67	90	65	77
SW03	Batch 1	82	74	88	90	70
SW04	Batch 1	65	33	84	46	63
SW08	Batch 1	87	84	88	83	86
SW09	Batch 1	78	82	72	76	81
SW11	Batch 2	84	47	74	77	84
SW13	Batch 1	19	0	41	70	0
SW15	Batch 1	0	0	16	16	9
SW17	Batch 2	0	0	0	0	69
SW18	Batch 2	16	54	74	60	76
SW21	Batch 1	2	71	78	80	78
SW22	Batch 2	1	0	0	4	1
SW23	Batch 2	52	3	14	1	2
SW25	Batch 2	39	4	1	0	0
SW27	Batch 2	72	1	4	11	9

Table 6-4. Amphipod control data

	Amphipod Survival (percent)				
Batch	Replicate 1	Replicate 2	Replicate 3	Replicate 4	Replicate 5
640-1	100	95	100	100	100
640-2	100	100	100	100	100
640-3	100	95	90	100	100

Table 6-5. Echinoderm control data

Batch	Echinoderm Fertilization (percent) Replicate 1	Replicate 2	Replicate 3	Replicate 4	Replicate 5
Batch 1	96	91	90	98	94
Batch 1	94	95	97	93	92
Batch 2	93	91	92	93	92
Batch 2	93	89	93	94	91
Batch 3	83	81	82	82	77
Batch 3	69	75	71	63	70

Table 6-6. Bivalve control data

Batch	Control type	Toxicant[a]	Concen-tration (μg/L)	Bivalve Combined Survival and Normality (percent) Replicate 1	Replicate 2	Replicate 3	Replicate 4	Replicate 5
Batch 1	Water	Copper	0	94	93	91	93	90
Batch 1	Water	Copper	2	97	96	93	95	98
Batch 1	Water	Copper	5	99	92	94	98	91
Batch 1	Water	Copper	10	73	76	76	74	79
Batch 1	Water	Copper	20	0	0	0	0	0
Batch 1	Water	Copper	40	0	0	0	0	0
Batch 2	Water	Copper	0	83	90	84	92	90
Batch 2	Water	Copper	2	76	83	84	86	78
Batch 2	Water	Copper	5	91	90	88	92	87
Batch 2	Water	Copper	10	90	84	94	91	86
Batch 2	Water	Copper	20	3	5	6	3	2
Batch 2	Water	Copper	40	0	0	0	0	0
Batch 1	Water			94	93	91	93	90
Batch 2	Water			83	90	84	92	90
Batch 1	Sediment			85	86	81	88	87
Batch 2	Sediment			70	75	65	15	83
Batch 1	Sediment			77	79	71	75	81
Batch 2	Sediment			82	80	74	76	89

[a] A toxicant is used for positive controls only.

Table 6-7. Amphipod serial dilution results

			Amphipod Survival (percent)				
Station	Batch	Dilution[a]	Replicate 1	Replicate 2	Replicate 3	Replicate 4	Replicate 5
NASSCO							
NA07	640-1	10	95	100	95	100	100
NA07	640-1	20	100	100	90	100	100
NA07	640-1	30	100	90	95	100	85
NA07	640-1	40	90	95	90	100	100
NA07	640-1	50	80	95	75	100	75
NA07	640-1	60	95	70	45	95	90
NA07	640-1	70	95	70	95	90	90
NA07	640-1	80	85	65	85	85	90
NA07	640-1	90	95	65	65	85	80
NA07	640-1	100	75	85	55	70	80
Southwest Marine							
SW04	640-1	10	100	100	90	100	100
SW04	640-1	20	100	100	100	100	100
SW04	640-1	30	100	95	100	100	100
SW04	640-1	40	100	90	90	100	100
SW04	640-1	50	100	90	95	95	95
SW04	640-1	60	100	100	95	90	95
SW04	640-1	70	100	95	95	90	100
SW04	640-1	80	100	100	85	90	100
SW04	640-1	90	95	95	90	100	95
SW04	640-1	100	75	95	100	100	95

[a] Percent of original sample.

Table 8-1. Major taxa abundances at reference stations

Station	Replicate	Crustacea	Echinoderms	Molluscs	Polychaetes	Other	Total
2441	1	18	30	28	250	30	356
2441	2	24	16	32	358	115	545
2441	3	42	48	18	322	68	498
2441	4	18	22	20	422	46	528
2441	5	12	46	42	470	29	599
2433	1	30	2	26	370	52	480
2433	2	14	0	66	364	8	452
2433	3	30	2	26	312	8	378
2433	4	52	8	42	324	30	456
2433	5	48	0	78	292	14	432
2440	1	46	2	58	472	4	582
2440	2	74	0	54	436	5	569
2440	3	96	0	84	344	6	530
2440	4	50	2	80	468	10	610
2440	5	336	0	86	474	8	904
2231	1	3,630	22	28	358	14	4,052
2231	2	4,188	8	30	474	22	4,722
2231	3	5,792	28	20	326	21	6,187
2231	4	6,734	38	44	610	27	7,453
2231	5	7,858	44	50	756	39	8,747
2243	1	92	2	112	414	114	734
2243	2	112	10	128	1,078	100	1,428
2243	3	254	0	98	556	177	1,085
2243	4	22	4	20	252	53	351
2243	5	210	4	112	878	135	1,339

Table 8-2. Summary of relative abundances and relative numbers of taxa of major taxonomic groups at shipyard and reference stations[a]

	Relative Abundance (percent)				Relative Taxa Richness (percent)			
Station	Polychaeta	Mollusca	Crustacea	Other	Polychaeta	Mollusca	Crustacea	Other
NASSCO								
NA01	45	15	32	7	34	14	40	12
NA03	53	20	24	3	35	17	35	14
NA04	68	18	10	4	48	10	32	10
NA05	74	15	9	2	36	15	36	13
NA06	72	8	14	6	38	13	33	17
NA07	65	16	14	4	42	12	31	15
NA09	79	8	8	5	36	15	26	23
NA11	61	29	6	3	42	12	35	12
NA12	70	16	11	3	36	16	31	17
NA15	77	9	10	3	43	13	35	9
NA16	75	11	11	3	35	18	32	15
NA17	71	13	11	5	37	15	32	15
NA19	73	14	6	6	41	14	24	21
NA20	58	29	3	10	50	20	15	15
NA22	74	15	10	2	34	11	39	16
Total	68	15	12	5	38	15	25	22
Southwest Marine								
SW02	69	18	11	2	46	11	28	14
SW03	52	22	24	2	36	13	36	15
SW04	84	2	13	1	37	12	28	22
SW08	60	5	25	10	40	14	26	20
SW09	71	11	16	2	41	15	29	15
SW11	50	26	21	3	31	13	38	18
SW13	64	18	14	3	39	15	29	18
SW15	54	34	8	4	44	17	20	19
SW17	66	25	6	2	42	16	28	14
SW18	48	35	15	2	45	18	26	12
SW21	80	12	6	2	43	15	35	7
SW22	62	23	11	3	38	15	39	8
SW23	57	27	10	6	42	17	33	8
SW25	58	21	14	6	33	14	30	23
SW27	56	26	13	5	38	14	28	20
Total	65	15	15	4	39	18	25	19
Reference								
2441	72	6	5	18	47	22	17	14
2433	76	11	8	6	43	21	18	17
2440	69	11	19	1	38	21	28	13
2231	8	1	91	1	39	16	26	18
2243	64	10	14	12	27	18	30	25
Total[b]	69	9	12	9	37	22	23	19

[a] Abundance and richness calculated across all five replicates at each station.

[b] Excludes 2231.

Table 8-3. Comparison of the 10 most abundant benthic taxa at the shipyard sites and the reference area

Species	Major Taxon	Relative Abundance (percent)
Reference Area		
Lumbrineris sp.	Polychaeta	22.0
Exogene lourei	Polychaeta	9.5
Leitoscoloplos pugettensis	Polychaeta	7.2
Diplocirrus sp. SD1	Polychaeta	6.0
Mediomastus sp.	Polychaeta	5.0
Pista percyi	Polychaeta	4.8
Nematoda	Nematoda	2.9
Edwardsia californica	Anthozoa	2.7
Paracerceis sculpta	Crustacea	2.4
Scyphoproctus oculatus	Polychaeta	2.3
Shipyard Sites		
Exogene lourei [a]	Polychaeta	17.1
Pseudopolydora paucibranciata	Polychaeta	14.6
Lumbrineris sp.[a]	Polychaeta	10.3
Musculista senhousia	Mollusca	8.4
Theora lubrica	Mollusca	4.7
Pista percyi [a]	Polychaeta	4.5
Leitoscoloplos pugettensis [a]	Polychaeta	3.6
Synaptotanais notabilis	Crustacea	3.4
Scyphoproctus oculatus [a]	Polychaeta	2.6
Mediomastus sp.[a]	Polychaeta	1.8

[a] Taxon at the shipyard site that was also one of the 10 most abundant taxa in the reference area.

Table 8-4. Transformations and tests for statistical comparisons

Variable	Transformation	Statistical Test
Total abundance	Logarithmic	Parametric
Crustacean abundance	Logarithmic	Parametric
Mollusc abundance	None	Non-parametric
Polychaete abundance	None	Non-parametric
Total richness	Logarithmic (square root also homogenizes variance)	Parametric
SDI	Logarithmic	Parametric
Percent dominance	None	Parametric
H′	None	Parametric

Note: H′ - Shannon-Wiener diversity index
SDI - Swartz' dominance index

Table 8-5. Results of statistical comparisons of shipyard and reference stations

Station	Abundance: Total	Abundance: Polychaetes	Abundance: Molluscs	Abundance: Crustaceans	Taxa Richness	SDI	Percent Dominance	H′ (diversity)
NASSCO								
NA01		**						
NA03								
NA04	**	**			**			
NA05								
NA06								
NA07								
NA09								
NA11						*		
NA12								
NA15	**	*			**			*
NA16								
NA17								
NA19								
NA20				**	**	*		*
NA22	**	**	*	**	**	*		**
Southwest Marine								
SW02						**	**	*
SW03		*						
SW04						**	**	**
SW08						**		
SW09								
SW11								
SW13								
SW15								
SW17					*	**		
SW18								
SW21	**			*	**			
SW22		*			**			
SW23	**	**			**			
SW25								
SW27								

Note: * - statistical significance at an alpha of 0.05
** - statistical significance at an alpha of 0.01
H′ - Shannon-Wiener diversity index
SDI - Swartz' dominance index

Table 8-6. Summary of the 10 most abundant benthic taxa at Station NA22

Taxon	Total Abundance (5 reps)	Benthic Group
NA22		
Polychaete *Lumbrineris*	148	
Polychaete *Cossura*	100	
Polychaete *Leitoscoloplos*	60	2
Mollusc *Haminoea*	50	
Polychaete *Pseudopolydora*	34	1
Mollusc *Theora*	22	2
Decapod *Ambidexter*	16	
Polychaete *Euchone*	16	
Polychaete *Mediomastus*	10	3
Polychaete *Cossura*	8	

Table 8-7. Summary of the 10 most abundant benthic taxa at each reference station

Taxon	Total Abundance (5 reps)	Benthic Group
Station 2441		
Polychaete *Lumbrineris*	804	
Polychaete *Leitoscoloplos*	274	2
Cnidarian *Edwardsia*	240	
Polychaete *Chaetozone*	146	
Echinoderm *Amphiodia*	104	
Polychaete *Pista*	68	2
Polychaete *Amphicteis*	60	
Polychaete *Euclymeninae*	40	
Echinoderm *Syncoryne*	36	
Mollusc *Macoma*	32	
Station 2433		
Polychaete *Lumbrineris*	530	
Polychaete *Diplocirrus*	520	
Polychaete *Leitoscoloplos*	192	2
Mollusc *Theora*	98	2
Polychaete *Chaetozone*	76	
Isopod *Neastacilla*	76	
Cnidarian *Acanthoptilum*	72	
Polychaete *Euclymeninae*	56	
Polychaete *Spiophanes*	54	
Mollusc *Tagelus*	40	
Station 2440		
Polychaete *Pista*	558	2
Polychaete *Leitoscoloplos*	440	2
Polychaete *Lumbrineris*	416	
Polychaete *Diplocirrus*	222	
Polychaete *Mediomastus*	230	3
Tanaid *Synaptotanais*	194	3
Ostracod *Euphilomedes*	128	
Polychaete *Exogene*	102	1
Mollusc *Theora*	98	2
Amphipod *Amphideutopus*	84	
Station 2243		
Polychaete *Exogene*	1,118	1
Polychaete *Lumbrineris*	1,082	
Nematodes	378	3
Polychaete *Mediomastus*	364	3
Polychaete *Scyphoproctus*	298	3
Isopod *Paracerceis*	296	3
Mollusc *Musculista*	192	1
Amphipod *Podocerus*	152	3
Polychaete *Pseudopolydora*	108	1
Cnidarian *Edwardsia*	104	

Table 8-8. Summary of the 10 most abundant benthic taxa at Stations SW13 and SW15

Taxon	Total Abundance (5 reps)	Benthic Group
SW13		
Mollusc *Theora*	332	2
Polychaete *Dorvillea*	300	3
Polychaete *Pista*	272	2
Polychaete *Lumbrineris*	268	
Polychaete *Cirratulidae*	222	
Polychaete *Leitoscoloplos*	196	2
Mollusc *Ostrea*	152	
Polychaete *Exogene*	136	1
Polychaete *Neanthes*	106	3
Polychaete *Harmothoe*	104	3
Amphipod *Podocerus*	232	3
SW15		
Mollusc *Theora*	914	2
Polychaete *Lumbrineris*	274	
Polychaete *Euchone*	258	
Mollusc *Ostrea*	152	
Polychaete *Lumbrineris*	174	
Polychaete *Pseudopolydora*	170	1
Polychaete *Dorvillea*	138	3
Polychaete *Mediomastus*	134	3
Polychaete *Syllis*	112	
Polychaete *Pherusa*	94	

Table 8-9. Summary of the 10 most abundant benthic taxa at Stations SW04 and SW08

Taxon	Total Abundance (5 reps)	Benthic Group
SW04		
Polychaete *Pseudopolydora*	8,074	1
Polychaete *Exogene*	4,150	1
Tanaid *Synaptonais*	1,210	3
Polychaete *Lumbrineris*	278	
Mollusc *Musculista*	226	1
Polychaete *Neanthes*	218	3
Amphipod *Grandidierella*	224	3
Isopod *Paranthura*	174	
Polychaete *Scyphoproctus*	152	3
Amphipod *Podocerus*	146	3
SW08		
Polychaete *Exogene*	2,860	1
Tanaid *Synaptotanais*	2,038	3
Polychaete *Pseudopolydora*	1,936	1
Polychaete *Scyphoproctus*	1,486	3
Oligochaetes	842	3
Mollusc *Musculita*	454	1
Amphipod *Grandidierella*	336	3
Polychaete *Polydora*	272	
Nematodes	216	3
Polychaete *Harmothoe*	192	3
Polychaete *Neanthes*	178	3

Table 8-10. Benthic community alterations at shipyard stations

Station	Difference	Comments
NASSCO		
NA01	Minor	Only one benthic metric (i.e., polychaete abundance) was different from reference conditions. In contrast, the abundance of sensitive crustaceans is significantly higher than reference conditions. Mean amphipod abundance (107/sample) is higher than the reference range (18–68/sample), and mean number of amphipod taxa (11/sample) is equal to the upper bound of the reference range (5–11/sample). The community at Station NA01 also clusters closely with the community at Station NA03, which does not exhibit differences based on the benthic metrics.
NA03	None	
NA04	Major	Three benthic metrics indicate differences from reference conditions (i.e., total abundance, polychaete abundance, and taxa richness)
NA05	None	
NA06	None	
NA07	None	
NA09	None	
NA11	Minor	Only one benthic metric (i.e., SDI) is different from reference. In contrast, two of the three most abundant taxa (i.e., *Exogene lourei*, *Lumbrineris* sp.) are dominant at one or more reference stations. In addition, the community at Station NA11 clusters closely with the community at Station NA05, which does not exhibit differences based on the benthic metrics.
NA12	None	
NA15	Major	Three benthic metrics are different from reference (i.e., total abundance, polychaete abundance, and taxa richness)
NA16	None	
NA17	None	
NA19	None	
NA20	Major	Four benthic metrics are different from reference (i.e., crustacean abundance, taxa richness, SDI, and H′)
NA22	Major	Seven benthic metrics are different from reference (i.e., all metrics except percent dominance)
Southwest Marine		
SW02	Minor	Two benthic metrics are different from reference (i.e., SDI and H′). However, mean crustacean abundance (104/sample) is considerably higher than the mean reference value (79/sample). In addition, the community at Station SW02 clusters closely with the community at Station SW09, which does not exhibit differences based on the benthic metrics.
SW03	Moderate	Only one benthic metric (i.e., polychaete abundance) is different from reference. In addition, mean amphipod abundance (66/sample) is near the upper bound of the reference range (18–68/sample), and the mean number of amphipod taxa (9/sample) is well within the reference range (5–11/sample). However, the community at Station SW03 clusters closely with the community at Station NA04, which exhibits differences based on three benthic metrics.
SW04	Minor	Two of the metrics are different from reference, but abundances of polychaetes and crustaceans, as well as overall abundance, are much higher than at reference stations.

Table 8-10. (cont.)

Station	Difference	Comments
SW08	None	SDI is different from reference, but none of the abundances of major taxonomic groups, nor total abundance, are different.
SW09	None	
SW11	None	
SW13	None	
SW15	None	
SW17	Moderate	Two benthic metrics are different from reference (i.e., taxa richness and SDI). In addition, mean crustacean abundance (40/sample) is low compared to the mean reference value (79/station), although the difference is not statistically significant.
SW18	None	
SW21	Major	Three benthic metrics are different from reference (i.e., total abundance, crustacean abundance, and taxa richness)
SW22	Major	Two benthic metrics are different from reference (polychaete abundance and taxa richness), and the community at Station SW22 clusters closely with the communities at Stations SW21 and SW23, which exhibit major differences.
SW23	Major	Three benthic metrics indicate differences (i.e., total abundance, polychaete abundance, and taxa richness)
SW25	None	
SW27	None	

Note: H′ - Shannon-Wiener diversity index
SDI - Swartz' dominance index

Table 8-11. Summary of BRI values, benthic response levels, and major benthic community metrics for each reference and shipyard station

	BRI Value	Response Level[a]	Percent of Taxa without Pollution Tolerance Scores	Mean Total Abundance[b]	Mean Taxa Richness	Mean Species Diversity[c]
Reference Stations						
2441	19.9	Reference	44	505	47.8	2.8
2433	16.8	Reference	36	440	34.8	2.57
2440	32.2	1	35	639	39	2.72
2231	31	Reference	53	6,232	64	0.79
2243	45.1	2	42	987	39	2.49
Mean Value[d]				643	40.2	2.65
NASSCO						
NA01	42.2	2	42	447	33.4	2.79
NA03	45.5	2	37	492	39.8	2.99
NA04	49.6	2	35	285 *	25.2 *	2.5
NA05	44.4	2	41	569	34.6	2.42
NA06	54.4	3	54	611	36.6	2.7
NA07	44.6	2	49	475	42.8	2.96
NA09	51.1	2	49	862	44	2.61
NA11	46	2	46	604	33.4	2.39
NA12	42.6	2	48	538	37.2	2.74
NA15	51	2	31	306 *	25.8 *	2.33 *
NA16	48	2	41	522	33.4	2.56
NA17	55.3	3	48	418	33.2	2.73
NA19	46.7	2	47	828	42.8	2.71
NA20	54	3	40	412	21.6 *	2.31 *
NA22	51.6	2	31	107 *	15.0 *	2.18 *
Southwest Marine						
SW02	52.1	2	45	976	39.2	2.35 *
SW03	49.9	2	48	361	30.6	2.79
SW04	41.1	1	50	3,175	35.6	1.58 *
SW08	41.5	1	51	2,457	41	2.42
SW09	53.2	3	51	572	39.2	2.74
SW11	42.4	2	52	777	44.2	2.92
SW13	43.6	2	62	742	52.6	3.17
SW15	37.8	1	61	806	58.8	3.14
SW17	45.7	2	38	621	30.0 *	2.37
SW18	39.5	1	44	829	42	2.77
SW21	53.2	3	26	315 *	24.0 *	2.38
SW22	55.1	3	46	363	26.2 *	2.41
SW23	50	2	33	316 *	26.6 *	2.57
SW25	41.3	1	48	611	40.2	2.79
SW27	42.9	2	49	927	47.8	2.9

Note: * - value is significantly less ($p \leq 0.05$) than mean reference value
BRI - benthic response index

[a] Based on the ranges of BRI values identified by Smith et al. (2003). For assigning response levels, each BRI value was rounded up to the next highest value, unless it was a whole number (i.e., 42.0 = 42, whereas 42.1 = 43).

