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Impact of offsite sediment transport and toxicity on remediation of a contaminated estuarine bay

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Abstract

Sediment of Ostrich Bay, an arm of Dyes Inlet on Puget Sound, was historically contaminated with ordnance compounds from an onshore US Navy facility. An initial recommendation for a sediment cover to mitigate benthic risks was followed by studies of sediment transport and deposition to determine whether contaminated sediment from Dyes Inlet or other offsite sources in Puget Sound may contribute to Ostrich Bay impacts. A Sediment Trend Analysis (STA[®]) identified net sediment transport pathways throughout the bay and inlet by examining changes in grain size distributions in multiple adjacent samples. Results indicated that fine-grained sedimentary material transports into and deposits throughout the Dyes Inlet system, with no erosion or transport out of Ostrich Bay. Echinoderm larvae mortality bioassay results were elevated in fine-grained sediments of both Ostrich Bay and Dyes Inlet. Ordnance compounds were undetected, and although sediment mercury concentrations were elevated at 0.48–1.4 mg/kg in both waterbodies, the relationship with toxicity was weak. Results of the studies and sedimentation modeling indicate that impacted sedimentary material deposits throughout the Dyes Inlet/Ostrich Bay system from unknown sources and will prevent natural recovery of Ostrich Bay as well as negate long-term effectiveness of active remedial measures. Stakeholders have recognized that remediation of the bay can be achieved only after the toxicity of depositing sediment decreases.

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1. Introduction

Puget Sound, WA, has been recognized by US EPA as an area of concern for adverse biological or ecological effects due to contaminated sediment (USEPA, 1997). Sediments of various bays and inlets of the sound contain greatly elevated concentrations of metals, petroleum hydrocarbons, and other chemicals from past industrial activities and urban development (Battelle, 1989; Malins et al., 1984; Lefkovitz et al., 1997). The sources of sediment contaminants have not always been easily discernable; their identification can be hampered by incompletely understood processes that govern transport and deposition of sedimentary particles. In this paper we present the results of sediment transport and sediment impact studies that greatly influenced

remediation plans for contaminated sediments of a Puget Sound bay.

The western shoreline of Ostrich Bay, an arm of Dyes Inlet on Puget Sound, was the site of a Naval ordnance formulation facility from World War I until 1959 (Fig. 1). The sediment of the bay was contaminated with ordnance and nitroaromatic compounds that either washed from the facility or were disposed directly into the bay. Sediment also contains mercury and a few semi-volatile organic compounds at concentrations above nearby reference areas. Because of the chemical contamination, the marine environment of Ostrich Bay was designated an operable unit of the Jackson Park Housing Complex/Naval Hospital-Bremerton CERCLA site in 1989. The bay is shallow (<6 m MLLW) and is underlain with glacial deposits typical of the Puget Sound area. In addition to a Navy housing facility and hospital on the site of the original ordnance facility, land use around the bay consists of rural residential neighborhoods and forested land.

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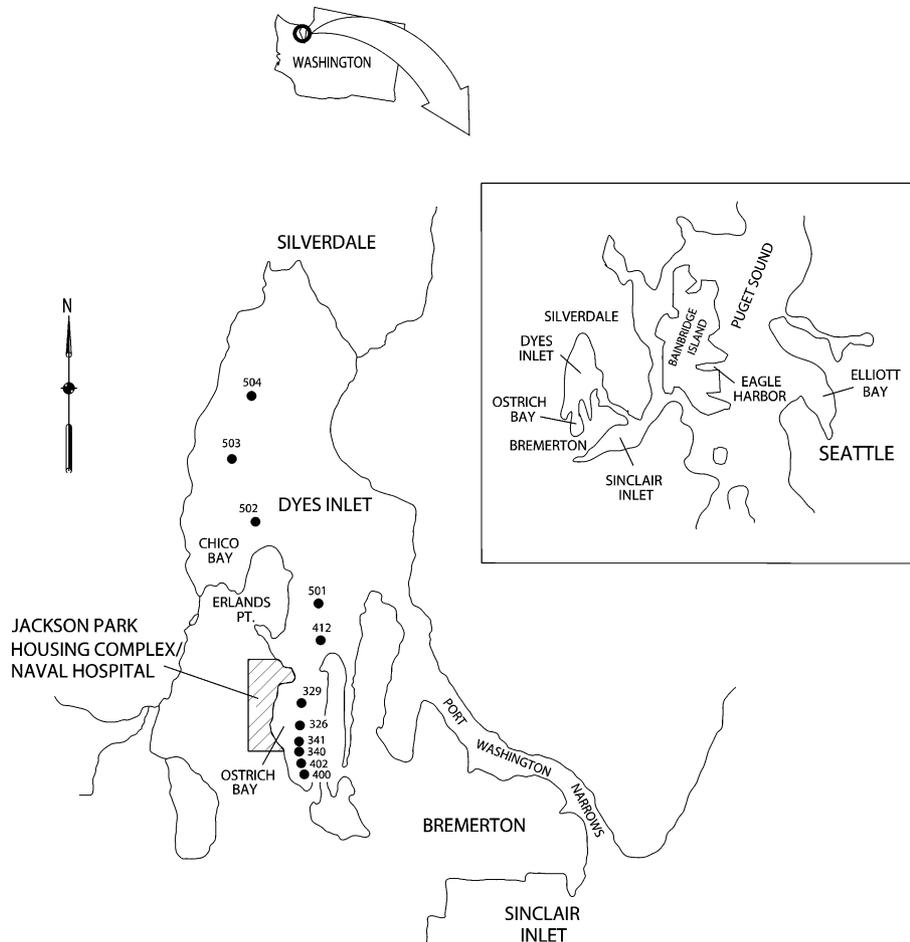


Fig. 1. Map of the study area on Puget Sound, WA. Sample areas are shown for Ostrich Bay and Dyes Inlet. Sample stations in the “500” series were located to characterize sediment transport gyres in Dyes Inlet. Stations located at 412 and 501 are grouped as “southern Dyes Inlet”; stations located at 503 and 504 are grouped as “northern Dyes Inlet”.

Sediment management standards of the Washington Department of Ecology (Ecology, 1991) were used during a 1994 remedial investigation of Ostrich Bay to evaluate sediment ecological risks and remediation alternatives to mitigate risks. Due to the lack of regulatory criteria for ordnance compounds in sediment, the bay was evaluated by biological criteria consisting of sediment bioassays. Most sediment stations in the bay were found to exceed the regulatory cleanup levels based on echinoderm larvae and/or amphipod survival bioassays. State regulators identified most of the sediments of the bay for active remediation, primarily by a sediment cover. However, the cause of sediment toxicity was unclear, since concentrations of ordnance compounds were either undetectable or relatively low (e.g., $<100 \mu\text{g}/\text{kg dw}$) and their toxicity to native marine benthos was unknown at the time. Although mercury concentrations up to $0.53 \text{ mg}/\text{kg}$ exceeded the average reference area concentration of $0.12 \text{ mg}/\text{kg}$, regression analyses found no significant correlation between the mercury concentration data and sediment toxicity. There was also no

known local source for the mercury in Ostrich Bay sediment.

Prior to the 1994 studies, a reconnaissance survey of Ostrich Bay (Battelle, 1989) had suggested that elevated mercury in the bay ($0.39 \text{ mg}/\text{kg}$) might be related to higher concentrations found in sediment samples collected from Dyes Inlet (range from 0.66 to $0.79 \text{ mg}/\text{kg}$) (see Fig. 1). In addition, a physical model of tidal currents (McGary and Lincoln, 1977) showed that surface currents in the Dyes Inlet system bifurcate into a clockwise current into northern Dyes Inlet and a counterclockwise current into southern Dyes Inlet, including Ostrich Bay. Based on these characteristics, it was hypothesized that impacted sediment in Ostrich Bay may be related to sediment conditions in Dyes Inlet. Impacted sediment of Dyes Inlet might be moving into Ostrich Bay during winter storms accompanied by strong north winds, or sediment conditions in both Ostrich Bay and Dyes Inlet could be influenced by material coming into the Dyes Inlet system through Port Washington Narrows (see Fig. 1).