[b] Per sample.

[c] Shannon-Wiener Diversity Index (H′).

[d] Excludes data for Station 2231.

Table 8-12. Relative abundance of the 20 most abundant benthic macroinvertebrate taxa at the reference stations and shipyard sites

Reference Stations		NASSCO Site		Southwest Marine Site	
Taxon	Relative Abundance (percent)	Taxon	Relative Abundance (percent)	Taxon	Relative Abundance (percent)
Lumbrineris sp.	22.0	*Exogone lourei*	17.9	*Pseudopolydora paucibranchiata*	19.0
Exogone lourei	9.5	*Lumbrineris* sp.	17.8	*Exogone lourei*	16.6
Leitoscoloplos pugettensis	7.2	*Musculista senhousia*	10.6	*Musculista senhousia*	7.2
Diplocirrus sp. SD1	6.0	*Pseudopolydora paucibranchiata*	6.4	*Lumbrineris* sp.	6.2
Mediomastus sp.	5.0	*Pista alata*	5.4	*Theora lubrica*	5.9
Pista alata	4.8	*Leitoscoloplos pugettensis*	4.5	*Synaptotanais notabilis*	5.0
Nematoda	2.9	*Scyphoproctus oculatus*	2.8	*Pista alata*	4.1
Edwardsia californica	2.7	*Theora lubrica*	2.6	*Leitoscoloplos pugettensis*	3.1
Paracerceis sculpta	2.4	*Mediomastus* sp.	2.3	*Scyphoproctus oculatus*	2.5
Scyphoproctus oculatus	2.3	*Prionospio heterobranchia*	1.8	*Dorvillea (Schistomeringos) longicornis*	1.8
Musculista senhousia	2.0	*Harmothoe imbricata*	1.6	*Prionospio heterobranchia*	1.7
Chaetozone corona	1.8	*Paracerceis sculpta*	1.5	*Mediomastus* sp.	1.5
Theora lubrica	1.6	*Scolanthus* sp. B	1.5	Oligochaeta	1.5
Synaptotanais notabilis	1.6	*Heterophoxus* cf *ellisi*	1.5	*Grandidierella japonica*	1.5
Lyonsia californica	1.5	*Protocirrineris* sp. A	1.4	*Harmothoe imbricata*	1.2
Podocerus fulanus	1.5	*Podocerus fulanus*	1.4	*Neanthes acuminata*	1.1
Amphideutopus oculatus	1.3	Nematoda	1.2	*Amphideutopus oculatus*	1.1
Euphilomedes carcharodonta	1.1	*Neanthes acuminata*	1.0	*Heterophoxus* cf *ellisi*	1.0
Pseudopolydora paucibranchiata	0.9	*Amphideutopus oculatus*	1.0	*Podocerus fulanus*	1.0
Amphiodia urtica	0.9	Oligochaeta	0.9	Nematoda	0.8
All 20 taxa	**79.1**	**All 20 taxa**	**84.9**	**All 20 taxa**	**83.7**

Table 8-13. Summary of p_i values and relative abundances of benthic macroinvertebrate taxa found at the reference stations and shipyard sites

p_i Value	Taxon	Relative Abundance (percent)		
		Reference Station	NASSCO	Southwest Marine
150.473	*Macoma nasuta*	<0.1	--	--
150.452	*Marphysa* sp.	--	--	<0.1
150.301	*Mayerella banksia*	0.1	0.2	0.2
122.293	*Pherusa capulata*	0.1	0.2	0.5
120.771	*Ambidexter panamensis*	--	0.1	0.1
97.387	*Aphelochaeta/Monticellina* complex	0.3	<0.1	0.1
96.217	*Pyromaia tuberculata*	0.2	0.8	0.5
94.277	*Leitoscoloplos pugettensis*	7.2	4.5	3.1
89.682	*Neanthes acuminata* complex	0.3	1.0	1.1
88.339	*Capitella capitata* complex	--	<0.1	0.1
77.062	Edwardsiidae	3.0	1.6	0.5
69.863	*Drilonereis* sp.	--	0.1	<0.1
69.863	*Musculista senhousia*	2.0	10.6	7.2
68.492	*Schmittius politus*	<0.1	<0.1	<0.1
67.935	*Pectinaria californiensis*	<0.1	<0.1	--
65.688	*Pista alata*	4.8	5.4	4.1
65.464	Synaptidae	0.4	<0.1	<0.1
64.715	*Syllis (Typosyllis)* spp.	0.2	0.2	0.3
62.203	*Rictaxis punctocaelatus*	0.1	--	<0.1
61.628	*Oxyurostylis pacifica*	<0.1	--	<0.1
57.289	*Paracerceis sculpta*	2.4	1.5	0.4
55.417	*Theora lubrica*	1.6	2.6	5.9
54.851	*Leptopecten latiauratus*	0.1	--	<0.1
53.556	*Haminoea vesicula*	0.1	0.4	0.1
53.355	*Nephtys ferruginea*	--	<0.1	--
53.290	*Alpheus californiensis*	<0.1	0.2	0.1
52.772	*Odontosyllis phosphorea*	0.1	0.1	0.3
52.640	*Nassarius tiarula*	0.2	0.4	0.1
51.323	*Philine auriformis*	0.1	--	--
48.162	*Exogone lourei*	9.5	17.9	16.6
47.994	*Bemlos macromanus*	0.2	0.6	<0.1
47.936	*Grandidierella japonica*	0.2	0.5	1.5
47.842	*Lumbrineris* sp.	22.4	18.9	7.1
47.047	*Paradexamine* sp.	0.1	0.2	0.4
46.062	*Poecilochaetus* sp. A	--	<0.1	<0.1
45.212	*Euchone* sp.	0.2	0.5	0.7
44.940	*Scyphoproctus* sp.	2.3	2.8	2.5
42.886	*Spiochaetopterus costarum*	--	--	<0.1
41.930	*Macoma yoldiformis*	0.4	--	--
41.732	*Nephtys cornuta*	<0.1	<0.1	<0.1
40.620	*Monoculodes* sp.	<0.1	<0.1	<0.1
39.757	*Malacoplax californiensis*	<0.1	--	--
37.542	*Pseudopolydora paucibranchiata*	0.9	6.4	19.0
37.414	*Aoroides* sp.	0.1	<0.1	0.3
37.356	*Eteone* sp.	<0.1	<0.1	<0.1
34.328	*Polydora* sp.	<0.1	<0.1	0.4
33.071	*Paraprionospio pinnata*	0.2	<0.1	--
32.809	Phoronida	0.3	<0.1	<0.1
32.335	*Armandia brevis*	0.1	0.4	0.1
31.255	*Cirriformia* sp.	--	<0.1	<0.1
29.193	*Mediomastus* sp.	5.0	2.3	1.5
28.772	*Paranthura elegans*	<0.1	0.1	0.6
28.468	*Diplocirrus* sp.	6.0	0.5	0.3
27.010	Pycnogonida	<0.1	<0.1	<0.1

Table 8-13. (cont.)

p_i Value	Taxon	Relative Abundance (percent)		
		Reference Station	NASSCO	Southwest Marine
26.322	*Synaptotanais notabilis*	1.6	0.5	5.0
26.309	*Prionospio (Prionospio) heterobranchia*	0.6	1.8	1.7
25.789	Ceriantharia	<0.1	<0.1	--
24.304	*Heterophoxus* sp.	0.1	1.5	1.0
22.722	*Euphilomedes carcharodonta*	1.1	0.4	0.4
22.480	*Poecilochaetus johnsoni*	--	<0.1	<0.1
21.632	*Aphrodita* sp.	--	<0.1	<0.1
19.896	*Cryptomya californica*	<0.1	<0.1	--
18.250	*Eumida longicornuta*	<0.1	<0.1	<0.1
18.174	*Argopecten ventricosus*	--	--	<0.1
15.229	*Scleroplax granulata*	0.1	--	<0.1
14.764	*Diopatra ornata*	0.1	--	<0.1
14.573	*Spiophanes missionensis*	0.7	<0.1	<0.1
13.043	*Amphideutopus oculatus*	1.3	1.0	1.1
12.682	*Podocerus fulanus*	1.5	1.4	1.0
10.319	*Serolis carinata*	0.3	0.4	0.3
9.942	*Rochefortia* sp.	0.1	--	--
8.368	*Syllis (Syllis) gracilis*	--	<0.1	0.2
6.715	*Metasychis disparidentatus*	0.1	--	--
4.949	*Prionospio lighti*	<0.1	--	--
4.060	*Glycera americana*	0.2	0.1	<0.1
0.789	*Pista fasciata*	0.0	--	--
0.733	*Leptochelia dubia*	0.1	0.1	0.1
0.664	*Laevicardium substriatum*	0.5	0.1	0.1
0.065	*Chaetozone corona*	1.8	--	<0.1
–2.751	Pennatulacea	0.7	--	--
–3.225	*Halosydna johnsoni*	<0.1	--	<0.1
–4.295	*Amaeana occidentalis*	0.3	<0.1	<0.1
–4.847	*Microspio pigmentata*	0.2	<0.1	<0.1
–4.874	*Neotrypaea* sp.	<0.1	<0.1	<0.1
–5.094	*Amphipholis* sp.	0.1	0.1	<0.1
–5.582	*Amphicteis scaphobranchiata*	0.5	--	<0.1
–6.496	*Notomastus* sp.	0.3	<0.1	0.1
–9.515	*Tagelus subteres*	0.9	<0.1	<0.1
–9.638	*Nephtys caecoides*	0.1	<0.1	<0.1
–11.479	*Scolelepis (Parascolelepis)* sp.	<0.1	--	<0.1
–12.356	*Solen* sp.	0.1	--	--
–14.040	*Melinna oculata*	0.1	<0.1	<0.1
–14.614	*Tectura depicta*	--	--	0.1
–16.227	*Sthenelanella uniformis*	0.1	--	<0.1
–16.324	*Crucibulum spinosum*	<0.1	<0.1	0.2
–19.478	Mactridae	0.1	--	--
–22.455	*Piromis* sp.	--	--	<0.1
–22.510	*Amphiodia* complex	1.0	--	--
–24.214	*Goniada littorea*	0.1	--	--
–25.411	*Apoprionospio pygmaea*	<0.1	--	--
–28.846	*Chione* sp.	0.1	<0.1	<0.1
–29.760	*Listriella melanica*	0.1	--	<0.1
–30.541	*Neastacilla californica*	0.6	--	--
–31.128	Ostreidae	--	0.1	0.7
–36.193	*Periploma/Thracia* complex	0.2	<0.1	--
–38.574	Aglajidae	--	--	<0.1
–40.832	*Nuculana* sp.	<0.1	--	--
–45.950	*Praxillella* sp.	0.1	--	--
–46.685	*Protothaca* sp.	--	<0.1	<0.1

Table 8-13. (cont.)

p_i Value	Taxon	Relative Abundance (percent)		
		Reference Station	NASSCO	Southwest Marine
–47.136	*Cooperella subdiaphana*	<0.1	--	--
–48.531	*Mytilus* sp.	--	<0.1	0.1
–51.157	*Tellina modesta*	0.1	--	--
–61.775	*Ampharete labrops*	0.1	--	--
–63.455	*Molgula* sp.	<0.1	--	--
–63.807	*Asteropella slatteryi*	0.3	<0.1	<0.1
–68.361	*Hiatella arctica*	--	--	<0.1
–75.217	*Ericthonius brasiliensis*	--	0.1	0.1
–105.945	*Ampelisca cristata*	0.1	--	--
–112.389	*Vargula tsujii*	<0.1	--	--

Note: Relative abundances are expressed as percent of total abundance at each site.

-- - taxon not found at site

Table 8-14. **BRI values for the final reference pool**

Investigation	Station	BRI
Shipyards Phase 1 (2001)	2441	20
Shipyards Phase 1 (2001)	2433	17
Chollas/Paleta (2001)	2433	24
Bight '98	2231	15
Bight '98	2233	28
Bight '98	2238	38
Bight '98	2240	28
Bight '98	2241	34
Bight '98	2242	35
Bight '98	2243	34
Bight '98	2244	32
Bight '98	2247	32
Bight '98	2252	6.0
Bight '98	2256	37
Bight '98	2257	37
Bight '98	2265	25
Bight '98	2433	21
Bight '98	2435	-3.6
Bight '98	2436	20
Bight '98	2440	29

Note: BRI - benthic response index

Table 8-15. Benthic macroinvertebrate metrics for stations at Response Level 3

Station	BRI Score	Abundance	Richness	Shannon-Wiener Diversity
SW09	54	574	40	2.8
SW21	54	316	24	2.4
SW22	56	364	27	2.4
NA06	55	615	37	2.7
NA17	56	421	34	2.7
NA20	55	413	22	2.3
Reference range	17–45	441–989	35–48	2.5–2.8

Table 8-16. Summary of total abundance and taxa richness at the shipyard and reference stations

Station	Benthic Alteration	Total Abundance (per m^2)	Taxa Richness (per sample)
NASSCO			
NA01	Minor	4,470	33.4
NA03	None	4,920	39.8
NA04	Major	2,850	25.2
NA05	None	5,690	34.6
NA06	None	6,110	36.6
NA07	None	4,750	42.8
NA09	None	8,620	44.0
NA11	Minor	6,040	33.4
NA12	None	5,380	37.2
NA15	Major	3,060	25.8
NA16	None	5,220	33.4
NA17	None	4,180	33.2
NA19	None	8,280	42.8
NA20	Major	4,120	21.6
NA22	Major	1,070	15.0
Southwest Marine			
SW02	Minor	9,760	39.2
SW03	Moderate	3,610	30.6
SW04	Minor	31,750	35.6
SW08	None	24,570	41.0
SW09	None	5,720	39.2
SW11	None	7,770	44.2
SW13	None	7,420	52.6
SW15	None	8,060	58.8
SW17	Moderate	6,210	30.0
SW18	None	8,290	42.0
SW21	Major	3,150	24.0
SW22	Major	3,630	26.2
SW23	Major	3,160	26.6
SW25	None	6,110	40.2
SW27	None	9,270	47.8
Reference Areas			
Mean		6,430	40.2

Table 8-17. Summary of microscopic and macroscopic lesions that were significantly elevated at one or more shipyard locations relative to the reference area

Lesion	Severity Score	Prevalence (percent) NASSCO Shipyard Inside	NASSCO Shipyard Outside	Southwest Marine Shipyard Inside	Southwest Marine Shipyard Outside	Reference Area
Microscopic Lesions						
Liver						
Abundant Lipofuscin	0	$\boxed{74}$	92	$\boxed{75}$	88	96
	1	$\boxed{12}$	6	$\boxed{6}$	12	4
	2	$\boxed{2}$	2	$\boxed{8}$	0	0
	3	$\boxed{12}$	0	$\boxed{12}$	0	0
Abundant Hemosiderin	0	98	$\boxed{78}$	98	80	94
	1	2	$\boxed{22}$	2	20	6
	2	0	$\boxed{0}$	0	0	0
	3	0	$\boxed{0}$	0	0	0
Kidney						
Nephritis	0	$\boxed{48}$	66	76	66	75
	1	$\boxed{48}$	32	22	32	25
	2	$\boxed{4}$	2	0	2	0
	3	$\boxed{0}$	0	2	0	0
Macroscopic Lesions						
Gills						
Shiny gill foci	0	12	10	$\boxed{0}$	0	10
	1	62	81	$\boxed{0}$	70	69
	2	24	8	$\boxed{100}$	28	20
	3	2	0	$\boxed{0}$	2	2

Note: Boxed values for shipyard locations are significantly greater relative to the reference values.

Table 8-18. Summary of microscopic and macroscopic lesions and other conditions that were significantly elevated in the reference area relative to one or more shipyard locations

Lesion	Severity Score	Prevalence (percent) NASSCO Shipyard Inside	NASSCO Shipyard Outside	Southwest Marine Shipyard Inside	Southwest Marine Shipyard Outside	Reference Area
Microscopic Examination						
Kidney						
Renal tubular regeneration	0	56	$\boxed{72}$	65	66	42
	1	44	$\boxed{28}$	31	28	42
	2	0	$\boxed{0}$	2	6	13
	3	0	$\boxed{0}$	2	0	2
Gonads						
Atresia of yolked follicles	0	0	0	$\boxed{0}$	0	0
	1	25	36	$\boxed{48}$	15	14
	2	58	36	$\boxed{29}$	54	19
	3	17	27	$\boxed{24}$	31	67
Macroscopic Examination						
Fins						
Caudal fin fraying	0	12	6	6	$\boxed{24}$	4
	1	88	94	88	$\boxed{76}$	94
	2	0	0	6	$\boxed{0}$	2
	3	0	0	0	$\boxed{0}$	0
Caudal fin reddening	0	$\boxed{92}$	$\boxed{92}$	82	86	67
	1	$\boxed{8}$	$\boxed{8}$	18	14	33
	2	$\boxed{0}$	$\boxed{0}$	0	0	0
	3	$\boxed{0}$	$\boxed{0}$	0	0	0
Body Cavity						
Diffuse opaque epicardium	0	$\boxed{84}$	34	$\boxed{94}$	30	40
	1	$\boxed{16}$	66	$\boxed{6}$	70	60
	2	$\boxed{0}$	0	$\boxed{0}$	0	0
	3	$\boxed{0}$	0	$\boxed{0}$	0	0
Mean number of Anisakis parasites	NA	$\boxed{0.16}$	0.92	$\boxed{0.20}$	0.62	1.12

Note: Boxed values for shipyard locations are significantly elevated relative to the reference values.

NA - not applicable

Table 8-19. Summary of prevalences of serious liver lesions in spotted sand bass from the shipyard locations and the reference area

	Prevalence (percent)				
	NASSCO Shipyard		Southwest Marine Shipyard		Reference
Liver Lesion	Inside	Outside	Inside	Outside	Area
Hydropic Vacuolation	0	0	0	0	0
Specific Degenerative Conditions					
Nuclear pleomorphism	0	0	0	0	0
Megalocytic hepatosis	0	0	0	0	0
Foci of Cellular Alteration					
Eosinophilic foci	0	0	0	0	0
Basophilic foci	0	0	0	0	0
Clear cell foci	0	0	0	0	0
Neoplasms					
Hepatocellular carcinoma	0	0	0	0	0
Liver cell adenoma	0	0	0	0	1.9
Cholangioma	0	0	0	0	0
Cholangiocellular carcinoma	0	0	0	0	1.9

Table 9-1. Correlations between chemical concentrations and adverse biological effects

Chemical	Amphipod Survival	Echinoderm Fertilization	Bivalve Development	Benthic Macroinvertebrate Total Abundance	Benthic Macroinvertebrate Total Richness
Arsenic	0.01	0.27	-0.02	0.13	0.05
Cadmium	0.31	0.38	-0.07	0.07	-0.08
Chromium	-0.10	0.09	-0.12	0.12	-0.01
Copper	0.03	0.32	-0.03	0.17	-0.01
Lead	0.03	0.39	0.00	0.08	-0.08
Mercury	-0.09	0.51	0.19	0.03	-0.10
Nickel	-0.03	0.14	-0.12	0.21	0.10
Selenium	0.16	0.00	**-0.15**	**-0.22**	**-0.18**
Silver	0.04	0.24	0.04	0.12	-0.04
Zinc	0.03	0.28	-0.07	0.19	0.03
Monobutyltin	0.05	-0.02	-0.09	0.23	0.09
Dibutyltin	0.09	-0.02	-0.01	0.13	-0.06
Tributyltin	0.08	0.05	-0.02	0.12	-0.08
Tetrabutyltin	0.07	0.05	-0.03	0.20	0.01
LPAH	0.00	0.31	0.00	0.18	0.05
HPAH	-0.10	0.31	0.06	0.19	0.09
PCB homologs	0.01	0.35	-0.02	0.13	-0.06
PCTs	-0.10	0.53	0.32	0.08	-0.04
DRO	0.07	0.51	-0.11	-0.10	**-0.21**
RRO	0.08	0.48	-0.09	**-0.19**	**-0.29**
Percent fines	**-0.17**	-0.08	**-0.20**	**-0.14**	-0.05
Percent clay	**-0.28**	**-0.35**	**-0.29**	-0.07	-0.06

Note: Correlations computed using Phase 1 data from triad stations.
Tabled values are Spearman rank correlation coefficients.
Bold values ($r < -0.1265$) are statistically significant ($p = 0.95$).