To determine whether impacted sediment in Ostrich Bay is related to the sediment in Dyes Inlet, the following information needs were identified: Is Ostrich Bay depositional? If so, where is sedimentary material coming from, and can it erode and transport out of the bay? If the sedimentary material is coming into the bay through Dyes Inlet, is the incoming material contaminated and toxic, and to what degree? To answer these questions, a series of studies were performed to calculate sedimentation rates for the bay, to model sediment transport and erosion potential, and to characterize the quality of sediment in the bay and of incoming sedimentary material by chemical analyses and toxicity testing. The objectives of this paper are to describe the results of these studies and to place them in context with management options for Ostrich Bay sediment. Ultimately, the studies were intended to answer the question of whether incoming sedimentary material contributes to the observed impacts in bottom sediment of Ostrich Bay or whether the observed impacts are related only to local sources (e.g. former activities at the onshore Navy facility).

2. Materials and methods

2.1. Sedimentology

Rates of natural sedimentation for Ostrich Bay were calculated from ^{210}Pb measurements on sediment cores. Sediment cores were collected in 1996 from three stations located in the mid-basin that are characterized by fine-grained sediment (mean percent fines, i.e. silt plus clay fractions, of 78%), and from three stations located in nearshore areas of the bay that are characterized by relatively coarse-grained sediment (mean of 33% fines). Sediments were collected with a 6-in. square, stainless steel Karsten gravity corer that minimizes shortening of the sediment core (Nevissi et al., 1989). Collection procedures and quality assurance measures followed Puget Sound Estuary Program protocols (USEPA, 1995). Core samples were sliced every 1 or 2 cm to depths of 60 cm for measurement of ^{210}Pb activity and mercury concentrations. Mercury was analyzed in sediment slices to further track the core dating (Santschi et al., 1999).

The activity of ^{210}Pb was determined from alpha particle counts from the granddaughter polonium-210 (Koide et al., 1973), the preferred approach for accurate sedimentation determination (Santschi et al., 1999). Sedimentation rates can be calculated when the unsupported ^{210}Pb activity is close to that of atmospheric fallout, under assumed steady-state conditions (Santschi et al., 1999). At the relatively high sedimentation rates found in Puget Sound, excess ^{210}Pb activity, with a half-life of 22.3 yr, has been proven useful for calculating sedimentation rates (Lavelle et al., 1986; Gobeil et al.,

1999). This approach has been used to date other Puget Sound sediments (Lefkovitz et al., 1997).

Excess ^{210}Pb was calculated as the difference between total and supported activities, where excess ^{210}Pb is supplied to the sediments from atmospheric deposition of releases from ^{226}Ra in soil, and supported ^{210}Pb arising from ^{226}Ra decay within the sediments. Supported ^{210}Pb activities were calculated from constant ^{210}Pb activities at depth (Santschi et al., 1999). Depth intervals used to calculate sedimentation rates were below the surficial bioturbation, or mixed layer, sediments. Sediment samples were analyzed for mercury using cold-vapor atomic absorption spectroscopy (Bloom and Creclius, 1987). Wet densities of sediment slices were calculated from sediment dry weights using a nonlinear relationship observed for the two metrics in Puget Sound (Lefkovitz et al., 1997). Sedimentation rates in gm/yr at each station were calculated from mass settling rates in $\text{g/cm}^3/\text{yr}$, determined from the core ^{210}Pb data, and surficial layer porosity estimates derived from the wet densities.

2.2. Sediment dynamics

Sediment Trend Analysis (STA[®]) was used to map sediment transport in the Dyes Inlet and Ostrich Bay system (Fig. 1). STA is a technique whereby patterns of net sediment transport are determined from relative changes in the grain size distributions of bottom sediments. Its methodology relates sediments in a given transport direction by a sediment transfer function that describes the relative probability of any particular size being moved (i.e. eroded, transported, and deposited). The shape of the sediment transfer function also determines the nature of the processes resulting in transport, and provides the interpretation with respect to erosion, deposition, or dynamic equilibrium. Directions of transport are determined statistically, based on the numbers of specific grain size trends exceeding random probability over the sampled area. A description of the theory and methodology for the STA is provided in McLaren and Bowles (1985). The approach has been used successfully to identify transport pathways for contaminated sediment in Pacific Northwest water bodies (McLaren and Little, 1987).

For the STA, surficial sediment samples were collected in 1997 from 404 stations throughout the Dyes Inlet system, including nearshore and intertidal flats. Stations were located on a grid of 200-m intervals to the north and west of the mid-point of Port Washington Narrows (Fig. 1). Some areas, particularly in the Port Washington Narrows, were not sampled because of hard substrate.

All samples were analyzed for complete grain-size distribution using a Malvern 26001 laser particle sizer, which uses lenses of different focal lengths to measure portions of the total range of grain sizes. Sieve data were

collected for grain sizes >1500 microns and merged using an algorithm developed by GeoSea Consulting (McLaren and Bowles, 1985). The distributions were entered into a microcomputer equipped with proprietary software to establish sediment trends and transport functions. Patterns of net sediment transport were determined over two-dimensions by exploring for sample sequences that produced statistically acceptable trends. For each trend, the function $X(s)$ was calculated as:

$$X(s) = d_2(s)/d_1(s)$$

where s is the grain size, $d_1(s)$ is the grain size at Station D1, and $d_2(s)$ is the grain size at Station D2.

The $X(s)$ distribution is compared with grain size distributions of the deposits $d_1(s)$ and $d_2(s)$, and is interpreted as follows (McLaren and Bowles, 1985): Under dynamic equilibrium, the shape of the X -distribution closely resembles $d_1(s)$ and $d_2(s)$. The probability of finding a particular sized grain in the deposit is equal to the probability of its transport and re-deposition; i.e. the bed is neither accreting nor eroding and is, therefore, in dynamic equilibrium. Under net accretion, the shapes of the three distributions are similar but the modes of $d_1(s)$ and $d_2(s)$ are coarser than the mode of X , and sediment is thus more readily deposited than transported. Under net erosion, again the shapes of the three distributions are similar but the modes of $d_1(s)$ and $d_2(s)$ are finer than the mode of X , and sediment is thus more easily eroded than deposited. Under total deposition, the X -distribution more or less increases monotonically over the complete size range of the deposits and sediments fine in the direction of transport. However, the bed is no longer mobile and is accreting under a “rain” of sediment that fines with distance. Once deposited there is no further transport. Total deposition is usually confined to cohesive, muddy sediments. More recently, it has been found that the X -distribution can be horizontal, which occurs when sediments are found far from their source and their deposition is no longer related strictly to size-sorting. In other words, there is an equal probability of all sizes being deposited. These trends occur where mean grain-size is very fine silt or clay (McLaren et al., 1993).

A final interpretation of a distribution was accepted only when all, or nearly all, of the samples were contained in mutually supporting sequences that produced a coherent pattern over the entire study area. Separate trends analyses were undertaken on the different facies (coarse and fine-grained materials) that were found to be present in Dyes Inlet and Ostrich Bay sediments.

The sediment transport study was supplemented by a fingerprint analysis of sediment metals in Ostrich Bay, Dyes Inlet, and Sinclair Inlet, which is located immediately southwest of the mouth of Port Washington Narrows (see Fig. 1). Fingerprinting was performed to identify potential relationships among neighboring embayments regarding sediment contamination. Comple-

mentary metals data for each bay were available from studies performed in 1984 and 1988 (Battelle, 1986, 1989). Stations used from these studies were limited to fine-grained sediments (i.e. silty sands, silts, and silty muds). Data on average metals concentrations were compared among the embayments, and ratios were generated of average concentrations of metals in Ostrich Bay sediment to those in other embayments.

2.3. Sediment quality

Evaluation of sediment quality consisted of collecting information on the chemistry and toxicity of sediments in Ostrich Bay and Dyes Inlet. Sediments were sampled at various locations in 1997. Stations were selected to represent the range of depths, chemical concentrations, and bioassay responses observed in the 1994 remedial investigation and in previous studies (e.g. Battelle, 1989). Six stations were selected in the mid-basin with fine-grained fraction greater than 50%, two stations of similar fine-grained sediment in southern Dyes Inlet (which includes the mouth of Ostrich Bay), and two stations in northern Dyes Inlet (Fig. 1). Additional stations located in nearshore and intertidal areas characterized by coarse-grained sediment were not used in the following analyses. Sediment was also collected in duplicate from two stations in Carr Inlet, a reference bay located in central Puget Sound with unimpacted sediment (Ecology, 1991). All stations were located by differential Global Positioning System to match stations sampled during 1994 (URS, 1994) and to match four stations sampled during 1988 (Battelle, 1989).