DRO - diesel-range organics
HPAH - high-molecular-weight polycyclic aromatic hydrocarbon
LPAH - low-molecular-weight polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl
PCT - polychlorinated terphenyl
RRO - residual-range organics

Table 9-2. Correlations among sediment chemicals

	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Selenium	Silver	Zinc	MBT	DBT
Arsenic	**1.00**	**0.66**	**0.65**	**0.92**	**0.86**	**0.63**	**0.60**	0.20	**0.69**	**0.97**	**0.77**	**0.80**
Cadmium	**0.66**	**1.00**	0.28	**0.61**	**0.66**	0.42	**0.62**	0.04	**0.49**	**0.71**	**0.45**	**0.51**
Chromium	**0.65**	0.28	**1.00**	**0.79**	**0.73**	**0.86**	**0.82**	-0.03	**0.90**	**0.62**	**0.75**	**0.74**
Copper	**0.92**	**0.61**	**0.79**	**1.00**	**0.90**	**0.78**	**0.74**	0.13	**0.82**	**0.94**	**0.84**	**0.86**
Lead	**0.86**	**0.66**	**0.73**	**0.90**	**1.00**	**0.77**	**0.66**	0.01	**0.81**	**0.89**	**0.80**	**0.86**
Mercury	**0.63**	0.42	**0.86**	**0.78**	**0.77**	**1.00**	**0.79**	-0.02	**0.85**	**0.61**	**0.65**	**0.64**
Nickel	**0.60**	**0.62**	**0.82**	**0.74**	**0.66**	**0.79**	**1.00**	-0.10	**0.84**	**0.60**	**0.62**	**0.63**
Selenium	0.20	0.04	-0.03	0.13	0.01	-0.02	-0.10	**1.00**	-0.04	0.19	0.05	0.12
Silver	**0.69**	**0.49**	**0.90**	**0.82**	**0.81**	**0.85**	**0.84**	-0.04	**1.00**	**0.70**	**0.76**	**0.76**
Zinc	**0.97**	**0.71**	**0.62**	**0.94**	**0.89**	**0.61**	**0.60**	0.19	**0.70**	**1.00**	**0.80**	**0.83**
Monobutyltin (MBT)	**0.77**	**0.45**	**0.75**	**0.84**	**0.80**	**0.65**	**0.62**	0.05	**0.76**	**0.80**	**1.00**	**0.96**
Dibutyltin (DBT)	**0.80**	**0.51**	**0.74**	**0.86**	**0.86**	**0.64**	**0.63**	0.12	**0.76**	**0.83**	**0.96**	**1.00**
Tributyltin (TBT)	**0.81**	**0.51**	**0.73**	**0.89**	**0.87**	**0.63**	**0.62**	0.13	**0.74**	**0.85**	**0.93**	**0.98**
Tetrabutyltin (TetBT)	**0.77**	**0.57**	**0.52**	**0.79**	**0.71**	**0.45**	**0.52**	0.15	**0.60**	**0.81**	**0.86**	**0.86**
LPAH	**0.71**	**0.58**	**0.67**	**0.76**	**0.81**	**0.62**	**0.67**	-0.24	**0.71**	**0.70**	**0.73**	**0.77**
HPAH	**0.68**	**0.52**	**0.76**	**0.76**	**0.84**	**0.73**	**0.72**	-0.26	**0.79**	**0.67**	**0.77**	**0.79**
PCBs (homologs)	**0.78**	**0.65**	**0.80**	**0.86**	**0.92**	**0.85**	**0.77**	0.00	**0.87**	**0.80**	**0.78**	**0.81**
PCTs	**0.69**	**0.68**	**0.58**	**0.75**	**0.74**	**0.83**	**0.70**	0.04	**0.68**	**0.69**	**0.54**	**0.56**
DRO	**0.54**	**0.56**	**0.74**	**0.71**	**0.80**	**0.80**	**0.74**	0.05	**0.80**	**0.58**	**0.63**	**0.68**
RRO	**0.48**	**0.48**	**0.67**	**0.66**	**0.75**	**0.74**	**0.64**	0.13	**0.72**	**0.53**	**0.58**	**0.64**
Percent fines	0.20	-0.11	**0.78**	0.38	0.29	**0.58**	**0.56**	-0.09	**0.59**	0.14	**0.48**	**0.46**

Table 9-2. (cont.)

	TBT	TetBT	LPAH	HPAH	PCBs	PCTs	DRO	RRO	Percent Fines
Arsenic	**0.81**	**0.77**	**0.71**	**0.68**	**0.78**	**0.69**	**0.54**	**0.48**	0.20
Cadmium	**0.51**	**0.57**	**0.58**	**0.52**	**0.65**	**0.68**	**0.56**	**0.48**	-0.11
Chromium	**0.73**	**0.52**	**0.67**	**0.76**	**0.80**	**0.58**	**0.74**	**0.67**	**0.78**
Copper	**0.89**	**0.79**	**0.76**	**0.76**	**0.86**	**0.75**	**0.71**	**0.66**	0.38
Lead	**0.87**	**0.71**	**0.81**	**0.84**	**0.92**	**0.74**	**0.80**	**0.75**	0.29
Mercury	**0.63**	**0.45**	**0.62**	**0.73**	**0.85**	**0.83**	**0.80**	**0.74**	**0.58**
Nickel	**0.62**	**0.52**	**0.67**	**0.72**	**0.77**	**0.70**	**0.74**	**0.64**	**0.56**
Selenium	0.13	0.15	-0.24	-0.26	0.00	0.04	0.05	0.13	-0.09
Silver	**0.74**	**0.60**	**0.71**	**0.79**	**0.87**	**0.68**	**0.80**	**0.72**	**0.59**
Zinc	**0.85**	**0.81**	**0.70**	**0.67**	**0.80**	**0.69**	**0.58**	**0.53**	0.14
Monobutyltin (MBT)	**0.93**	**0.86**	**0.73**	**0.77**	**0.78**	**0.54**	**0.63**	**0.58**	**0.48**
Dibutyltin (DBT)	**0.98**	**0.86**	**0.77**	**0.79**	**0.81**	**0.56**	**0.68**	**0.64**	**0.46**
Tributyltin (TBT)	**1.00**	**0.87**	**0.78**	**0.80**	**0.80**	**0.55**	**0.71**	**0.68**	0.43
Tetrabutyltin (TetBT)	**0.87**	**1.00**	**0.71**	**0.66**	**0.65**	**0.52**	**0.52**	**0.45**	0.18
LPAH	**0.78**	**0.71**	**1.00**	**0.96**	**0.78**	**0.61**	**0.72**	**0.63**	0.38
HPAH	**0.80**	**0.66**	**0.96**	**1.00**	**0.84**	**0.64**	**0.78**	**0.70**	**0.51**
PCBs (homologs)	**0.80**	**0.65**	**0.78**	**0.84**	**1.00**	**0.76**	**0.87**	**0.80**	0.40
PCTs	**0.55**	**0.52**	**0.61**	**0.64**	**0.76**	**1.00**	**0.64**	**0.58**	0.18
DRO	**0.71**	**0.52**	**0.72**	**0.78**	**0.87**	**0.64**	**1.00**	**0.96**	**0.47**
RRO	**0.68**	**0.45**	**0.63**	**0.70**	**0.80**	**0.58**	**0.96**	**1.00**	**0.45**
Percent fines	0.43	0.18	0.38	**0.51**	0.40	0.18	**0.47**	**0.45**	**1.00**

Note: Correlations computed using Phase 1 data from triad stations.
Tabled values are Pearson product-moment correlation coefficients calculated on natural log transformed data.
Bold values ($r > 0.434$) are statistically significant ($p = 0.95$).

DRO - diesel-range organics
HPAH - high-molecular-weight polycyclic aromatic hydrocarbon
LPAH - low-molecular-weight polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl
PCT - polychlorinated terphenyl
RRO - residual-range organics

Table 9-3. Stations sampled for pesticides in Phase 2

Station	Rationale
NA04	Low to moderate concentrations of metals and organics, but major differences in the benthic community when compared with the reference area
NA11	Low concentrations of metals and organics, but toxicity to amphipods
NA22	Low concentrations of metals and organics, but toxicity to bivalves
SW04	High concentrations of most metals, but no toxicity

Table 9-4. Correlations between pesticide concentrations and adverse biological effects

Chemical	Amphipod Survival	Echinoderm Fertilization	Bivalve Development	Benthic Macroinvertebrate Total Abundance	Benthic Macroinvertebrate Total Richness
alpha-Chlordane	0.70	**-0.42**	**-0.77**	-0.35	**-0.50**
gamma-Chlordane	0.70	**-0.42**	**-0.77**	-0.35	**-0.50**
4,4'-DDD	0.70	**-0.42**	**-0.77**	-0.35	**-0.50**
4,4'-DDE	0.70	**-0.42**	**-0.77**	-0.35	**-0.50**
4,4'-DDT	0.18	0.20	0.10	0.70	0.63

Note: Correlations computed using Phase 1 biological data and Phase 2 chemical from four triad stations.
Tabled values are Spearman rank correlation coefficients.
Bold values (r < –0.3877) are statistically significant (*p* = 0.95).

Table 9-5. Decision matrix for triad measurement of aquatic life effects

Likelihood of Toxicity	Benthic Community Differences from Reference	Likelihood of Beneficial Use Impairment
High	Major	Highly likely
High	Moderate	Highly likely
High	Minor	Likely
High	None	Possible
Medium	Major	Highly likely
Medium	Moderate	Likely
Medium	Minor	Possible
Medium	None	Possible
Low	Major	Highly likely
Low	Moderate	Likely
Low	Minor	Unlikely
Low	None	Unlikely

Table 9-6. Likelihood of toxicity based on combination of tests

Amphipod Survival	Echinoderm Fertilization	Bivalve Development	Likelihood of Toxicity
Yes	Yes	Yes	High
Yes	Yes	No	High
Yes	No	Yes	Medium
Yes	No	No	Medium
No	Yes	Yes	Medium
No	Yes	No	Low
No	No	Yes	Low
No	No	No	Low

Table 9-7. Potential effects on aquatic life beneficial use based on benthos and toxicity data

Station	Toxicity				Benthic Community Differences from Reference	Potential Beneficial Use Impairment
	Amphipod Survival	Echinoderm Fertilization	Bivalve Development	Likelihood of Toxicity		
NA01	No	No	No	Low	Minor	Unlikely
NA03	No	No	No	Low	None	Unlikely
NA04	No	No	No	Low	Major	Highly likely
NA05	No	No	No	Low	None	Unlikely
NA06	Yes	No	No	Medium	None	Possible
NA07	Yes	No	No	Medium	None	Possible
NA09	No	No	Yes	Medium	None	Possible
NA11	Yes	No	No	Medium	Minor	Possible
NA12	No	No	Yes	Medium	None	Possible
NA15	No	No	No	Low	Major	Highly likely
NA16	No	No	Yes	Medium	None	Possible
NA17	No	No	No	Low	None	Unlikely
NA19	No	No	Yes	Medium	None	Possible
NA20	No	No	No	Low	Major	Highly likely
NA22	No	No	Yes	Medium	Major	Highly likely
SW02	No	No	No	Low	Minor	Unlikely
SW03	No	No	No	Low	Moderate	Likely
SW04	No	No	No	Low	Minor	Unlikely
SW08	No	No	No	Low	None	Unlikely
SW09	No	No	No	Low	None	Unlikely
SW11	Yes	No	No	Medium	None	Possible
SW13	No	No	Yes	Medium	None	Possible
SW15	No	No	Yes	Medium	None	Possible
SW17	No	No	Yes	Medium	Moderate	Likely
SW18	Yes	No	No	Medium	None	Possible
SW21	No	No	No	Low	Major	Highly likely
SW22	No	No	Yes	Medium	Major	Highly likely
SW23	No	No	Yes	Medium	Major	Highly likely
SW25	No	No	Yes	Medium	None	Possible
SW27	Yes	No	Yes	High	None	Possible

Table 9-8. Relationships of sediment chemicals to biological effects

Chemical	Related to Amphipod toxicity	Echinoderm toxicity	Bivalve toxicity	Benthic macroinverte-brate total abundance	Benthic macroinverte-brate total richness	*Macoma* tissue bioaccumu-lation	Selected for derivation of a cleanup level
Arsenic	No	No	No	No	No	Yes[a]	Yes
Cadmium	No	No	No	No	No	No	No
Chromium	No	No	No	No	No	No	No
Copper	No	No	No	No	No	Yes	Yes
Lead	No	No	No	No	No	Yes	Yes
Mercury	No	No	No	No	No	Yes	Yes
Nickel	No	No	No	No	No	No	No
Silver	No	No	No	No	No	No	No
Zinc	No	No	No	No	No	Yes[a]	Yes
Tributyltin	No	No	No	No	No	Yes	Yes
HPAH	No	No	No	No	No	Yes	Yes
Total PCB homologs	No	No	No	No	No	Yes	Yes
Polychlorinated terphenyls	No	No	No	No	No	No	No
Diesel-range organics	No	No	No	No	Yes	--[b]	Yes
Residual-range organics	No	No	No	Yes	Yes	--[b]	Yes

Note: HPAH - high-molecular-weight polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

[a] The relationship is controlled by a single point

[b] Not evaluated

Table 9-9. Chemical ranges at triad stations

Analyte	Units[a]	Minimum Value	Maximum Value
Arsenic	mg/kg	6.6	96
Copper	mg/kg	96	1,900
Lead	mg/kg	53	480
Mercury	mg/kg	0.24	3.5
Zinc	mg/kg	190	4,600
Tributyltin	μg/kg	38	2,800
HPAH	μg/kg	2,000	26,000
Total PCB homologs	μg/kg	170	7,900
DRO	mg/kg	55 *U*	1,200
RRO	mg/kg	210 *U*	1,800

Note: DRO - diesel-range organics
HPAH - high-molecular-weight polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl
RRO - residual-range organics
U - undetected; half the quantitation limit shown

[a] All concentrations are on a dry-weight basis.

Table 9-10. Apparent effects thresholds

Chemical	Units[a]	Amphipod Survival	Echinoderm Fertilization	Bivalve Development	Any Benthic Difference[b]	Moderate to Major Benthic Difference	Exceedance of 95%UPL of Reference Pool BRI
Arsenic	mg/kg	96 *G*	96 *G*	96 *G*	27	96 *G*	96 *G*
Copper	mg/kg	1,900 *G*	1,900 *G*	1,900 *G*	1,000	1,900 *G*	1,900 *G*
Lead	mg/kg	480 *G*	480 *G*	480 *G*	250	480 *G*	480 *G*
Mercury (total)	mg/kg	3.9 *G*	3.9 *G*	3.9 *G*	3.2	3.9 *G*	2.5
Zinc	mg/kg	4,600 *G*	4,600 *G*	4,600 *G*	1,200	4,600 *G*	4,600 *G*
Tributyltin	µg/kg	2,800 *G*	2,800 *G*	2,800 *G*	1,900	2,800 *G*	2,800 *G*
HPAH	µg/kg	26,000 *G*	26,000 *G*	26,000 *G*	27,000 *G*	27,000 *G*	26,000 *G*
Total PCB homologs	µg/kg	8,100 *G*	8,100 *G*	8,100 *G*	3,000	8,100 *G*	5,800
DRO	mg/kg	1,100 *G*	1,100 *G*	1,100 *G*	490	1,200 *G*	490
RRO	mg/kg	1,800 *G*	1,800 *G*	1,800 *G*	1,300	1,800 *G*	1,300

Note: 95%UPL - 95 percent upper prediction limit
BRI - benthic response index
DRO - diesel-range organics
G - the true no-effect level may be greater than the level shown
HPAH - high-molecular-weight polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl
RRO - residual-range organics

[a] All concentrations are based on the dry weight of sediment.

[b] Also the lowest apparent effects threshold; see text.

Table 9-11. Performance measures for AETs

AET	Sensitivity (percent)	Specificity (percent)	Efficiency (percent)	Overall Reliability (percent)
Amphipod survival	0	100	NC [a]	70
Bivalve development	0	100	NC [a]	70
Moderate to major benthic community difference from reference	0	100	NC [a]	70
Any benthic community difference from reference (also LAET)	11	90	33	67
BRI AET	11	90	33	67

Note: AET - apparent effects threshold
BRI - benthic response index
LAET - lowest apparent effects threshold

[a] Cannot be calculated; no effects are predicted.

Table 9-12. Sediment concentrations predicted from pore water : sediment relationships

Chemical	Water Quality Criterion[a]	Predicted Sediment Concentration	Sediment Units	95 Percent Lower Confidence Limit on Sediment Prediction	95 Percent Upper Confidence Limit on Sediment Prediction
Copper	3.1 μg/L	90	mg/kg	-2,060	2,206
Lead	8.1 μg/L	180	mg/kg	-39	411
Mercury	940 ng/L	35	mg/kg	28	63
Zinc	81 μg/L	5,958	mg/kg	-2,802	50,519
PCB homologs	30 ng/L	215	μg/kg	-801	3,223

Note: PCB - polychlorinated biphenyl

[a] California Toxics Rule marine chronic criterion.

Table 9-13. Sum of 34 PAH compounds in pore water samples

Survey Station	Sum of 34 PAH Compounds (μg/L)
2231	0.359
2243	0.326 *U*
2433	0.310 *U*
2440	1.14
2441	0.341 *U*
NA01	0.615
NA06	0.485
NA13	0.419
NA16	0.361
NA17	0.397
SW01	0.906
SW02	367
SW04	0.991
SW08	1.79
SW24	1.82
SW25	0.740
SW28	0.590

Note: PAH - polycyclic aromatic hydrocarbon
U - undetected; half the quantitation limit shown

Table 10-1. Chemical concentrations in eelgrass samples used in wildlife receptor exposure modeling

		Reference		NASSCO Inside		Southwest Marine Inside	
Chemical	Units	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value
Total PCBs	μg/kg	30	30	48	48	123	123
Tributyltin	μg/kg	3.3 [a]	3.3 [a]	3.2 [a]	3.2 [a]	20	20
Arsenic	mg/kg	2.4	2.4	3.9	3.9	5.5	5.5
Cadmium	mg/kg	0.42	0.42	0.75	0.75	0.77	0.77
Chromium	mg/kg	2.4	2.4	9.7	9.7	18	18
Copper	mg/kg	32	32	195	195	209	209
Lead	mg/kg	4.5	4.5	19	19	25	25
Mercury (total)	mg/kg	0.030 [a]	0.030 [a]	0.13	0.13	0.26	0.26
Nickel	mg/kg	2.8	2.8	3.9	3.9	6.3	6.3
Selenium	mg/kg	0.60 [a]	0.60 [a]	0.65 [a]	0.65 [a]	0.65 [a]	0.65 [a]
Zinc	mg/kg	170	170	346	346	354	354
Total PAHs	μg/kg	506 [a]	506 [a]	877	877	2,665	2,665

Note: All values expressed on a dry weight basis.

PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

[a] Chemical undetected in all samples.

Table 10-2. Chemical concentrations in forage fish samples used in wildlife receptor exposure modeling

		Reference		NASSCO Inside		NASSCO Outside		Southwest Marine Inside		Southwest Marine Outside	
Chemical	Units	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value
Total PCBs	µg/kg	997	1,079	1,505	1,505	1,797	2,024	2,273	2,415	1,772	1,772
Tributyltin	µg/kg	32	41	28	28	41	51	65	76	122	122
Arsenic	mg/kg	2.4	2.5	2.5	2.5	2.9	3.2	3.3	3.6	3.9	3.9
Cadmium	mg/kg	0.031 [a]	0.032 [a]	0.031 [a]	0.031 [a]	0.019 [a]	0.020 [a]	0.022 [a]	0.033 [a]	0.020 [a]	0.020 [a]
Chromium	mg/kg	3.8	4.9	0.47 [a]	0.47 [a]	0.56 [a]	0.60 [a]	0.51 [a]	0.52 [a]	0.59 [a]	0.59 [a]
Copper	mg/kg	7.7	8.0	4.1	4.1	5.5	5.8	9.9	11	7.0	7.0
Lead	mg/kg	0.67	0.68	0.24	0.24	0.34	0.42	1.4	1.5	0.80	0.80
Mercury (total)	mg/kg	0.063	0.074	0.088	0.088	0.092	0.099	0.088	0.10	0.11	0.11
Nickel	mg/kg	2.1	2.5	0.56	0.56	0.58	0.83	0.84	0.96	0.67	0.67
Selenium	mg/kg	1.0	1.3	0.47 [a]	0.47 [a]	0.56 [a]	0.60 [a]	0.51 [a]	0.52 [a]	0.59 [a]	0.59 [a]
Zinc	mg/kg	117	123	141	141	175	188	142	150	149	149
Total PAHs	µg/kg	261 [a]	269 [a]	266 [a]	266 [a]	317 [a]	337 [a]	326	331	335 [a]	335 [a]

Note: All values expressed on a dry weight basis.

PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

[a] Chemical undetected in all samples.

Table 10-3. Chemical concentrations in spotted sandbass samples used in wildlife receptor exposure modeling

		Reference		NASSCO Inside		NASSCO Outside		Southwest Marine Inside		Southwest Marine Outside	
Chemical	Units	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value
Total PCBs	μg/kg	1,425	1,911	3,763	8,108	1,711	2,129	4,009	8,170	2,424	4,025
Tributyltin	μg/kg	41	82	83	160	161	213	129	258	126	182
Arsenic	mg/kg	1.7	2.0	1.9	2.5	2.7	3.6	2.3	2.6	2.5	3.4
Cadmium	mg/kg	0.090	0.12	0.084	0.084	0.074	0.080	0.11	0.16	0.066	0.084
Chromium	mg/kg	0.54 [a]	0.59 [a]	0.52 [a]	0.59 [a]	2.7	2.7	1.6	1.6	0.49 [a]	0.57 [a]
Copper	mg/kg	3.4	7.1	3.9	8.7	4.6	9.4	9.0	27	6.3	12
Lead	mg/kg	0.31	0.51	0.74	1.0	0.84	1.8	0.99	1.4	0.68	1.3
Mercury (total)	mg/kg	0.42	0.62	0.62	0.75	0.58	0.81	0.52	0.66	0.54	0.71
Nickel	mg/kg	0.88	0.92	0.99	1.2	1.1	1.7	1.1	1.3	0.96	1.4
Selenium	mg/kg	0.54 [a]	0.59 [a]	1.8	2.4	1.6	2.4	2.5	3.7	1.2	1.3
Zinc	mg/kg	53	58	54	66	60	85	60	81	48	53
Total PAHs	μg/kg	308 [a]	336 [a]	340 [a]	357 [a]	321 [a]	340 [a]	349 [a]	379 [a]	325 [a]	359 [a]

Note: All values expressed on a dry weight basis.

PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

[a] Chemical undetected in all samples.

Table 10-4. Chemical concentrations in mussel samples used in wildlife receptor exposure modeling

		Reference		NASSCO Inside		Southwest Marine Inside	
Chemical	Units	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value
Total PCBs	μg/kg	722	722	600	700	861	933
Tributyltin	μg/kg	144	144	425	470	521	547
Arsenic	mg/kg	9.4	9.4	15	15	16	17
Cadmium	mg/kg	0.29	0.29	0.34	0.37	0.38	0.44
Chromium	mg/kg	4.3	4.3	4.5	6.0	2.6	2.6
Copper	mg/kg	24	24	65	80	48	51
Lead	mg/kg	2.9	2.9	5.5	6.5	4.3	4.3
Mercury (total)	mg/kg	0.072	0.072	0.11	0.12	0.10	0.11
Nickel	mg/kg	3.4	3.4	7.5	9.5	3.8	3.9
Selenium	mg/kg	3.6	3.6	3.3	3.5	3.8	4.0
Zinc	mg/kg	72	72	90	100	101	107
Total PAHs	μg/kg	778	778	1,775	2,300	4,814	4,895

Note: All values expressed on a dry weight basis.

PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-5. Chemical concentrations in sediment samples used in wildlife receptor exposure modeling

		Original Reference		Final Reference Pool		NASSCO Inside		NASSCO Outside		Southwest Marine Inside		Southwest Marine Outside	
Chemical	Units	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value	Mean Detected Value	Maximum Detected Value
Total PCBs	μg/kg	100	270	90	270	600	1,700	250	380	1,900	7,100	250	330
Tributyltin	μg/kg	7.0	35	8.0	35	200	1,400	90	410	500	3,300	37	49
Arsenic	mg/kg	4.7	8.8	5.8	9.1	11	18	8.0	11	15	73	8.0	10
Cadmium	mg/kg	0.15	0.31	0.14	0.31	0.31	0.46	0.20	0.39	0.60	2.9	0.13	0.21
Chromium	mg/kg	25	59	34	67	70	100	45	67	70	110	44	70
Copper	mg/kg	50	98	60	157	250	510	120	180	400	1,500	140	320
Lead	mg/kg	25	68	28	64	90	130	59	83	120	430	59	99
Mercury (total)	mg/kg	0.24	0.46	0.28	0.63	0.90	2.3	0.55	0.71	1.2	4.2	0.70	2.1
Nickel	mg/kg	6.0	12	10	19	17	27	12	18	20	100	11	13
Selenium	mg/kg	0.80	1.0	0.30	1.0	1.0	1.3	1.0	1.0	1.2	1.5	1.2	1.2
Zinc	mg/kg	100	260	120	233	320	620	190	290	500	3,400	190	310
Total PAHs	μg/kg	500	2,500	700	3,200	4,000	16,000	1,600	3,000	14,000	57,000	1,900	4,300

Note: All values expressed on a dry weight basis.

PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-6. Exposure parameters for wildlife receptors

Receptor	Body Weight (kg)	Food Ingestion Rate (kg/day dry wt)	Sediment Ingestion Rate (kg/day dry wt)	Area Use Factor	Time Use Factor	Dietary Composition
Piscivorous Birds						
California brown pelican	3.174 [a]	0.25 [b]	0.005 [c]	0.01	1	100 percent medium-sized fish
California least tern	0.045 [d]	0.0053 [e]	0.00011 [c]	0.01	1	100 percent small fish
Western grebe	1.2 [f]	0.062 [e]	0.0031 [g]	0.01	1	100 percent small fish
Mollusc-Eating Bird						
Surf scoter	1.05 [h]	0.056 [e]	0.0028 [g]	0.01	1	100 percent molluscs
Marine Mammal						
California sea lion	75 [i]	1.54 [j]	0.0308 [c]	0.01	1	100 percent medium-sized fish
Marine Reptile						
East Pacific green turtle	95.0 [k]	0.35 [l]	0.0186 [m]	0.01	1	100 percent eelgrass

[a] Mean female weight from Dunning (1993).

[b] Based on Nagy et al. (1999) equation for Pelecaniformes.

[c] Based on minimum percentage soil in wildlife diets reported in Beyer et al. (1994).

[d] Mean adult body weight from Thompson et al. (1997).

[e] Based on U.S. EPA (1993a) equation for non-passerine birds.

[f] Mean female body weight from Storer and Neuchterlein (1992).

[g] Estimated using the range of percentage soil in the diets of duck species reported in Beyer et al. (1994).

[h] Mean female wintering weight in California from Savard et al. (1998).

[i] Median female weight from Peterson and Bartholomew (1967).

[j] Based on Nagy et al. (1999) equation for Carnivora.

[k] Median adult female weight, as cited in NMFS and FWS (1998).

[l] Cited in Bjorndal (1985).

[m] Estimated using the range of percentage soil in the diet of eastern painted turtle reported in Beyer et al. (1994).

Table 10-7. Species-specific area use factors for aquatic-dependent wildlife

Receptor	Extent of Receptor's Preferred Habitat in San Diego Bay (acres)	Assessment Unit Area as a Proportion of Preferred Habitat [a]			
		Inside NASSCO Leasehold	Outside NASSCO Leasehold	Inside Southwest Marine Leasehold	Outside Southwest Marine Leasehold
East Pacific green turtle	3,734	0.011	0.016	0.006	0.006
California least tern	13,374	0.003	0.004	0.002	0.002
California brown pelican	11,219	0.004	0.005	0.002	0.002
Western grebe	11,219	0.004	0.005	0.002	0.002
Surf scoter	11,375	0.004	0.005	0.002	0.002
California sea lion	10,396	0.004	0.006	0.002	0.002

[a] Acreages of areas inside the NASSCO leasehold, outside the NASSCO leasehold, inside the Southwest Marine leasehold, and outside the Southwest Marine leasehold are 40.3, 58.8, 21.0, and 22.8 acres, respectively.

Table 10-8. Toxicity reference values for risk evaluation for wildlife receptors

	TRVs (all in mg/kg-day)			
	Avian [a]		Mammalian	
	NOAEL	LOAEL	NOAEL	LOAEL
Polychlorinated biphenyls	0.41	1.8	0.14	0.27
Tributyltin	6.8	17	23	35
Metals				
Arsenic	20	50	0.13	1.3
Cadmium	1.5	20	1.0	10
Chromium	0.86	4.3	3.3	69
Copper	47	62	12	15
Lead	3.9	NA	11.0	90
Mercury	0.032	0.064	0.0	0.16
Nickel	77	110	40	80
Selenium	0.40	0.80	0.20	0.33
Zinc	130	NA	160	320
Polycyclic aromatic hydrocarbons				
(benzo[a]pyrene)	0.14	1.4	1.0	10

Note: LOAEL - lowest-observed-adverse-effect level
NA - not available
NOAEL - no-observed-adverse-effect level
TRV - toxicity reference value

[a] Avian TRVs also used for reptilian receptors.

Table 10-9. Hazard quotients for receptors occurring at the original reference area calculated using average chemical concentrations and species-specific area-use factors

	Brown Pelican		Least Tern		Western Grebe		Surf Scoter		California Sea Lion		Pacific Green Turtle	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	2.7E-03	6.2E-04	2.9E-03	6.5E-04	1.3E-03	2.9E-04	9.5E-04	2.2E-04	2.1E-03	1.1E-03	5.7E-06	1.3E-06
Tributyltin	4.7E-06	1.9E-06	5.6E-06	2.3E-06	2.5E-06	9.9E-07	1.1E-05	4.5E-06	3.6E-07	2.4E-07	3.5E-08	1.4E-08
Metals												
Arsenic	6.9E-05	2.8E-05	1.4E-04	5.8E-05	6.7E-05	2.7E-05	2.6E-04	1.0E-04	2.8E-03	2.8E-04	8.7E-06	3.5E-06
Cadmium	4.9E-05	3.7E-06	2.6E-05	2.0E-06	1.3E-05	9.9E-07	1.1E-04	7.9E-06	1.9E-05	1.9E-06	1.9E-05	1.4E-06
Chromium	9.6E-04	1.9E-04	5.9E-03	1.2E-03	3.0E-03	6.0E-04	3.4E-03	6.9E-04	6.5E-05	3.1E-06	2.9E-04	5.9E-05
Copper	7.4E-05	5.6E-05	2.2E-04	1.6E-04	1.1E-04	8.5E-05	3.1E-04	2.3E-04	7.5E-05	6.0E-05	4.9E-05	3.7E-05
Lead	1.6E-04	--	3.5E-04	--	2.5E-04	--	5.7E-04	--	1.5E-05	1.9E-06	1.0E-04	--
Mercury (total)	1.1E-02	5.3E-03	2.5E-03	1.3E-03	1.2E-03	6.1E-04	1.4E-03	7.0E-04	2.8E-03	5.5E-04	9.0E-05	4.5E-05
Nickel	1.0E-05	7.2E-06	3.4E-05	2.4E-05	1.6E-05	1.1E-05	2.6E-05	1.8E-05	5.2E-06	2.6E-06	2.7E-06	1.9E-06
Selenium	1.1E-03	5.5E-04	3.1E-03	1.5E-03	1.4E-03	6.9E-04	4.9E-03	2.4E-03	5.8E-04	3.5E-04	1.0E-04	5.2E-05
Zinc	3.3E-04	--	1.1E-03	--	4.9E-04	--	3.2E-04	--	7.0E-05	3.5E-05	8.8E-05	--
Total PAHs	1.8E-03	1.8E-04	2.3E-03	2.3E-04	1.1E-03	1.1E-04	3.1E-03	3.1E-04	6.5E-05	6.5E-06	2.5E-04	2.5E-05

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-10. Hazard quotients for receptors occurring at the final reference pool area calculated using average chemical concentrations and species-specific area-use factors

	Brown Pelican		Least Tern		Western Grebe		Surf Scoter		California Sea Lion		Pacific Green Turtle	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	2.7E-03	6.2E-04	2.9E-03	6.5E-04	1.3E-03	2.9E-04	9.5E-04	2.2E-04	2.1E-03	1.1E-03	5.6E-06	1.3E-06
Tributyltin	4.7E-06	1.9E-06	5.6E-06	2.3E-06	2.5E-06	1.0E-06	1.1E-05	4.5E-06	3.6E-07	2.4E-07	3.6E-08	1.4E-08
Metals												
Arsenic	7.0E-05	2.8E-05	1.5E-04	5.8E-05	6.8E-05	2.7E-05	2.6E-04	1.0E-04	2.8E-03	2.8E-04	8.9E-06	3.6E-06
Cadmium	4.9E-05	3.7E-06	2.6E-05	2.0E-06	1.3E-05	9.7E-07	1.1E-04	7.9E-06	1.9E-05	1.9E-06	1.9E-05	1.4E-06
Chromium	1.1E-03	2.2E-04	6.1E-03	1.2E-03	3.3E-03	6.6E-04	3.7E-03	7.4E-04	7.6E-05	3.6E-06	3.4E-04	6.7E-05
Copper	7.7E-05	5.8E-05	2.2E-04	1.7E-04	1.2E-04	8.9E-05	3.1E-04	2.4E-04	7.9E-05	6.3E-05	5.0E-05	3.8E-05
Lead	1.8E-04	--	3.7E-04	--	2.7E-04	--	5.9E-04	--	1.6E-05	2.0E-06	1.0E-04	--
Mercury (total)	1.1E-02	5.3E-03	2.5E-03	1.3E-03	1.2E-03	6.2E-04	1.4E-03	7.2E-04	2.8E-03	5.5E-04	9.5E-05	4.7E-05
Nickel	1.1E-05	7.8E-06	3.5E-05	2.5E-05	1.7E-05	1.2E-05	2.7E-05	1.9E-05	5.6E-06	2.8E-06	2.9E-06	2.0E-06
Selenium	1.1E-03	5.4E-04	3.0E-03	1.5E-03	1.3E-03	6.7E-04	4.8E-03	2.4E-03	5.6E-04	3.4E-04	1.0E-04	5.0E-05
Zinc	3.3E-04	--	1.1E-03	--	4.9E-04	--	3.2E-04	--	7.1E-05	3.5E-05	8.9E-05	--
Total PAHs	1.8E-03	1.8E-04	2.3E-03	2.3E-04	1.1E-03	1.1E-04	3.1E-03	3.1E-04	6.6E-05	6.6E-06	2.6E-04	2.6E-05

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-11. Hazard quotients for receptors occurring inside the NASSCO leasehold calculated using average chemical concentrations and species-specific area-use factors

	Brown Pelican		Least Tern		Western Grebe		Surf Scoter		California Sea Lion		Pacific Green Turtle	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	7.3E-03	1.7E-03	4.4E-03	9.9E-04	1.9E-03	4.4E-04	8.2E-04	1.9E-04	5.5E-03	2.9E-03	1.3E-05	3.1E-06
Tributyltin	1.0E-05	4.0E-06	5.6E-06	2.2E-06	2.9E-06	1.2E-06	3.4E-05	1.4E-05	7.8E-07	5.1E-07	1.5E-07	5.8E-08
Metals												
Arsenic	8.5E-05	3.4E-05	1.6E-04	6.4E-05	7.9E-05	3.2E-05	4.1E-04	1.7E-04	3.4E-03	3.4E-04	1.5E-05	5.9E-06
Cadmium	4.7E-05	3.6E-06	2.9E-05	2.2E-06	1.6E-05	1.2E-06	1.3E-04	9.5E-06	1.9E-05	1.9E-06	3.3E-05	2.5E-06
Chromium	1.8E-03	3.5E-04	2.6E-03	5.1E-04	2.4E-03	4.8E-04	5.0E-03	9.9E-04	1.2E-04	5.7E-06	1.1E-03	2.1E-04
Copper	1.5E-04	1.1E-04	2.3E-04	1.7E-04	1.8E-04	1.4E-04	8.8E-04	6.7E-04	1.5E-04	1.2E-04	2.9E-04	2.2E-04
Lead	5.1E-04	--	6.2E-04	--	6.3E-04	--	1.4E-03	--	4.7E-05	5.8E-06	4.1E-04	--
Mercury (total)	1.6E-02	7.9E-03	3.9E-03	1.9E-03	2.1E-03	1.1E-03	2.6E-03	1.3E-03	4.1E-03	8.3E-04	3.7E-04	1.9E-04
Nickel	1.4E-05	9.5E-06	1.4E-05	9.7E-06	9.5E-06	6.6E-06	5.8E-05	4.0E-05	6.8E-06	3.4E-06	4.1E-06	2.9E-06
Selenium	3.5E-03	1.8E-03	1.4E-03	7.2E-04	6.7E-04	3.4E-04	4.4E-03	2.2E-03	1.8E-03	1.1E-03	1.2E-04	5.8E-05
Zinc	3.6E-04	--	1.3E-03	--	6.2E-04	--	4.3E-04	--	7.7E-05	3.9E-05	1.8E-04	--
Total PAHs	2.4E-03	2.4E-04	2.9E-03	2.9E-04	1.7E-03	1.7E-04	7.5E-03	7.5E-04	8.6E-05	8.6E-06	5.2E-04	5.2E-05

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-12. Hazard quotients for receptors occurring outside the NASSCO leasehold calculated using average chemical concentrations and species-specific area-use factors

	Brown Pelican		Least Tern		Western Grebe		California Sea Lion	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	3.3E-03	7.5E-04	5.2E-03	1.2E-03	2.3E-03	5.2E-04	2.5E-03	1.3E-03
Tributyltin	1.9E-05	7.6E-06	7.4E-06	3.0E-06	3.5E-06	1.4E-06	1.5E-06	9.6E-07
Metals								
Arsenic	1.1E-04	4.5E-05	1.8E-04	7.2E-05	8.5E-05	3.4E-05	4.6E-03	4.6E-04
Cadmium	4.1E-05	3.1E-06	1.8E-05	1.3E-06	9.9E-06	7.4E-07	1.6E-05	1.6E-06
Chromium	3.3E-03	6.5E-04	2.0E-03	4.0E-04	1.7E-03	3.4E-04	2.2E-04	1.1E-05
Copper	1.2E-04	8.9E-05	2.0E-04	1.5E-04	1.3E-04	9.6E-05	1.2E-04	9.6E-05
Lead	4.1E-04	--	4.6E-04	--	4.4E-04	--	3.8E-05	4.6E-06
Mercury (total)	1.4E-02	7.2E-03	3.8E-03	1.9E-03	1.9E-03	9.7E-04	3.8E-03	7.5E-04
Nickel	1.4E-05	9.8E-06	1.3E-05	8.8E-06	7.9E-06	5.5E-06	7.1E-06	3.5E-06
Selenium	3.2E-03	1.6E-03	1.7E-03	8.5E-04	7.9E-04	3.9E-04	1.6E-03	1.0E-03
Zinc	3.9E-04	--	1.6E-03	--	7.3E-04	--	8.2E-05	4.1E-05
Total PAHs	2.0E-03	2.0E-04	2.9E-03	2.9E-04	1.5E-03	1.5E-04	7.2E-05	7.2E-06

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-13. Hazard quotients for receptors occurring inside the Southwest Marine leasehold calculated using average chemical concentrations and species-specific area-use factors

	Brown Pelican		Least Tern		Western Grebe		Surf Scoter		California Sea Lion		Pacific Green Turtle	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	7.8E-03	1.8E-03	6.6E-03	1.5E-03	3.0E-03	6.8E-04	1.2E-03	2.8E-04	5.9E-03	3.1E-03	3.8E-05	8.6E-06
Tributyltin	1.6E-05	6.5E-06	1.3E-05	5.2E-06	6.9E-06	2.8E-06	4.3E-05	1.7E-05	1.2E-06	8.2E-07	4.8E-07	1.9E-07
Metals												
Arsenic	1.0E-04	4.1E-05	2.1E-04	8.6E-05	1.1E-04	4.2E-05	4.5E-04	1.8E-04	4.1E-03	4.1E-04	2.1E-05	8.3E-06
Cadmium	6.3E-05	4.7E-06	2.7E-05	2.0E-06	1.8E-05	1.4E-06	1.5E-04	1.1E-05	2.5E-05	2.5E-06	3.5E-05	2.6E-06
Chromium	2.7E-03	5.5E-04	2.6E-03	5.2E-04	2.4E-03	4.8E-04	3.8E-03	7.5E-04	1.9E-04	8.9E-06	1.7E-03	3.4E-04
Copper	2.8E-04	2.2E-04	4.5E-04	3.4E-04	3.3E-04	2.5E-04	7.7E-04	5.8E-04	2.9E-04	2.3E-04	3.2E-04	2.5E-04
Lead	6.8E-04	--	1.1E-03	--	9.7E-04	--	1.4E-03	--	6.3E-05	7.7E-06	5.3E-04	--
Mercury (total)	1.3E-02	6.7E-03	4.1E-03	2.1E-03	2.4E-03	1.2E-03	2.7E-03	1.4E-03	3.5E-03	7.0E-04	6.7E-04	3.4E-04
Nickel	1.5E-05	1.1E-05	1.9E-05	1.3E-05	1.2E-05	8.6E-06	3.3E-05	2.3E-05	7.8E-06	3.9E-06	6.3E-06	4.4E-06
Selenium	5.0E-03	2.5E-03	1.6E-03	7.8E-04	7.3E-04	3.7E-04	5.2E-03	2.6E-03	2.6E-03	1.6E-03	1.2E-04	5.9E-05
Zinc	4.2E-04	--	1.4E-03	--	6.6E-04	--	5.2E-04	--	9.0E-05	4.5E-05	1.9E-04	--
Total PAHs	3.5E-03	3.5E-04	5.1E-03	5.1E-04	3.8E-03	3.8E-04	2.1E-02	2.1E-03	1.3E-04	1.3E-05	1.6E-03	1.6E-04