Sediment samples were collected from the top 4-cm horizon with a 0.1 m² van Veen grab sampler. Collection procedures and sample preparation and storage followed Puget Sound Estuary Program protocols (USEPA, 1996). Sediment bioassays were performed using the 10-day amphipod (*Ampelisca abdita*) mortality test and the echinoderm larvae (*Dendraster excentricus*) effective mortality test, following protocols for conducting bioassays on Puget Sound sediments (USEPA, 1995). The echinoderm larvae test exposes fertilized embryos in five replicate chambers to sediment that is stirred and mixed in seawater. Although the test has been criticized for being a combined solid phase/elutriate test using a pelagic species that may not directly relate to assessing toxicity to benthos (Loehr, 1999), it has been defended as a realistic surrogate for testing post-larval recruitment and survival processes (Dinnel, 2000). In addition, the echinoderm larvae test has been standardized for national use (ASTM, 2001) and is presently required by Washington state regulations for contaminated sediment sites (Ecology, 1991). The test was the most sensitive in the present and previous studies of the site (URS, 1994), and was found to be the most sensitive test at the site in more recent studies (Carr et al., 1998; Carr

and Nipper, 1999). It is considered to be sensitive to a variety of chemical contaminants, but not to grain size or other physical parameters (USEPA, 1995), and is the major focus of sediment quality and regulatory concerns at this site. Toxicity-based cleanup criteria for Ostrich Bay sediments were set at 45.3% mortality for the amphipod bioassay and 38.9% total effective mortality for the echinoderm larvae bioassay, following Washington state guidelines (Ecology, 1991).

Chemical and conventional sediment analyses were performed following Puget Sound Estuary Program protocols. Mercury was analyzed by cold vapor atomic adsorption; cadmium by ICP; nitroaromatic and ordnance compounds by modified EPA method 8330 using HPLC (USEPA, 1990); total sulfides by EPA method 9030A (USEPA, 1990) and unionized ammonia in sediment porewater and bioassay overlying water by ion selective ammonia electrode and pH meter followed by conversion from total ammonia (Bower and Bidwell, 1978).

Temporal trends in mercury concentrations were investigated for Ostrich Bay, Dyes Inlet, and Carr Inlet. Data for 1988 were taken from the earlier reconnaissance survey (Battelle, 1989) and consist of Stations 4, 5 and 6 for Dyes Inlet, and Station 2 for Ostrich Bay. Data for 1994 were taken from the Remedial Investigation for the site (URS, 1994) and consist of seven stations in Ostrich Bay including OB 326 in the mid-bay, and four stations in Carr Inlet. A single mercury concentration for 1996 comes from the surficial layer of the sediment core that was collected from Station OB 326 for the sedimentation rate determination (EA, 1998; and present study). Data for 1997 are from the present study (EA, 1998), and consist of six stations in Ostrich Bay including OB 326, Stations 502, 503, 504 in Dyes Inlet, and four stations in Carr Inlet. Stations in the 1997 study were matched by approximate location with the reconnaissance survey stations of 1988 (Battelle, 1989); i.e., Stations 502, 503, and 504 in Dyes Inlet in 1997 were matched with the 1988 data from Stations 4, 5 and 6, respectively; and 1997 data from Station OB 326 in Ostrich Bay was matched with 1988 data from Station 2. Historical data and data from the present study are from surficial sample collections (4–5 cm) with >50% fines, except for the 1997 Carr Inlet samples at 20% fines. Student's *t* test was used for statistical comparison between means of the 1994 and 1997 data sets of Ostrich Bay mercury concentrations.