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-14. Hazard quotients for receptors occurring outside the Southwest Marine leasehold calculated using average chemical concentrations and species-specific area-use factors

	Brown Pelican		Least Tern		Western Grebe		California Sea Lion	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	4.7E-03	1.1E-03	5.1E-03	1.2E-03	2.2E-03	5.1E-04	3.6E-03	1.8E-03
Tributyltin	1.5E-05	5.9E-06	2.1E-05	8.5E-06	9.4E-06	3.8E-06	1.1E-06	7.4E-07
Metals								
Arsenic	1.1E-04	4.2E-05	2.4E-04	9.7E-05	1.1E-04	4.5E-05	4.2E-03	4.2E-04
Cadmium	3.6E-05	2.7E-06	1.7E-05	1.3E-06	9.0E-06	6.8E-07	1.4E-05	1.4E-06
Chromium	1.3E-03	2.5E-04	2.0E-03	4.0E-04	1.7E-03	3.4E-04	8.5E-05	4.1E-06
Copper	1.5E-04	1.2E-04	2.5E-04	1.9E-04	1.5E-04	1.2E-04	1.6E-04	1.2E-04
Lead	3.8E-04	--	6.0E-04	--	5.0E-04	--	3.5E-05	4.3E-06
Mercury (total)	1.4E-02	6.9E-03	4.6E-03	2.3E-03	2.3E-03	1.2E-03	3.6E-03	7.2E-04
Nickel	1.2E-05	8.5E-06	1.4E-05	9.5E-06	8.2E-06	5.7E-06	6.1E-06	3.0E-06
Selenium	2.3E-03	1.2E-03	1.8E-03	9.0E-04	8.4E-04	4.2E-04	1.2E-03	7.3E-04
Zinc	3.2E-04	--	1.4E-03	--	6.3E-04	--	6.7E-05	3.3E-05
Total PAHs	2.0E-03	2.0E-04	3.1E-03	3.1E-04	1.6E-03	1.6E-04	7.4E-05	7.4E-06

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-15. Hypothetical hazard quotients for receptors occurring at the original reference area calculated using maximum chemical concentrations and species-specific area-use factors

	Brown Pelican		Least Tern		Western Grebe		Surf Scoter		California Sea Lion		Pacific Green Turtle	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	3.7E-03	8.4E-04	3.1E-03	7.1E-04	1.4E-03	3.1E-04	9.6E-04	2.2E-04	2.8E-03	1.4E-03	7.3E-06	1.7E-06
Tributyltin	9.6E-06	3.8E-06	7.2E-06	2.9E-06	3.3E-06	1.3E-06	1.1E-05	4.5E-06	7.4E-07	4.8E-07	5.2E-08	2.1E-08
Metals												
Arsenic	8.5E-05	3.4E-05	1.6E-04	6.4E-05	7.7E-05	3.1E-05	2.6E-04	1.1E-04	3.4E-03	3.4E-04	9.5E-06	3.8E-06
Cadmium	6.5E-05	4.9E-06	3.0E-05	2.2E-06	1.6E-05	1.2E-06	1.1E-04	8.1E-06	2.5E-05	2.5E-06	1.9E-05	1.4E-06
Chromium	1.6E-03	3.2E-04	8.4E-03	1.7E-03	4.7E-03	9.5E-04	4.5E-03	9.0E-04	1.1E-04	5.3E-06	4.5E-04	9.0E-05
Copper	1.5E-04	1.2E-04	2.5E-04	1.9E-04	1.4E-04	1.1E-04	3.3E-04	2.5E-04	1.6E-04	1.2E-04	5.3E-05	4.0E-05
Lead	3.8E-04	--	6.2E-04	--	5.4E-04	--	8.7E-04	--	3.5E-05	4.3E-06	1.4E-04	--
Mercury (total)	1.6E-02	7.8E-03	3.1E-03	1.5E-03	1.6E-03	7.8E-04	1.6E-03	7.9E-04	4.1E-03	8.1E-04	1.2E-04	5.8E-05
Nickel	1.2E-05	8.3E-06	4.1E-05	2.9E-05	2.1E-05	1.4E-05	2.8E-05	1.9E-05	6.0E-06	3.0E-06	3.0E-06	2.1E-06
Selenium	1.2E-03	6.0E-04	3.8E-03	1.9E-03	1.7E-03	8.5E-04	4.9E-03	2.4E-03	6.3E-04	3.8E-04	1.1E-04	5.3E-05
Zinc	3.8E-04	--	1.2E-03	--	5.4E-04	--	3.5E-04	--	8.1E-05	4.1E-05	9.3E-05	--
Total PAHs	2.2E-03	2.2E-04	2.7E-03	2.7E-04	1.5E-03	1.5E-04	3.4E-03	3.4E-04	7.9E-05	7.9E-06	3.1E-04	3.1E-05

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-16. Hypothetical hazard quotients for receptors occurring at the revised reference area calculated using maximum chemical concentrations and species-specific area-use factors

	Brown Pelican		Least Tern		Western Grebe		Surf Scoter		California Sea Lion		Pacific Green Turtle	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	3.7E-03	8.4E-04	3.1E-03	7.1E-04	1.4E-03	3.1E-04	9.6E-04	2.2E-04	2.8E-03	1.4E-03	7.3E-06	1.7E-06
Tributyltin	9.6E-06	3.8E-06	7.2E-06	2.9E-06	3.3E-06	1.3E-06	1.1E-05	4.5E-06	7.4E-07	4.8E-07	5.2E-08	2.1E-08
Metals												
Arsenic	8.5E-05	3.4E-05	1.6E-04	6.4E-05	7.7E-05	3.1E-05	2.6E-04	1.1E-04	3.4E-03	3.4E-04	9.5E-06	3.8E-06
Cadmium	6.5E-05	4.9E-06	3.0E-05	2.2E-06	1.6E-05	1.2E-06	1.1E-04	8.1E-06	2.5E-05	2.5E-06	1.9E-05	1.4E-06
Chromium	1.8E-03	3.5E-04	8.6E-03	1.7E-03	5.0E-03	9.9E-04	4.7E-03	9.4E-04	1.2E-04	5.7E-06	4.8E-04	9.7E-05
Copper	1.7E-04	1.3E-04	2.8E-04	2.1E-04	1.7E-04	1.3E-04	3.7E-04	2.8E-04	1.8E-04	1.4E-04	5.8E-05	4.4E-05
Lead	3.6E-04	--	5.9E-04	--	5.1E-04	--	8.4E-04	--	3.3E-05	4.1E-06	1.4E-04	--
Mercury (total)	1.6E-02	7.8E-03	3.2E-03	1.6E-03	1.7E-03	8.5E-04	1.7E-03	8.7E-04	4.1E-03	8.2E-04	1.4E-04	6.9E-05
Nickel	1.3E-05	9.3E-06	4.3E-05	3.0E-05	2.3E-05	1.6E-05	3.0E-05	2.1E-05	6.7E-06	3.3E-06	3.3E-06	2.3E-06
Selenium	1.2E-03	6.0E-04	3.8E-03	1.9E-03	1.7E-03	8.5E-04	4.9E-03	2.4E-03	6.3E-04	3.8E-04	1.1E-04	5.3E-05
Zinc	3.8E-04	--	1.2E-03	--	5.3E-04	--	3.4E-04	--	8.0E-05	4.0E-05	9.2E-05	--
Total PAHs	2.3E-03	2.3E-04	2.8E-03	2.8E-04	1.6E-03	1.6E-04	3.6E-03	3.6E-04	8.2E-05	8.2E-06	3.3E-04	3.3E-05

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-17. Hypothetical hazard quotients for receptors occurring inside the NASSCO leasehold calculated using maximum chemical concentrations and species-specific area-use factors

	Brown Pelican		Least Tern		Western Grebe		Surf Scoter		California Sea Lion		Pacific Green Turtle	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	1.6E-02	3.6E-03	4.4E-03	1.0E-03	2.0E-03	4.6E-04	1.0E-03	2.3E-04	1.2E-02	6.2E-03	2.4E-05	5.5E-06
Tributyltin	2.2E-05	8.7E-06	9.7E-06	3.9E-06	7.5E-06	3.0E-06	4.2E-05	1.7E-05	1.7E-06	1.1E-06	8.4E-07	3.3E-07
Metals												
Arsenic	1.1E-04	4.5E-05	1.7E-04	6.8E-05	8.8E-05	3.5E-05	4.2E-04	1.7E-04	4.6E-03	4.6E-04	1.6E-05	6.5E-06
Cadmium	4.9E-05	3.7E-06	3.2E-05	2.4E-06	1.9E-05	1.4E-06	1.4E-04	1.0E-05	1.9E-05	1.9E-06	3.4E-05	2.5E-06
Chromium	2.4E-03	4.7E-04	3.4E-03	6.8E-04	3.3E-03	6.6E-04	6.8E-03	1.4E-03	1.6E-04	7.7E-06	1.2E-03	2.4E-04
Copper	3.2E-04	2.4E-04	3.6E-04	2.7E-04	3.3E-04	2.5E-04	1.2E-03	9.1E-04	3.2E-04	2.6E-04	3.1E-04	2.4E-04
Lead	7.3E-04	--	8.6E-04	--	8.9E-04	--	1.8E-03	--	6.7E-05	8.2E-06	4.5E-04	--
Mercury (total)	2.0E-02	9.8E-03	4.9E-03	2.5E-03	3.3E-03	1.6E-03	3.9E-03	2.0E-03	5.1E-03	1.0E-03	5.4E-04	2.7E-04
Nickel	1.8E-05	1.3E-05	1.7E-05	1.2E-05	1.3E-05	9.0E-06	7.5E-05	5.3E-05	9.1E-06	4.6E-06	4.7E-06	3.3E-06
Selenium	4.7E-03	2.4E-03	1.5E-03	7.3E-04	6.9E-04	3.5E-04	4.8E-03	2.4E-03	2.5E-03	1.5E-03	1.2E-04	5.9E-05
Zinc	4.7E-04	--	1.4E-03	--	6.8E-04	--	5.4E-04	--	1.0E-04	5.0E-05	1.9E-04	--
Total PAHs	3.8E-03	3.8E-04	4.9E-03	4.9E-04	3.9E-03	3.9E-04	1.2E-02	1.2E-03	1.4E-04	1.4E-05	8.6E-04	8.6E-05

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-18. Hypothetical hazard quotients for receptors occurring outside the NASSCO leasehold calculated using maximum chemical concentrations and species-specific area-use factors

	Brown Pelican		Least Tern		Western Grebe		California Sea Lion	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	4.1E-03	9.4E-04	5.8E-03	1.3E-03	2.6E-03	5.9E-04	3.1E-03	1.6E-03
Tributyltin	2.6E-05	1.0E-05	1.0E-05	4.1E-06	5.5E-06	2.2E-06	2.0E-06	1.3E-06
Metals								
Arsenic	1.5E-04	6.0E-05	2.0E-04	8.0E-05	9.6E-05	3.8E-05	6.0E-03	6.0E-04
Cadmium	4.6E-05	3.4E-06	2.2E-05	1.6E-06	1.4E-05	1.0E-06	1.8E-05	1.8E-06
Chromium	3.7E-03	7.3E-04	2.7E-03	5.3E-04	2.4E-03	4.7E-04	2.5E-04	1.2E-05
Copper	2.2E-04	1.7E-04	2.3E-04	1.8E-04	1.6E-04	1.2E-04	2.2E-04	1.8E-04
Lead	7.1E-04	--	6.3E-04	--	6.1E-04	--	6.5E-05	8.0E-06
Mercury (total)	2.0E-02	1.0E-02	4.2E-03	2.1E-03	2.2E-03	1.1E-03	5.3E-03	1.1E-03
Nickel	2.1E-05	1.5E-05	1.8E-05	1.3E-05	1.2E-05	8.1E-06	1.0E-05	5.2E-06
Selenium	4.8E-03	2.4E-03	1.8E-03	9.1E-04	8.3E-04	4.2E-04	2.5E-03	1.5E-03
Zinc	5.5E-04	--	1.8E-03	--	8.0E-04	--	1.2E-04	5.8E-05
Total PAHs	2.3E-03	2.3E-04	3.3E-03	3.3E-04	1.8E-03	1.8E-04	8.2E-05	8.2E-06

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-19. Hypothetical hazard quotients for receptors occurring inside the Southwest Marine leasehold calculated using maximum chemical concentrations and species-specific area-use factors

	Brown Pelican		Least Tern		Western Grebe		Surf Scoter		California Sea Lion		Pacific Green Turtle	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	1.6E-02	3.6E-03	7.3E-03	1.7E-03	3.5E-03	8.0E-04	1.7E-03	3.8E-04	1.2E-02	6.3E-03	8.7E-05	2.0E-05
Tributyltin	3.8E-05	1.5E-05	2.5E-05	9.9E-06	1.8E-05	7.3E-06	5.6E-05	2.2E-05	2.9E-06	1.9E-06	2.1E-06	8.4E-07
Metals												
Arsenic	1.6E-04	6.3E-05	3.0E-04	1.2E-04	1.9E-04	7.5E-05	5.6E-04	2.2E-04	6.4E-03	6.4E-04	3.2E-05	1.3E-05
Cadmium	1.2E-04	8.7E-06	7.2E-05	5.4E-06	6.1E-05	4.6E-06	2.1E-04	1.6E-05	4.6E-05	4.6E-06	4.1E-05	3.1E-06
Chromium	3.5E-03	6.9E-04	3.7E-03	7.4E-04	3.6E-03	7.2E-04	5.0E-03	1.0E-03	2.4E-04	1.1E-05	1.9E-03	3.7E-04
Copper	9.6E-04	7.3E-04	1.0E-03	7.8E-04	9.5E-04	7.2E-04	1.4E-03	1.1E-03	9.8E-04	7.8E-04	4.2E-04	3.1E-04
Lead	2.0E-03	--	3.1E-03	--	3.0E-03	--	3.5E-03	--	1.9E-04	2.3E-05	8.4E-04	--
Mercury (total)	1.8E-02	9.1E-03	6.9E-03	3.4E-03	5.1E-03	2.5E-03	5.4E-03	2.7E-03	4.8E-03	9.5E-04	1.0E-03	5.2E-04
Nickel	3.4E-05	2.4E-05	4.5E-05	3.2E-05	4.0E-05	2.8E-05	6.2E-05	4.3E-05	1.7E-05	8.5E-06	1.0E-05	7.3E-06
Selenium	7.4E-03	3.7E-03	1.6E-03	8.1E-04	7.6E-04	3.8E-04	5.4E-03	2.7E-03	3.8E-03	2.3E-03	1.2E-04	6.0E-05
Zinc	9.0E-04	--	2.0E-03	--	1.3E-03	--	1.1E-03	--	1.9E-04	9.5E-05	2.8E-04	--
Total PAHs	8.5E-03	8.5E-04	1.2E-02	1.2E-03	1.2E-02	1.2E-03	3.0E-02	3.0E-03	3.1E-04	3.1E-05	2.8E-03	2.8E-04

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-20. Hypothetical hazard quotients for receptors occurring outside the Southwest Marine leasehold calculated using maximum chemical concentrations and species-specific area-use factors

	Brown Pelican		Least Tern		Western Grebe		California Sea Lion	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	7.7E-03	1.8E-03	5.1E-03	1.2E-03	2.3E-03	5.1E-04	5.9E-03	3.1E-03
Tributyltin	2.1E-05	8.5E-06	2.1E-05	8.5E-06	9.5E-06	3.8E-06	1.6E-06	1.1E-06
Metals								
Arsenic	1.4E-04	5.6E-05	2.4E-04	9.7E-05	1.1E-04	4.6E-05	5.6E-03	5.6E-04
Cadmium	4.7E-05	3.5E-06	1.9E-05	1.4E-06	1.0E-05	7.8E-07	1.8E-05	1.8E-06
Chromium	1.8E-03	3.6E-04	2.7E-03	5.5E-04	2.5E-03	4.9E-04	1.2E-04	5.9E-06
Copper	3.0E-04	2.3E-04	3.4E-04	2.5E-04	2.5E-04	1.9E-04	3.1E-04	2.5E-04
Lead	6.6E-04	--	8.4E-04	--	7.6E-04	--	6.1E-05	7.4E-06
Mercury (total)	1.9E-02	9.3E-03	5.6E-03	2.8E-03	3.5E-03	1.7E-03	4.8E-03	9.7E-04
Nickel	1.7E-05	1.2E-05	1.4E-05	9.9E-06	8.9E-06	6.2E-06	8.5E-06	4.2E-06
Selenium	2.5E-03	1.3E-03	1.8E-03	9.0E-04	8.4E-04	4.2E-04	1.3E-03	8.0E-04
Zinc	3.6E-04	--	1.4E-03	--	6.5E-04	--	7.6E-05	3.8E-05
Total PAHs	2.5E-03	2.5E-04	3.5E-03	3.5E-04	2.0E-03	2.0E-04	9.1E-05	9.1E-06

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-21. Hypothetical hazard quotients for receptors foraging exclusively at the original reference area calculated using average chemical concentrations

	Brown Pelican		Least Tern		Western Grebe		Surf Scoter		California Sea Lion		Pacific Green Turtle	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	2.7E-01	6.2E-02	2.9E-01	6.5E-02	1.3E-01	2.9E-02	9.5E-02	2.2E-02	2.1E-01	1.1E-01	2.8E-04	6.5E-05
Tributyltin	4.7E-04	1.9E-04	5.6E-04	2.3E-04	2.5E-04	9.9E-05	1.1E-03	4.5E-04	3.6E-05	2.4E-05	1.8E-06	7.1E-07
Metals												
Arsenic	6.9E-03	2.8E-03	1.4E-02	5.8E-03	6.7E-03	2.7E-03	2.6E-02	1.0E-02	2.8E-01	2.8E-02	4.3E-04	1.7E-04
Cadmium	4.9E-03	3.7E-04	2.6E-03	2.0E-04	1.3E-03	9.9E-05	1.1E-02	7.9E-04	1.9E-03	1.9E-04	9.4E-04	7.0E-05
Chromium	9.6E-02	1.9E-02	5.9E-01	1.2E-01	3.0E-01	6.0E-02	3.4E-01	6.9E-02	6.5E-03	3.1E-04	1.5E-02	2.9E-03
Copper	7.4E-03	5.6E-03	2.2E-02	1.6E-02	1.1E-02	8.5E-03	3.1E-02	2.3E-02	7.5E-03	6.0E-03	2.4E-03	1.8E-03
Lead	1.6E-02	--	3.5E-02	--	2.5E-02	--	5.7E-02	--	1.5E-03	1.9E-04	5.0E-03	--
Mercury (total)	1.1E+00	5.3E-01	2.5E-01	1.3E-01	1.2E-01	6.1E-02	1.4E-01	7.0E-02	2.8E-01	5.5E-02	4.5E-03	2.3E-03
Nickel	1.0E-03	7.2E-04	3.4E-03	2.4E-03	1.6E-03	1.1E-03	2.6E-03	1.8E-03	5.2E-04	2.6E-04	1.3E-04	9.4E-05
Selenium	1.1E-01	5.5E-02	3.1E-01	1.5E-01	1.4E-01	6.9E-02	4.9E-01	2.4E-01	5.8E-02	3.5E-02	5.2E-03	2.6E-03
Zinc	3.3E-02	--	1.1E-01	--	4.9E-02	--	3.2E-02	--	7.0E-03	3.5E-03	4.4E-03	--
Total PAHs	1.8E-01	1.8E-02	2.3E-01	2.3E-02	1.1E-01	1.1E-02	3.1E-01	3.1E-02	6.5E-03	6.5E-04	1.2E-02	1.2E-03

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-22. Hypothetical hazard quotients for receptors foraging exclusively at the final reference pool area calculated using average chemical concentrations

	Brown Pelican		Least Tern		Western Grebe		Surf Scoter		California Sea Lion		Pacific Green Turtle	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	2.7E-01	6.2E-02	2.9E-01	6.5E-02	1.3E-01	2.9E-02	9.5E-02	2.2E-02	2.1E-01	1.1E-01	2.8E-04	6.4E-05
Tributyltin	4.7E-04	1.9E-04	5.6E-04	2.3E-04	2.5E-04	1.0E-04	1.1E-03	4.5E-04	3.6E-05	2.4E-05	1.8E-06	7.2E-07
Metals												
Arsenic	7.0E-03	2.8E-03	1.5E-02	5.8E-03	6.8E-03	2.7E-03	2.6E-02	1.0E-02	2.8E-01	2.8E-02	4.5E-04	1.8E-04
Cadmium	4.9E-03	3.7E-04	2.6E-03	2.0E-04	1.3E-03	9.7E-05	1.1E-02	7.9E-04	1.9E-03	1.9E-04	9.4E-04	7.0E-05
Chromium	1.1E-01	2.2E-02	6.1E-01	1.2E-01	3.3E-01	6.6E-02	3.7E-01	7.4E-02	7.6E-03	3.6E-04	1.7E-02	3.4E-03
Copper	7.7E-03	5.8E-03	2.2E-02	1.7E-02	1.2E-02	8.9E-03	3.1E-02	2.4E-02	7.9E-03	6.3E-03	2.5E-03	1.9E-03
Lead	1.8E-02	--	3.7E-02	--	2.7E-02	--	5.9E-02	--	1.6E-03	2.0E-04	5.2E-03	--
Mercury (total)	1.1E+00	5.3E-01	2.5E-01	1.3E-01	1.2E-01	6.2E-02	1.4E-01	7.2E-02	2.8E-01	5.5E-02	4.7E-03	2.4E-03
Nickel	1.1E-03	7.8E-04	3.5E-03	2.5E-03	1.7E-03	1.2E-03	2.7E-03	1.9E-03	5.6E-04	2.8E-04	1.4E-04	1.0E-04
Selenium	1.1E-01	5.4E-02	3.0E-01	1.5E-01	1.3E-01	6.7E-02	4.8E-01	2.4E-01	5.6E-02	3.4E-02	5.0E-03	2.5E-03
Zinc	3.3E-02	--	1.1E-01	--	4.9E-02	--	3.2E-02	--	7.1E-03	3.5E-03	4.5E-03	--
Total PAHs	1.8E-01	1.8E-02	2.3E-01	2.3E-02	1.1E-01	1.1E-02	3.1E-01	3.1E-02	6.6E-03	6.6E-04	1.3E-02	1.3E-03

Note:
-- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-23. Hypothetical hazard quotients for receptors foraging exclusively inside the NASSCO leasehold calculated using average chemical concentrations

	Brown Pelican		Least Tern		Western Grebe		Surf Scoter		California Sea Lion		Pacific Green Turtle	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	7.3E-01	1.7E-01	4.4E-01	9.9E-02	1.9E-01	4.4E-02	8.2E-02	1.9E-02	5.5E-01	2.9E-01	6.7E-04	1.5E-04
Tributyltin	1.0E-03	4.0E-04	5.6E-04	2.2E-04	2.9E-04	1.2E-04	3.4E-03	1.4E-03	7.8E-05	5.1E-05	7.3E-06	2.9E-06
Metals												
Arsenic	8.5E-03	3.4E-03	1.6E-02	6.4E-03	7.9E-03	3.2E-03	4.1E-02	1.7E-02	3.4E-01	3.4E-02	7.4E-04	3.0E-04
Cadmium	4.7E-03	3.6E-04	2.9E-03	2.2E-04	1.6E-03	1.2E-04	1.3E-02	9.5E-04	1.9E-03	1.9E-04	1.7E-03	1.3E-04
Chromium	1.8E-01	3.5E-02	2.6E-01	5.1E-02	2.4E-01	4.8E-02	5.0E-01	9.9E-02	1.2E-02	5.7E-04	5.3E-02	1.1E-02
Copper	1.5E-02	1.1E-02	2.3E-02	1.7E-02	1.8E-02	1.4E-02	8.8E-02	6.7E-02	1.5E-02	1.2E-02	1.5E-02	1.1E-02
Lead	5.1E-02	--	6.2E-02	--	6.3E-02	--	1.4E-01	--	4.7E-03	5.8E-04	2.1E-02	--
Mercury (total)	1.6E+00	7.9E-01	3.9E-01	1.9E-01	2.1E-01	1.1E-01	2.6E-01	1.3E-01	4.1E-01	8.3E-02	1.9E-02	9.3E-03
Nickel	1.4E-03	9.5E-04	1.4E-03	9.7E-04	9.5E-04	6.6E-04	5.8E-03	4.0E-03	6.8E-04	3.4E-04	2.1E-04	1.5E-04
Selenium	3.5E-01	1.8E-01	1.4E-01	7.2E-02	6.7E-02	3.4E-02	4.4E-01	2.2E-01	1.8E-01	1.1E-01	5.8E-03	2.9E-03
Zinc	3.6E-02	--	1.3E-01	--	6.2E-02	--	4.3E-02	--	7.7E-03	3.9E-03	9.2E-03	--
Total PAHs	2.4E-01	2.4E-02	2.9E-01	2.9E-02	1.7E-01	1.7E-02	7.5E-01	7.5E-02	8.6E-03	8.6E-04	2.6E-02	2.6E-03

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-24. Hypothetical hazard quotients for receptors foraging exclusively outside the NASSCO leasehold calculated using average chemical concentrations

	Brown Pelican		Least Tern		Western Grebe		California Sea Lion	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	3.3E-01	7.5E-02	5.2E-01	1.2E-01	2.3E-01	5.2E-02	2.5E-01	1.3E-01
Tributyltin	1.9E-03	7.6E-04	7.4E-04	3.0E-04	3.5E-04	1.4E-04	1.5E-04	9.6E-05
Metals								
Arsenic	1.1E-02	4.5E-03	1.8E-02	7.2E-03	8.5E-03	3.4E-03	4.6E-01	4.6E-02
Cadmium	4.1E-03	3.1E-04	1.8E-03	1.3E-04	9.9E-04	7.4E-05	1.6E-03	1.6E-04
Chromium	3.3E-01	6.5E-02	2.0E-01	4.0E-02	1.7E-01	3.4E-02	2.2E-02	1.1E-03
Copper	1.2E-02	8.9E-03	2.0E-02	1.5E-02	1.3E-02	9.6E-03	1.2E-02	9.6E-03
Lead	4.1E-02	--	4.6E-02	--	4.4E-02	--	3.8E-03	4.6E-04
Mercury (total)	1.4E+00	7.2E-01	3.8E-01	1.9E-01	1.9E-01	9.7E-02	3.8E-01	7.5E-02
Nickel	1.4E-03	9.8E-04	1.3E-03	8.8E-04	7.9E-04	5.5E-04	7.1E-04	3.5E-04
Selenium	3.2E-01	1.6E-01	1.7E-01	8.5E-02	7.9E-02	3.9E-02	1.6E-01	1.0E-01
Zinc	3.9E-02	--	1.6E-01	--	7.3E-02	--	8.2E-03	4.1E-03
Total PAHs	2.0E-01	2.0E-02	2.9E-01	2.9E-02	1.5E-01	1.5E-02	7.2E-03	7.2E-04

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-25. Hypothetical hazard quotients for receptors foraging exclusively inside the Southwest Marine leasehold calculated using average chemical concentrations

	Brown Pelican		Least Tern		Western Grebe		Surf Scoter		California Sea Lion		Pacific Green Turtle	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	7.8E-01	1.8E-01	6.6E-01	1.5E-01	3.0E-01	6.8E-02	1.2E-01	2.8E-02	5.9E-01	3.1E-01	1.9E-03	4.3E-04
Tributyltin	1.6E-03	6.5E-04	1.3E-03	5.2E-04	6.9E-04	2.8E-04	4.3E-03	1.7E-03	1.2E-04	8.2E-05	2.4E-05	9.6E-06
Metals												
Arsenic	1.0E-02	4.1E-03	2.1E-02	8.6E-03	1.1E-02	4.2E-03	4.5E-02	1.8E-02	4.1E-01	4.1E-02	1.0E-03	4.2E-04
Cadmium	6.3E-03	4.7E-04	2.7E-03	2.0E-04	1.8E-03	1.4E-04	1.5E-02	1.1E-03	2.5E-03	2.5E-04	1.8E-03	1.3E-04
Chromium	2.7E-01	5.5E-02	2.6E-01	5.2E-02	2.4E-01	4.8E-02	3.8E-01	7.5E-02	1.9E-02	8.9E-04	8.4E-02	1.7E-02
Copper	2.8E-02	2.2E-02	4.5E-02	3.4E-02	3.3E-02	2.5E-02	7.7E-02	5.8E-02	2.9E-02	2.3E-02	1.6E-02	1.2E-02
Lead	6.8E-02	--	1.1E-01	--	9.7E-02	--	1.4E-01	--	6.3E-03	7.7E-04	2.7E-02	--
Mercury (total)	1.3E+00	6.7E-01	4.1E-01	2.1E-01	2.4E-01	1.2E-01	2.7E-01	1.4E-01	3.5E-01	7.0E-02	3.4E-02	1.7E-02
Nickel	1.5E-03	1.1E-03	1.9E-03	1.3E-03	1.2E-03	8.6E-04	3.3E-03	2.3E-03	7.8E-04	3.9E-04	3.2E-04	2.2E-04
Selenium	5.0E-01	2.5E-01	1.6E-01	7.8E-02	7.3E-02	3.7E-02	5.2E-01	2.6E-01	2.6E-01	1.6E-01	5.9E-03	2.9E-03
Zinc	4.2E-02	--	1.4E-01	--	6.6E-02	--	5.2E-02	--	9.0E-03	4.5E-03	9.6E-03	--
Total PAHs	3.5E-01	3.5E-02	5.1E-01	5.1E-02	3.8E-01	3.8E-02	2.1E+00	2.1E-01	1.3E-02	1.3E-03	8.2E-02	8.2E-03

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 10-26. Hypothetical hazard quotients for receptors foraging exclusively outside the Southwest Marine leasehold calculated using average chemical concentrations

	Brown Pelican		Least Tern		Western Grebe		California Sea Lion	
Chemical	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Total PCBs	4.7E-01	1.1E-01	5.1E-01	1.2E-01	2.2E-01	5.1E-02	3.6E-01	1.8E-01
Tributyltin	1.5E-03	5.9E-04	2.1E-03	8.5E-04	9.4E-04	3.8E-04	1.1E-04	7.4E-05
Metals								
Arsenic	1.1E-02	4.2E-03	2.4E-02	9.7E-03	1.1E-02	4.5E-03	4.2E-01	4.2E-02
Cadmium	3.6E-03	2.7E-04	1.7E-03	1.3E-04	9.0E-04	6.8E-05	1.4E-03	1.4E-04
Chromium	1.3E-01	2.5E-02	2.0E-01	4.0E-02	1.7E-01	3.4E-02	8.5E-03	4.1E-04
Copper	1.5E-02	1.2E-02	2.5E-02	1.9E-02	1.5E-02	1.2E-02	1.6E-02	1.2E-02
Lead	3.8E-02	--	6.0E-02	--	5.0E-02	--	3.5E-03	4.3E-04
Mercury (total)	1.4E+00	6.9E-01	4.6E-01	2.3E-01	2.3E-01	1.2E-01	3.6E-01	7.2E-02
Nickel	1.2E-03	8.5E-04	1.4E-03	9.5E-04	8.2E-04	5.7E-04	6.1E-04	3.0E-04
Selenium	2.3E-01	1.2E-01	1.8E-01	9.0E-02	8.4E-02	4.2E-02	1.2E-01	7.3E-02
Zinc	3.2E-02	--	1.4E-01	--	6.3E-02	--	6.7E-03	3.3E-03
Total PAHs	2.0E-01	2.0E-02	3.1E-01	3.1E-02	1.6E-01	1.6E-02	7.4E-03	7.4E-04

Note: -- - not calculated (no toxicity reference value available)
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl

Table 11-1. Human health tissue residue guidelines

Chemical	Human Health Tissue Residue Guidelines (μg/kg)	Basis
Metals		
Arsenic, inorganic	1,000	OEHHA (1999)
Cadmium	3,000	OEHHA (1999)
Chromium	120,000	Calculated based on OEHHA (1999)
Copper	120,000	Calculated based on OEHHA (1999)
Mercury, total	300	OEHHA (1999)
Nickel	67,000	Calculated based on OEHHA (1999)
Selenium	2,000	OEHHA (2001)
Silver	17,000	Calculated based on OEHHA (1999)
Zinc	1,000,000	Calculated based on OEHHA (1999)
Organometallic Compounds		
Tributyltin	1,000	Calculated based on OEHHA (1999)
Polycyclic Aromatic Hydrocarbons		
Naphthalene	67,000	Calculated based on OEHHA (1999)
Acenaphthene	200,000	Calculated based on OEHHA (1999)
Fluorene	130,000	Calculated based on OEHHA (1999)
Anthracene	1,000,000	Calculated based on OEHHA (1999)
Fluoranthene	130,000	Calculated based on OEHHA (1999)
Pyrene	67,000	Calculated based on OEHHA (1999)
Benz[a]anthracene	28	Calculated based on OEHHA (1999)
Chrysene	280	Calculated based on OEHHA (1999)
Benzo[b]fluoranthene	28	Calculated based on OEHHA (1999)
Benzo[k]fluoranthene	28	Calculated based on OEHHA (1999)
Benzo[a]pyene	2.8	Calculated based on OEHHA (1999)
Indeno[1,2,3-cd]pyrene	28	Calculated based on OEHHA (1999)
Dibenz[a,h]anthracene	8.1	Calculated based on OEHHA (1999)
Polychlorinated Biphenyls		
Total polychlorinated biphenyls[a]	20	OEHHA (1999)

[a] Expressed as the sum of Aroclors® 1248, 1254, and 1260, as in OEHHA (1999).

Table 11-2. Toxicity criteria used to develop human health tissue residue guidelines

Chemical	CSF (mg/kg-day)$^{-1}$	RfD (mg/kg-day)	Source
Metals			
Arsenic, inorganic	1.5	0.0003	U.S. EPA (2003)
Cadmium	NA	0.0005	U.S. EPA (2003)
Chromium	NA	0.003	U.S. EPA (2003)
Copper	NA	0.037	U.S. EPA (2003)
Mercury, total	NA	0.0001	U.S. EPA (2003)
Nickel	NA	0.02	U.S. EPA (2003)
Selenium	NA	0.005	U.S. EPA (2003)
Silver	NA	0.005	U.S. EPA (2003)
Zinc	NA	0.3	U.S. EPA (2003)
Organometallic Compounds			
Tributyltin	NA	0.0003	U.S. EPA (2003)
Polycyclic Aromatic Hydrocarbons			
Naphthalene	NA	0.02	U.S. EPA (2003)
Acenaphthene	NA	0.06	U.S. EPA (2003)
Fluorene	NA	0.04	U.S. EPA (2003)
Anthracene	NA	0.3	U.S. EPA (2003)
Fluoranthene	NA	0.04	U.S. EPA (2003)
Pyrene	NA	0.02	U.S. EPA (2003)
Benz[a]anthracene	1.2	NA	OEHHA (2001)
Chrysene	0.12	NA	OEHHA (2001)
Benzo[b]fluoranthene	1.2	NA	OEHHA (2001)
Benzo[k]fluoranthene	1.2	NA	OEHHA (2001)
Benzo[a]pyene	12	NA	OEHHA (2001)
Indeno[1,2,3-cd]pyrene	1.2	NA	OEHHA (2001)
Dibenz[a,h]anthracene	4.1	NA	OEHHA (2001)
Polychlorinated Biphenyls			
Total PCBs[a]	2		U.S. EPA (2003)
Total PCBs (as Aroclor® 1254)[b]		0.00002	U.S. EPA (2003)

Note: CSF - cancer slope factor
NA - not available
PCB - polychlorinated biphenyl
RfD - reference dose

[a] To be applied to the sum of Aroclors® 1248, 1254, and 1260, as in OEHHA (1999). Aroclors® 1248 and 1254 were not detected in any sample, so the concentration of total PCBs reflects only Aroclor 1260® in this assessment.

[b] RfDs are available only for Aroclors® 1254 and 1016, neither of which were detected in any sample. The RfD for Aroclor® 1254 was used as a surrogate.

Table 11-3. Screening of shipyard-related chemicals in fish and lobster tissue

Chemical	Maximum Spotted Sand Bass Concentration (µg/kg)	Maximum Lobster Tissue Concentration (µg/kg)	Human Health Tissue Residue Guideline (µg/kg)
Metals			
Arsenic, inorganic[a]	28	532	1,000
Cadmium	2.5 *U*	50	3,000
Chromium	50 *U*	50 *U*	10,000
Copper	460	17,900	120,000
Mercury, total	224	$\boxed{521}$	300
Nickel	20 *U*	50 *U*	67,000
Selenium	500	300	2,000
Silver	2 *U*	21	17,000
Zinc	4,900	32,400	1,000,000
Organometallic Compounds			
Tributyltin	23	9.6	1,000
Polycyclic Aromatic Hydrocarbons			
Naphthalene	5 *U*	5 *U*	67,000
Acenaphthene	5 *U*	5 *U*	200,000
Fluorene	5 *U*	5 *U*	130,000
Anthracene	5 *U*	5 *U*	1,000,000
Fluoranthene	5 *U*	5 *U*	130,000
Pyrene	5 *U*	5 *U*	67,000
Benz[a]anthracene	5 *U*	5 *U*	28
Chrysene	5 *U*	5 *U*	280
Benzo[b]fluoranthene	5 *U*	5 *U*	28
Benzo[k]fluoranthene	5 *U*	5 *U*	28
Benzo[a]pyrene	5 *U*	5 *U*	2.8
Indeno[1,2,3-cd]pyrene	5 *U*	5 *U*	28
Dibenz[a,h]anthracene	5 *U*	5 *U*	8.1
Polychlorinated Biphenyls			
Total PCBs[b]	$\boxed{400}$	$\boxed{21}$	20

Note: Chemicals not detected in any sample from a station are qualified with a "*U*" and one-half the quantitation limit is listed.

Chemical concentrations exceeding a tissue residue guideline are enclosed in a box.

PCB - polychlorinated biphenyl

[a] Inorganic arsenic concentration was estimated assuming that 4 percent of total arsenic was inorganic, as described in Section 11.

[b] Expressed as the sum of Aroclors® 1248, 1254, and 1260, as in OEHHA (1999). Aroclors 1248® and 1254 were not detected in any sample and thus were not included in the total PCB concentration.

Table 11-4. Screening of total PCBs and mercury in fish and lobster against human health reference concentrations

Location	Spotted Sand Bass Fillets	Edible Lobster Tissue		Whole Body Lobster
	Total PCBs[a] (μg/kg wet)	Total PCBs[a] (μg/kg wet)	Total Mercury (μg/kg wet)	Total PCBs[a] (μg/kg wet)
Reference	40	12	73	31
	10 *U*	10 *U*	100	26 *J*
	31	10 *U*	71	25 *J*
	19	15	73	41
	55	10 *U*	110	22 *J*
Minimum	5 [b]	5 [b]	71	22
Maximum	55	15	110	41
NASSCO Inside	27	11	107	47
	34	10 *U*	521	25
	38	10 *U*	55	45
	46	11	316	76
	18	10 *U*	68	49
Maximum	46	11	521	76
Arithmetic Mean	33	7	213	48
95%UCL	53	NA	2,507	83
Exposure Concentration	46	11	$\boxed{521}$	$\boxed{76}$
NASSCO Outside	57			
	40			
	35			
	27			
	32			
Maximum	57			
Arithmetic Mean	38			
95%UCL	54			
Exposure Concentration	54			
Southwest Marine Inside	27	10 *U*	68	38
	190	12	44	40
	69	10 *U*	65	64
	400	21	109	30
	140	10	84	41
Maximum	400	21	109	64
Arithmetic Mean	165	11	74	43
95%UCL	2,566	30	114	59
Exposure Concentration	$\boxed{400}$	21	109	$\boxed{59}$
Southwest Marine Outside	110			
	69			
	41			
	41			
	39			
Maximum	110			
Arithmetic Mean	60			
95%UCL	115			
Exposure Concentration	$\boxed{110}$			

Note: The exposure concentration is the lesser of the maximum detected concentration and the 95%UCL. Exposure concentrations exceeding the range of reference concentrations are enclosed in a box.

95%UCL - 95 percent upper confidence limit
PCB - polychlorinated biphenyl
U - undetected in sample at the quantitation limit listed

[a] Expressed as the sum of Aroclor® 1248, 1254, and 1260, as in OEHHA (1999). Aroclors® 1248 and 1254 were not detected in any sample and thus were not included in the total PCB concentration.

[b] Minimum concentration listed is one-half the quantitation limit for the non-detected samples.

Table 11-5. Exposure assumptions used in the human health risk assessment

Parameter		Units	Value
Target Risk	TR	$(mg/kg\text{-}day)^{-1}$	0.00001
Target Hazard Quotient	THQ	unitless	1
Fish or Shellfish Consumption Rate	CR	kg/day	0.021
Body Weight	BW	kg	70
Exposure Duration	ED	years	30
Exposure Frequency	EF	days/year	365
Fraction Ingested from Site[a]	FI	unitless	0.034, 0.005, 0.023, 0.002
Averaging Time for Carcinogens	ATc	days	25,550
Averaging Time for Noncarcinogens	ATn	days	10,950
Conversion Factor	CF	μg/mg	1,000

[a] The four values given are for NASSCO inside the leasehold, NASSCO outsite the leasehold, Southwest Marine inside the leasehold, and Southwest Marine outside the leasehold, respectively. The calculated values are based on shoreline/pier length for inside the leaseholds and based on area for outside the leaseholds, as shown in Table 11-6.

Table 11-6. Calculation of fractional intake for fish and shellfish consumption

Location	Area (m^2)	Fractional Intake by Area	Length (km)	Fractional Intake by Length[a]
NASSCO Inside	163,091	0.004	3.2	0.034
NASSCO Outside	238,147	0.005	--	
Southwest Marine Inside	85,143	0.002	2.1	0.023
Southwest Marine Outside	92,080	0.002	--	
Total	578,460	0.013	5.3	0.057
San Diego Bay	44,722,748		93.2	

Note: -- - not applicable

[a] Measurements for shipyards include all shoreline and pier lengths. Measurements for San Diego Bay include only shoreline.

Table 11-7. Chemical intake and estimated human health risks

	Spotted Sand Bass Fillets	Edible Lobster Tissue	
	Total PCBs (μg/kg wet)	Total PCBs (μg/kg wet)	Total Mercury (μg/kg wet)
NASSCO Inside			
Exposure Concentration (μg/kg wet)	--	--	521
Estimated Intakes (mg/kg-day)			
Cancer	--	--	--
Noncancer	--	--	5×10^{-6}
Risk Estimates			
Cancer Risk	--	--	
Noncancer Hazard Index	--	--	0.05
Southwest Marine Inside			
Exposure Concentration (μg/kg wet)	400	21	--
Estimated Intakes (mg/kg-day)			
Cancer	1×10^{-6}	6×10^{-8}	--
Noncancer	--	--	--
Risk Estimates			
Cancer Risk	2×10^{-6}	1×10^{-7}	--
Noncancer Hazard Index	--	--	--
Southwest MarineOutside			
Exposure Concentration (μg/kg wet)	110	--	--
Estimated Intakes (mg/kg-day)			
Cancer	3×10^{-8}	--	--
Noncancer	--	--	--
Risk Estimates			
Cancer Risk	6×10^{-8}	--	--
Noncancer Hazard Index	--	--	--

Note: -- - risk was not calculated because the chemical was screened out in this area
PCB - polychlorinated biphenyl

[a] Values are reported only for the area inside the NASSCO leasehold because all chemicals were screened out in the area outside the leasehold.

Table 12-1. Sediment cleanup levels and other benchmarks

Chemical[a]	Units	LAET	Final Reference Pool Sediment 95%UPL	Effects Range-Low[b]	Effects Range-Median[b]
Arsenic	mg/kg	27	9	8.2	70
Copper	mg/kg	1,000	120	34	270
Lead	mg/kg	250	48	47	218
Mercury	mg/kg	3.2	0.56	0.15	0.71
Zinc	mg/kg	1,200	210	150	410
Tributyltin	μg/kg	1,900	5.1		
HPAH	μg/kg	26,000 *G*	340	1,700	9,600
PCBs	μg/kg			23	180
Total PCB congeners	μg/kg		36		
Total PCB homologs	μg/kg	3,000			
Diesel-range organics	mg/kg	490			
Residual-range organics	mg/kg	1,300			

Note: 95%UPL - 95 percent upper prediction limit
G - the true no-effect level may be greater than the value shown
HPAH - high-molecular-weight polycyclic aromatic hydrocarbon
LAET - lowest apparent effects threshold
PCB - polychlorinated biphenyl

[a] All values are reported on a dry weight basis.

[b] Long et al. (1995).

Table 12-2. LAET exceedances

Station	Arsenic	Copper	Lead	Mercury	Zinc	TBT	Total PCB Homologs	HPAH	DRO	RRO
NA01									7.1	3.7
NA04				1.3					11.0	5.1
NA09				1.9			2.4	1.4	16.7	6.9
NA16				1.2					16.3	6.2
NA20									1.4	
NA21									6.1	3.7
NA27										1.2
NA28									1.0	1.5
SW02				1.7			2.9		2.4	1.4
SW04	4.1	2.2	1.9	2.3	3.8	2.6	9.0	2.2	4.3	2.4
SW08		1.5	1.4	1.9	1.1	3.7	4.3		1.9	1.2
SW17									1.2	
SW20							2.2		1.2	
SW21							1.2		1.1	
SW24							1.7	2.2	10.2	4.0
SW25									1.0	
SW27									1.2	
SW28							1.1		5.1	2.7
SW30									1.2	
SW36									1.3	

Note: Values are the maximum exceedance factor at any depth: the ratio of the chemical concentration to the LAET.

DRO - diesel-range organics
HPAH - high-molecular-weight polycyclic aromatic hydrocarbon
LAET - lowest apparent effects threshold
PCB - polychlorinated biphenyl
RRO - residual-range organics
TBT - tributyltin

Table 12-3. Summary of LAET exceedances

Station	Maximum Sample Depth	Maximum Depth of LAET Exceedance	Description of Exceedances
NA01	5.5 ft	5.5 ft	DRO, RRO
NA02	3.7 ft	none	
NA03	2 cm	none	
NA04	8.3 ft	8.3 ft	Cadmium, chromium, mercury, selenium, silver, DRO, RRO
NA05	2 cm	none	
NA06	3.9 ft	none	
NA07	2 cm	none	
NA08	2 cm	none	
NA09	8.8 ft	6 ft	Mercury, DRO, RRO, PCBs, HPAH
NA10	2 cm	none	
NA11	2 cm	none	
NA12	2 cm	none	
NA13	3.2 ft	none	
NA14	2 cm	none	
NA15	2 cm	none	
NA16	6.1 ft	4 ft	Mercury, DRO, RRO
NA17	5.1 ft	none	
NA18	2 cm	none	
NA19	5.8 ft	none	
NA20	8.1 ft	8.1 ft	DRO only at 6–8 ft
NA21	7.6 ft	4 ft	DRO, RRO
NA22	2 cm	none	
NA23	4.7 ft	none	
NA24	4 ft	none	
NA25	5.2 ft	none	
NA26	7.5 ft	none	
NA27	2 cm	2 cm	RRO
NA28	2 cm	2 cm	DRO, RRO
NA29	4.4 ft	none	
NA30	3.4 ft	none	
NA31	3 ft	none	

Table 12-3. (cont.)

Station	Maximum Sample Depth	Maximum Depth of LAET Exceedance	Description of Exceedances
SW01	5.4 ft	none	
SW02	4.9 ft	2 ft	Mercury, PCB, DRO, RRO
SW03	2 cm	none	
SW04	4.1 ft	4.1 ft	Arsenic, copper, lead, mercury, zinc, TBT, HPAH, PCB, DRO, RRO
SW05	2 cm	none	
SW06	2 cm	none	
SW07	2 cm	none	
SW08	6.5 ft	4 ft	Copper, lead, mercury, zinc, TBT, PCB, DRO, RRO
SW09	2 cm	none	
SW10	2.9 ft	none	
SW11	2 cm	none	
SW12	3.7 ft	none	
SW13	2 cm	none	
SW14	2 cm	none	
SW15	2 cm	none	
SW16	2 cm	none	
SW17	6.2 ft	4 ft	DRO
SW18	2 cm	none	
SW19	5.4 ft	none	
SW20	2.4 ft	1.5 ft	PCB, DRO
SW21	2 cm	2 cm	PCB, DRO
SW22	2 cm	none	
SW23	2 cm	none	
SW24	3 ft	3 ft	PCB, HPAH, DRO, RRO; only HPAH at surface
SW25	4.2 ft	4.2 ft	Only DRO at 2–4.2 ft: 500 vs. criterion of 490
SW26	2 cm	none	
SW27	5.6 ft	2 ft	DRO
SW28	5.3 ft	4 ft	PCB, DRO, RRO
SW29	7 ft	none	
SW30	8.7 ft	4 ft	DRO at 2–4 ft

Table 12-3. (cont.)

Station	Maximum Sample Depth	Maximum Depth of LAET Exceedance	Description of Exceedances
SW31	2.9 ft	none	
SW32	2.8 ft	none	
SW33	2.5 ft	none	
SW34	2.8 ft	none	
SW36	4.25 ft	4.25 ft	DRO at 2–4.25 ft

Note: DRO - diesel-range organics
HPAH - high-molecular-weight polycyclic aromatic hydrocarbon
LAET - lowest apparent effects threshold
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl
PCT - polychlorinated terphenyl
RRO - residual-range organics

Table 12-4. Truth table for LAET exceedance as a cause of aquatic life effects

	LAET Exceedance (E)	Likely Beneficial Use Impairment (I)	Exclusive Causation (E ○→ I)	Non-exclusive Causation (E → I)	Stations
1	T	T	T	T	SW21
2	T	F	F	F	SW02, SW04
3	F	T	F	T	NA04, NA15, NA20, NA22, SW03, SW17, SW22, SW23
4	F	F	T	T	NA01, NA03, NA05, NA06, NA07, NA09, NA11, NA12, NA16, NA17, NA19, SW08, SW09, SW11, SW13, SW15, SW18, SW25,

Note: LAET - lowest apparent effects threshold

Table 12-5. Truth table for LAET exceedance and high percent fines as a cause of aquatic life effects

	LAET Exceedance and High Fines (E)	Likely Beneficial Use Impairment (I)	Exclusive Causation (E ○→ I)	Non-exclusive Causation (E → I)	Stations
1	T	T	T	T	SW21
2	T	F	F	F	
3	F	T	F	T	NA04, NA15, NA20, NA22, SW03, SW17, SW22, SW23
4	F	F	T	T	NA01, NA03, NA05, NA06, NA07, NA09, NA11, NA12, NA16, NA17, NA19, SW02, SW04, SW08, SW09, SW11, SW13, SW15, SW18, SW25, SW27

Note: LAET - lowest apparent effects threshold

Table 15-1. Preliminary technology screening matrix

General Response Action	Remedial Technology	Process Options	Effectiveness	Implementability	Cost	Retained (Yes/No)
I. No Action	NA	NA	Potentially effective.	Implementable.	Low	Yes
II. Natural Recovery	NA	NA	Potentially effective.	Implementable.	Low	Yes
III. Containment	Subaqueous Capping	1. Thick Sand/Clay/Gravel Cap	Containment of sediments may not be reliable due to prop wash.	Capping in shallow areas restricts future navigation.	Moderate	No
		2. Thin Sand/Sediment Cap	Potentially effective where chemical concentrations are low. Low water quality impacts.	May conflict with future navigation, construction, or dredging.	Low	No
IV. Removal	Dredging	1. Mechanical	Effective. Use of enclosed bucket would decrease unwanted sediment resuspension associated with this technology.	Relatively straightforward with conventional technologies. Not feasible near operational features.	Moderate	Yes
		2. Hydraulic	Effective, but adds substantial water to dredged material.	Relatively straightforward with conventional technologies. Can be implemented beneath overwater structures.	Moderate	Yes
V. *In Situ* Treatment	A. Immobilization (Solidification/Stabilization)		Potentially effective for inorganic contaminants (i.e., metals). High moisture and salinity may impair stabilization.	May conflict with future navigation, construction, or dredging.	Moderate	No
	B. Biological Treatment		Not effective for removing metals.	Difficult to implement in harbor. May adversely impact water quality.	Moderate	No
	C. Chemical Treatment		Not effective for removing metals.	Difficult to implement in harbor. May adversely impact water quality.	High	No
VI. *Ex Situ* Treatment	A. Dewatering		Effective.	Implementable. Probably necessary for upland disposal.	Moderate	Yes
	B. Physical Separation (Sediment Washing)		Inefficient for fine-grained material.	Based on available data, percentage of fine-grained materials is too high to implement effectively.	High	No
	C. Thermal Desorption		Not effective for removing metals.	Fine-grained material can pass through system. Would probably require prescreening and dewatering.	High	No
	D. Thermal Destruction		Effective for organics. Not effective for removing nonvolatile metals. Volatile metals can vaporize and are difficult to remove from emissions. Nonvolatile metals remain in residual ash and noncombustible material.	Fine-grained material can pass through system. Would probably require prescreening and dewatering.	High	No
	E. Immobilization (Solidification/Stabilization)		Generally effective for containment of metals. May be less effective with organic contaminants.	Would probably require prescreening and dewatering. Can increase waste volume by more than 20 percent.	Moderate	Yes[a]
	F. Biological Treatment		Not effective for removing metals.	Potentially implementable.	Moderate	No
	G. Chemical Treatment		Not effective for removing metals.	Potentially implementable.	High	No
VII. Disposal	A. Reuse (Beach Replenishment/ Habitat Restoration or Enhancement)		Effective for clean or treated sediment.	Based on available data, percentage of fine-grained materials is too high to meet reuse criteria. Physical separation of coarse-grained materials for reuse could be very expensive.	Moderate	No
	B. Ocean Disposal		Effective.	Sediment must comply with EPA's ocean dumping regulations and the Corps' permitting regulations.	Moderate	Yes
	C. Nearshore Confined Disposal		Potentially effective. May not be sufficient volume to accommodate all material.	Potentially implementable.	Moderate	Yes
	D. Confined Aquatic Disposal		Potentially effective.	Potentially implementable, but no known sites are available.	Moderate	No
	E. Geotextile Bag Containment		High percentage of fine-grained materials may prevent bags from dewatering.	Can slow production considerably.	High	No
	F. Uplands Disposal		Effective.	Sediment may first require treatment.	Moderate to High	Yes

Note: NA - not applicable

[a] Solidification/stabilization is retained for possible limited use in a nearshore confined disposal facility.

Table 18-1. Preliminary cost estimate—Alternative A: Monitored natural recovery

Item	Unit	Unit Cost	Total Quantity	Total NPV Cost	Assumptions	Direct Cost Quote References[a]	
Monitoring Costs							
Natural Recovery Monitoring - Year 1	10 samples	$75,000	3.4	$255,000	Southwest Marine: 13 acres leasehold; 8 non-disturbed (= 4 stations). 29 acres outside (=6 stations). NASSCO: 39 acres leasehold; 24 non-disturbed (=12 stations). 61 acres outside (=12 stations).	Middle Waterway, 2003 (Manson Construction/ MWAC)	Hylebos Waterway, 2003 (Miller Contracting/Port of Tacoma)
Natural Recovery Monitoring - Year 2	10 samples	$75,000	3.4	$240,362		As above	As above
Natural Recovery Monitoring - Year 5	10 samples	$75,000	3.4	$219,965		As above	As above
Natural Recovery Monitoring - Year 10	10 samples	$75,000	3.4	$189,744		As above	As above
Grand Total				**$900,000**		As above	As above

Note: NPV - net present value

[a] The estimated costs associated with this alternative reflect both direct cost quotes from contractors and Anchor Environmental LLC's sediment remedial design and construction experience. These costs address all aspects of construction of each alternative and provide specifics on which contractor's quotes were used, in conjunction with engineering assumptions, for various construction elements to arrive at a reliable estimate of costs. Direct quotes from contractors were applied to specific application at the shipyards. Actual costs will only be possible after a complete design package is released under a competitive bidding process with qualified contractors.

Table 18-2. Preliminary cost estimate—Alternative B1: Cleanup to LAET criteria with offsite disposal

Item	Unit	Unit Cost	Total Quantity	Total Cost	Assumptions	Direct Cost Quote References[a]	
Direct Construction Costs							
Pre-Construction							
Mobilization/demobilization	LS	$150,000	1	$150,000		2003 Cost Estimate for Campbell Shipyard (in conjunction with Ninyo & Moore and R.E. Staite)	2002 Cost Estimates for Middle Waterway and Lockheed Shipyards
Site preparation	LS	as shown		$50,000		As above	As above
Demolition	LS	as shown		$125,000	No specific areas for demolition identified; nominal cost is included for miscellaneous demolition activites.	2003 Demolition costs for Campbell Shipyard (in conjunction with Blaylock Engineering Group)	2001 Cost Estimate for Lockheed Shipyards (demolition input from C.L. Cristich Co.)
Dredging					Includes 1 ft allowable overdredge plus additional contingency foot.		
Unconstrained open-water dredging (outside of leasehold area)	yd^3	$6	0	$0	$5 to $6 per cubic yard is typical for a large volume of unconstrained dredging outside of shipyard area.	R. Carpenter (R.E. Staite) and B. Lofgren (former Manson Const. Co.), 2002 personal communications	2003 Middle Waterway construction bids (Miller Contracting, Manson Construction)
Constrained dredging from inner shipyard (within leasehold area)	yd^3	$12	74,850	$898,200	Higher cost for dredging within leasehold line, near piers, in areas of ship traffic, etc.	As above	As above
Dredging surface/subsurface debris	yd^3	$80	800	$64,000	Volume based on 1percent of total dredge volume. Includes landfill disposal.	Southwest Marine 2002 communication with Sonas Soil Resource Recovery and La Paz County Landfill, Arizona	Duwamish/Diagonal 2003 bids (Miller Contracting, Manson Construction)
Engineering controls (silt curtain, oil boom)	LS	$70,000	1	$70,000	Silt curtain ($50k) and about 1,500 LF of oil boom at $12/LF.	Hylebos Waterway, 2003 (Miller Contracting/Port of Tacoma)	2002 Cost Estimates for Middle Waterway and Lockheed Shipyards
Protection of Marine Structures and Revetments					Costs assume setback of dredging from marine structures and revetments, and placement of riprap berm to reinstate lateral resistance (as per Anchor 2002).	Aggregate purchase costs obtained 2002 for Campbell Shipyards (in conjunction with URS and Ninyo & Moore)	Techniques documented in Moffat & Nichols (2002) may allow additional sediment removal at significantly increased cost
Stone retaining structures along revetments	Ton	$30	2,500	$75,000		As above	As above
Stone revetment berms along pier faces and dolphins	Ton	$30	9,000	$270,000		As above	As above
Stone revetments along bulkheads	Ton	$30	2,500	$75,000		As above	As above
Upland Disposal—Regional Landfill							
Upland staging, offload, dewatering area	LS	$100,000	1	$100,000	Accounts for construction of dewatering area on shipyard property; does not include possible leasing price from use of other properties ($2,000–$5,000 per acre per month?).	Cost Estimate for Campbell Shipyard (in conjunction with Ninyo & Moore and R.E. Staite)	America's Cup Harbor, San Diego
Rehandling and dewatering	yd^3	$14	67,350	$942,900	Assume stockpiling for dewatering, with some potential addition of lime or cement admixture to assist.	Port of San Diego, TAMT and Campbell Shipyards (R.E. Staite)	East Waterway and Terminal 25, 2003 (Manson Construction and Hurlen)
Disposal (including transport and tipping fee)	Ton	$50	101,025	$5,051,250	Regional hazardous waste landfill outside of San Diego County (as opposed to $31/ton for Otay or Sycamore landfills). Updated 8/03 from $45/ton (previous cost assumptions) per experience with PCB disposal (Campbell shipyard).	Southwest Marine 2002 communication with Sonas Soil Resource Recovery and La Paz County Landfill, Arizona	2003 Cost Estimate for Campbell Shipyard (in conjunction with Ninyo & Moore and R.E. Staite)
Onsite Disposal of Sediments							
Mechanical placement without treatment (clamshell)	yd^3	$6	0	$0		B. Lofgren (former Manson Const. Co.), 2002 personal communication	Hylebos Waterway, 2003 (Miller Contracting/Port of Tacoma)

Table 18-2. (cont.)

Item	Unit	Unit Cost	Total Quantity	Total Cost	Assumptions	Direct Cost Quote References[a]	
Volume of sediment undergoing stabilization	yd^3	--	7,500	--		--	--
Purchase cement admixture (8 percent)	Ton	$75	900	$67,500		LA Corps of Engineers Pilot Study (Anchor, 2001)	East Waterway, 2001 (Manson Construction)
Mix cement, place stabilized sediment in lifts, compact	Ton	$25	11,250	$281,250		Tenth Avenue Marine Terminal, 2001 (R.E. Staite)	East Waterway, 2001 (Manson Construction)
Total Direct Construction Costs				**$8,200,000**			
Construction Management	Percent	8%	2	$656,000		Typical value for cost estimating	--
Design	Percent	15%	2	$1,230,000		Typical value for cost estimating	--
Contingency	Percent	30%	2	$3,025,800		Typical for pre-design cost estimating	--
Monitoring Costs							
Water quality monitoring during construction	Week	$12,000	10	$120,000		Middle Waterway, 2003 (Manson Construction/MWAC)	Hylebos Waterway, 2003 (Miller Contracting/Port of Tacoma)
Post-dredging confirmational sampling	Sample	$10,000	15	$150,000	2 confirmational samples per acre	As above	As above
Natural recovery monitoring						As above	As above
Year 1	10 samples	$75,000	0.6	$45,000	5 acres at each shipyard = 3 samples each.	As above	As above
Year 2	10 samples	$75,000	0.6	$42,417	As above	As above	As above
Year 5	10 samples	$75,000	0.6	$38,817	As above	As above	As above
Year 10	10 samples	$75,000	0.6	$33,484	As above	As above	As above
Long-term groundwater and CDF monitoring	Event	$40,000	4	$141,972	4 events per major CDF		
Other (non-construction) Costs							
Permitting/EIR	LS	$400,000	1	$400,000			
Eel grass habitat mitigation	Acre	$100,000	0	$40,000			
CDF 404 compensatory mitigation		TBD					
Habitat evaluation (EFH) costs	LS	as shown		$50,000		Quotes to Southwest Marine, 2003	
Supplemental design sampling	LS	as shown		$100,000			
Construction bid support	LS	$17,500	2	$35,000			
Internal shipyard costs		TBD					
RWQCB past costs	Year	$50,000	4	$200,000			
RWQCB oversight costs	Year	$50,000	6	$300,000			
Grand Total				**$14,800,000**			

Note: CDF - nearshore confined disposal facility
LAET - lowest apparent effects threshold
LS - lump sum
RWQCB - California Regional Water Quality Control Board, San Diego Region

[a] The estimated costs associated with this alternative reflect both direct cost quotes from contractors and Anchor Environmental LLC's sediment remedial design and construction experience. These costs address all aspects of construction of each alternative and provide specifics on which contractor's quotes were used, in conjunction with engineering assumptions, for various construction elements to arrive at a reliable estimate of costs. Direct quotes from contractors were applied to specific application at the shipyards. Actual costs will only be possible after a complete design package is released under a competitive bidding process with qualified contractors.

Table 18-3. Preliminary cost estimate—Alternative B2: Cleanup to LAET criteria with onsite disposal (CDF)

Item	Unit	Unit Cost	Total Quantity	Total Cost	Assumptions	Direct Cost Quote References[a]	
Direct Construction Costs							
Pre-Construction							
Mobilization/demobilization	LS	$180,000	1	$180,000	Includes equipment for sheetpile installation and CDF construction.	2003 Cost Estimate for Campbell Shipyard (in conjunction with Ninyo & Moore and R.E. Staite)	2002 Cost Estimates for Middle Waterway and Lockheed Shipyards
Site preparation	LS	as shown		$80,000	Includes preparation of area for CDF construction.	As above	As above
Demolition	LS	as shown		$175,000	No specific areas for demolition identified; nominal cost is included for miscellaneous demolition activites.	2003 Demolition costs for Campbell Shipyard (in conjunction with Blaylock Engineering Group)	2001 Cost Estimate for Lockheed Shipyards (demolition input from C.L. Cristich Co.)
Dredging					Includes 1 ft allowable overdredge plus additonal contingency foot.		
Unconstrained open-water dredging (outside of leasehold area)	yd^3	$6	0	$0	$5 to $6 per cubic yard is typical for a large volume of unconstrained dredging outside of shipyard area.	R. Carpenter (R.E. Staite) and B. Lofgren (former Manson Const. Co.), 2002 personal communications	2003 Middle Waterway construction bids (Miller Contracting, Manson Construction)
Constrained dredging from inner shipyard (within leasehold area)	yd^3	$12	67,510	$810,120	Higher cost for dredging within leasehold line, near piers, in areas of ship traffic, etc.	As above	As above
Dredging surface/subsurface debris	yd^3	$80	600	$48,000	Volume based on 1percent of total dredge volume. Includes landfill disposal.	Southwest Marine 2002 communication with Sonas Soil Resource Recovery and La Paz County Landfill, Arizona	Duwamish/Diagonal 2003 bids (Miller Contracting, Manson Construction)
Engineering controls (silt curtain, oil boom)	LS	$70,000	1	$70,000	Silt curtain ($50k) and about 1,500 LF of oil boom at $12/LF.	Hylebos Waterway, 2003 (Miller Contracting/Port of Tacoma)	2002 Cost Estimates for Middle Waterway and Lockheed Shipyards
Protection of Marine Structures and Revetments					Costs assume setback of dredging from marine structures and revetments, and placement of riprap berm to reinstate lateral resistance (as per Anchor 2002).	Aggregate purchase costs obtained 2002 for Campbell Shipyards (in conjunction with URS and Ninyo & Moore)	Techniques documented in Moffat & Nichols (2002) may allow additional sediment removal at significantly increased cost
Stone retaining structures along revetments	Ton	$30	2,500	$75,000		As above	As above
Stone revetment berms along pier faces and dolphins	Ton	$30	9,000	$270,000		As above	As above
Stone revetments along bulkheads	Ton	$30	2,500	$75,000		As above	As above
CDF Construction							
Purchase and install combination sheetpiling	ft^2	$70	46,000	$3,220,000		2003 Cost estimate for Campbell Shipyard (in conjunction with Blaylock Engineering Group)	Preliminary discussions with Triton Engineers (2003)
Tierods and anchoring system	LS	$1,000,000	1	$1,000,000		As above	As above

Table 18-3. (cont.)

Item	Unit	Unit Cost	Total Quantity	Total Cost	Assumptions	Direct Cost Quote References[a]	
Purchase and place sand/gravel surface cover	Ton	$27	9,600	$259,000		2002 Cost estimate for Campbell Shipyards (in conjunction with URS and Ninyo & Moore)	Hylebos Waterway, 2003 (Miller Contracting/Port of Tacoma)
Impermeable asphalt surfacing layer	ft^2	$2	94,000	$188,000		Eagle Harbor, 1997 (Wilder Construction), and WSDOT bids, adjusted	2002 Cost Estimate for Hylebos Waterway
Onsite Disposal of Sediments							
Mechanical placement without treatment (clamshell)	yd^3	$6	29,500	$177,000		B. Lofgren (former Manson Const. Co.), 2002 personal communication	Hylebos Waterway, 2003 (Miller Contracting/Port of Tacoma)
Volume of sediment undergoing stabilization	yd^3	--	38,050	--	All of material placed behind Southwest Marine bulkhead extension, and approximately half of material placed in CDF.	--	--
Purchase cement admixture (8 percent)	Ton	$75	4,566	$342,450		LA Corps of Engineers Pilot Study (Anchor, 2001)	East Waterway, 2001 (Manson Construction)
Mix cement, place stabilized sediment in lifts, compact	Ton	$25	57,075	$1,426,875		Tenth Avenue Marine Terminal, 2001 (R.E. Staite)	East Waterway, 2001 (Manson Construction)
Total Direct Construction Costs				**$8,400,000**			
Construction Management	Percent	8%	2	$672,000		Typical value for cost estimating	--
Design	Percent	15%	2	$1,260,000		Typical value for cost estimating	--
Contingency	Percent	30%	2	$3,099,600		Typical for pre-design cost estimating	--
Monitoring Costs							
Water quality monitoring during construction	Week	$12,000	10	$120,000		Middle Waterway, 2003 (Manson Construction/MWAC)	Hylebos Waterway, 2003 (Miller Contracting/Port of Tacoma)
Post-dredging confirmational sampling	Sample	$10,000	15	$150,000	2 confirmational samples per acre	As above	As above
Natural recovery monitoring						As above	As above
Year 1	10 samples	$75,000	0.6	$45,000	5 acres at each shipyard = 3 samples each.	As above	As above
Year 2	10 samples	$75,000	0.6	$42,417	5 acres at each shipyard = 3 samples each.	As above	As above
Year 5	10 samples	$75,000	0.6	$38,817	5 acres at each shipyard = 3 samples each.	As above	As above
Year 10	10 samples	$75,000	0.6	$33,484	5 acres at each shipyard = 3 samples each.	As above	As above
Long-term groundwater and CDF monitoring	Event	$40,000	8	$283,944	4 events per CDF		
Other (non-construction) Costs							
Permitting/EIR	LS	$400,000	1	$400,000			
Eel grass habitat mitigation	Acre	$100,000	0	$40,000			
CDF 404 compensatory mitigation		TBD					
Habitat evaluation (EFH) costs	LS	as shown		$50,000		Quotes to Southwest Marine, 2003	

Table 18-3. (cont.)

Item	Unit	Unit Cost	Total Quantity	Total Cost	Assumptions	Direct Cost Quote References[a]
Supplemental design sampling	LS	as shown		$100,000		
Construction bid support	LS	$17,500	2	$35,000		
Internal shipyard costs		TBD				
RWQCB past costs	Year	$50,000	4	$200,000		
RWQCB oversight costs	Year	$50,000	6	$300,000		
Grand Total				**$15,300,000**		

Note: CDF - nearshore confined disposal facility
LAET - lowest apparent effects threshold
LS - lump sum
RWQCB - California Regional Water Quality Control Board, San Diego Region

[a] The estimated costs associated with this alternative reflect both direct cost quotes from contractors and Anchor Environmental LLC's sediment remedial design and construction experience. These costs address all aspects of construction of each alternative and provide specifics on which contractor's quotes were used, in conjunction with engineering assumptions, for various construction elements to arrive at a reliable estimate of costs. Direct quotes from contractors were applied to specific application at the shipyards. Actual costs will only be possible after a complete design package is released under a competitive bidding process with qualified contractors.

Table 18-4. Preliminary cost estimate—Alternative C: Cleanup to final reference pool chemistry

Item	Unit	Unit Cost	Total Quantity	Total Cost	Assumptions	Direct Cost Quote References[a]	
Direct Construction Costs							
Pre-Construction							
Mobilization/demobilization	LS	$200,000	1	$200,000	Does not reflect cost of additional remobilizations if more than one construction season is needed. Includes equipment for sheetpile installation and CDF construction.	2003 Cost Estimate for Campbell Shipyard (in conjunction with Ninyo & Moore and R.E. Staite)	2002 Cost Estimates for Middle Waterway and Lockheed Shipyards
Site preparation	LS	as shown		$80,000	Includes preparation of area for CDF construction.	As above	As above
Demolition	LS	as shown		$250,000	No specific areas for demolition identified; nominal cost is included for miscellaneous demolition activites.	2003 Demolition costs for Campbell Shipyard (in conjunction with Blaylock Engineering Group)	2001 Cost Estimate for Lockheed Shipyards (demolition input from C.L. Cristich Co.)
Dredging					Includes 1 foot allowable overdredge plus additonal contingency foot.		
Unconstrained open-water dredging (outside of leasehold area)	yd^3	$6	630,000	$3,780,000	$5 to $6 per cubic yard is typical for a large volume of unconstrained dredging outside of shipyard area.	R. Carpenter (R.E. Staite) and B. Lofgren (former Manson Const. Co.), 2002 personal communications	2003 Middle Waterway construction bids (Miller Contracting, Manson Construction)
Constrained dredging from inner shipyard (within leasehold area)	yd^3	$12	553,440	$6,641,000	Higher cost for dredging within leasehold line, near piers, in areas of ship traffic, etc.	As above	As above
Dredging surface/subsurface debris	yd^3	$80	11,800	$944,000	Volume based on 1 percent of total dredge volume. Includes landfill disposal.	Southwest Marine 2002 communication with Sonas Soil Resource Recovery and La Paz County Landfill, Arizona	Duwamish/Diagonal 2003 bids (Miller Contracting, Manson Construction)
Engineering controls (silt curtain, oil boom)	LS	$100,000	1	$100,000	Silt curtain ($50k) and about 2,000 LF of oil boom at $12/LF.	Hylebos Waterway, 2003 (Miller Contracting/Port of Tacoma)	2002 Cost Estimates for Middle Waterway and Lockheed Shipyards
Protection of Marine Structures and Revetments					Costs assume setback of dredging from marine structures and revetments, and placement of stone blankets or berms to reinstate lateral resistance (as per Anchor, 2002).	Aggregate purchase costs obtained 2002 for Campbell Shipyards (in conjunction with URS and Ninyo & Moore)	Techniques documented in Moffat & Nichols (2002) may allow additional sediment removal at significantly increased cost
Stone retaining structures along revetments	Ton	$30	9,000	$270,000	As above	As above	As above
Stone revetment berms along pier faces	Ton	$30	45,000	$1,350,000	As above	As above	As above
Stone revetments along bulkheads	Ton	$30	5,500	$165,000	As above	As above	As above
Upland Disposal—Regional Landfill							
Upland staging, offload, dewatering area	LS	$100,000	1	$100,000	Accounts for construction of dewatering area on shipyard property; does not include possible leasing price from use of other properties.	Cost Estimate for Campbell Shipyard (in conjunction with Ninyo & Moore and R.E. Staite)	America's Cup Harbor, San Diego
Rehandling and dewatering	yd^3	$14	540,000	$7,560,000	Assume stockpiling for dewatering, with some potential addition of lime or cement admixture to assist.	Port of San Diego, TAMT and Campbell Shipyards (R.E. Staite)	East Waterway and Terminal 25, 2003 (Manson Construction and Hurlen)
Disposal (including transport and tipping fee)	Ton	$50	810,000	$40,500,000	Regional hazardous waste landfill outside of San Diego County (as opposed to $31/ton for Otay or Sycamore landfills). Updated 8/03 from $45/ton (previous cost assumptions) per experience with PCB disposal (Campbell shipyard).	Southwest Marine 2002 communication with Sonas Soil Resource Recovery and La Paz County Landfill, Arizona	2003 Cost Estimate for Campbell Shipyard (in conjunction with Ninyo & Moore and R.E. Staite)

Table 18-4. (cont.)

Item	Unit	Unit Cost	Total Quantity	Total Cost	Assumptions	Direct Cost Quote References[a]	
Open Water Disposal							
Disposal (including transport and tipping fee)	yd^3	$8	575,000	$4,600,000	Unit rate based on discussion with local contractors, assuming haul to LA-5 disposal site. 291,000 X 0.3 = 87,300 yd^3	NASSCO Drydock berth expansion, 1998	B. Lofgren (former Manson Const. Co.), 2002 personal communications
CDF Construction							
Purchase and install combination sheetpiling	ft^2	$70	46,000	$3,220,000		2003 Cost estimate for Campbell Shipyard (in conjunction with Blaylock Engineering Group)	Preliminary discussions with Triton Engineers (2003)
Tierods and anchoring system	LS	$1,000,000	1	$1,000,000		see above	see above
Purchase and place sand/gravel surface cover	Ton	$27	9,600	$259,000		2002 Cost estimate for Campbell Shipyards (in conjunction with URS and Ninyo & Moore)	Hylebos Waterway, 2003 (Miller Contracting/Port of Tacoma)
Impermeable asphalt surfacing layer	ft^2	$2	94,000	$188,000		Eagle Harbor, 1997 (Wilder Construction), and WSDOT bids, adjusted	2002 Cost Estimate for Hylebos Waterway
Onsite Disposal of Sediments							
Mechanical placement without treatment (clamshell)	yd^3	$6	33,500	$201,000		B. Lofgren (former Manson Const. Co.), 2002 personal communication	Hylebos Waterway, 2003 (Miller Contracting/Port of Tacoma)
Volume of sediment undergoing stabilization	yd^3	--	36,000	--	All of material placed behind Southwest Marine bulkhead extension, and approximately half of material placed in CDF	--	--
Purchase cement admixture (8%)	Ton	$75	4,320	$324,000		LA Corps of Engineers Pilot Study (Anchor, 2001)	East Waterway, 2001 (Manson Construction)
Mix cement, place stabilized sediment in lifts, compact	Ton	$25	54,000	$1,350,000		Tenth Avenue Marine Terminal, 2001 (R.E. Staite)	East Waterway, 2001 (Manson Construction)
Total Direct Construction Costs				**$73,100,000**			
Construction Management	Percent	8%		$5,848,000		Typical value for cost estimating	--
Design	Percent	15%		$10,965,000		Typical value for cost estimating	--
Contingency	Percent	30%		$26,973,900		Typical for pre-design cost estimating	--
Monitoring Costs							
Water quality monitoring during construction	Week	$12,000	65	$780,000		Middle Waterway, 2003 (Manson Construction / MWAC)	Hylebos Waterway, 2003 (Miller Contracting/Port of Tacoma)
Post-dredging confirmational sampling	Sample	$10,000	284	$2,840,000	2 confirmational samples per acre	As above	As above
Long-term groundwater and CDF monitoring	Event	$40,000	8	$283,944	4 events per CDF	As above	As above
Other (Non-Construction) Costs							
Permitting/EIR	LS	$400,000	1	$400,000			
Eelgrass habitat mitigation	Acre	$100,000	1.5	$150,000			
CDF 404 compensatory mitigation		TBD				--	--

Table 18-4. (cont.)

Item	Unit	Unit Cost	Total Quantity	Total Cost	Assumptions	Direct Cost Quote References[a]	
Habitat evaluation (EFH) costs	LS	as shown		$50,000		Quotes to Southwest Marine, 2003	
Supplemental design sampling	LS	as shown		$200,000			
Construction bid support	LS	$17,500	2	$35,000		Middle Waterway, 2003 (Anchor Environmental)	
Internal shipyard costs		TBD				--	--
Past costs						--	--
RWQCB oversight costs	Year	$50,000	6	$300,000			
Grand Total				**$121,900,000**		--	--

Note: CDF - nearshore confined disposal facility
LF - linear foot
LS - lump sum
RWQCB - California Regional Water Quality Control Board, San Diego Region

[a] The estimated costs associated with this alternative reflect both direct cost quotes from contractors and Anchor Environmental LLC's sediment remedial design and construction experience. These costs address all aspects of construction of each alternative and provide specifics on which contractor's quotes were used, in conjunction with engineering assumptions, for various construction elements to arrive at a reliable estimate of costs. Direct quotes from contractors were applied to specific application at the shipyards. Actual costs will only be possible after a complete design package is released under a competitive bidding process with qualified contractors.

Table 19-1. Estimated fraction of aquatic life impairment under current conditions and cleanup alternatives

Impairment Category[a]	Category as Probability	Effective Impairment Factor[b]	Current: Number of Stations Affected	Current: Fraction of Stations Affected	Current: Effective Impairment	Remedial Alternative B[c]: Number of Stations Affected	Remedial Alternative B[c]: Fraction of Stations Affected	Remedial Alternative B[c]: Effective Impairment	Remedial Alternative C[d]: Number of Stations Affected	Remedial Alternative C[d]: Fraction of Stations Affected	Remedial Alternative C[d]: Effective Impairment
Highly likely	0.95	0.475	7	0.23	0.11	6	0.20	0.095	0	0.00	0
Likely	0.75	0.375	2	0.07	0.025	2	0.07	0.025	2	0.07	0.025
Possible	0.25	0.125	13	0.43	0.054	14	0.47	0.058	20	0.67	0.083
Unlikely	0.05	0.025	8	0.27	0.007	8	0.27	0.007	8	0.27	0.007
Sums					**0.20**			**0.19**			**0.12**

[a] See Section 9.2 of the text; probabilities corresponding to the categories are assigned based on judgment.

[b] Based on maximum current impairment of 50 percent of benthic macroinvertebrate abundance.

[c] Assumes remediated stations are subsequently likely to have possible effects, because chemicals exceeding LAET are not causes of effects.

[d] Stations with disturbance are assumed to have likely effects even after cleanup; others are assumed to have possible effects.

Table 19-2. Comparison of overall long-term beneficial use conditions for current shipyard status and dredging alternatives

Beneficial Use	Relative Value	Current Shipyard Status (percent)	Alternative A (percent)	Alternatives B1 or B2 (percent)	Alternative C (percent)
Aquatic life	1	80	80	82	89
Aquatic-dependent wildlife	1	100	100	100	100
Human health	2	100	100	100	98
Overall		**95**	**95**	**95**	**96**

Notes: A condition of 100 percent represents no impairment.

Aquatic life conditions reflect the impairments calculated in Table 19-1.

The human health condition for Alternative C reflects increased risk from dredging and transportation activities.

Overall beneficial use condition is the average of the individual conditions, weighted by their relative value.