3. Results and discussion

3.1. Sedimentology

^{210}Pb activity-depth profiles of Ostrich Bay sediment cores typically showed a plateau in the top 10–16 cm with

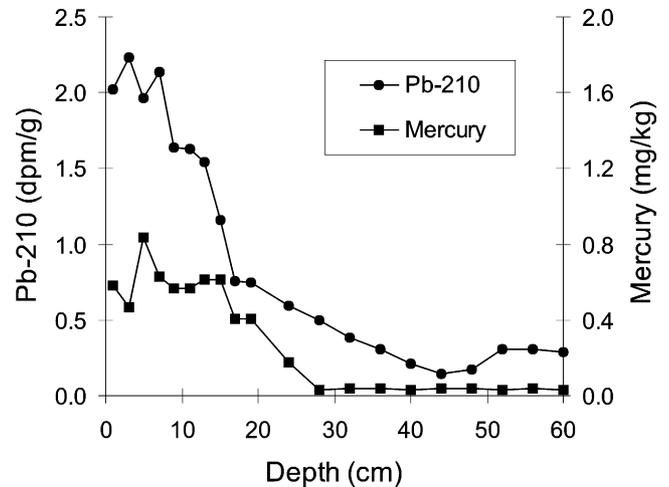


Fig. 2. Lead-210 activity-depth and mercury concentration-depth profiles for fine-grained sediment in the mid-basin of Ostrich Bay.

a bottoming out around 0.2–0.4 decays per minute (dpm)/g at 45 cm (Fig. 2), a pattern that is consistent with bioturbation depths in shallow bays of Puget Sound and with background activity in Puget Sound (Lefkovitz et al., 1997). Mercury concentrations also typically showed a plateau in surficial sediments from 0.4 to 0.8 mg/kg (Fig. 2). Concentrations decreased to less than 0.1 mg/kg at 30–40 cm depth, which approaches the background concentration of 0.04 mg/kg reported for deeper sediment of the Sound dated prior to 1900 (Romberg et al., 1984). An average sedimentation rate for mid-basin, fine-grained material (i.e., >50% fines) was calculated from ^{210}Pb and sediment density data at 0.8 cm/yr.

The ^{210}Pb activity-depth and mercury concentration-depth profiles are consistent with both net deposition in the bay and deposition and erosion without net sedimentation. For example, surficial bioturbators could be responsible for mixing the top 10–16 cm of sediment, and deeper bioturbators could less efficiently mix underlying sediment down to 30–40 cm depth, where ^{210}Pb activity and mercury concentrations bottom out. Bioturbation has long been known to produce a surface mixed layer and to redistribute contaminants deeper into the sediment (Guinasso and Schink, 1975). The net effect of such turnover in sedimentary material and decreasing bioturbation with depth would make it difficult to estimate a net natural sedimentation rate. However, results of the sediment transport study described below clearly show that fine-grained material is transported and deposited as fresh sediment in Ostrich Bay, and there is no erosion or removal from the bay. Additionally, modeling of sediment dynamics from tidal currents and bottom currents generated by wind action (USACE, 1984) estimated negligible rates of erosion of Ostrich Bay sediment (EA, 1998). Results of those studies support the interpretation that the ^{210}Pb activity-depth profiles are due to net deposition of sediment material.

3.2. Sediment dynamics

The STA connected neighboring sediment stations on the basis of progressive changes in grain size distributions. From this, lines were constructed connecting sediment stations along transport pathways. The constructed lines for both Dyes Inlet and Ostrich Bay are shown for sandy sediments in Fig. 3a and for fine-grained sediments in Fig. 3b. Detailed results and conclusions of the STA are provided in a technical report (GeoSea Consulting, 1997) and are summarized below.

In the Dyes Inlet basin, transport lines for sandy sediments originate at the north end of the Port Washington Narrows where sandy sediments first become deposited by flood currents. Sediments inside the narrows were too coarse for sampling. The “jet stream” that emerges from Port Washington Narrows bifurcates into two gyres along the western basin (Fig. 3). Some sandy material in the south gyre transports along shallow isobaths into Ostrich Bay. The pattern of transport suggests a flood tidal delta at the entrance into Dyes Inlet, as evident from the 60-ft isobath in Fig. 3a. Transport lines along the shoreline frequently start at the shoreline rather than continuing from the main basin lines, which suggests that local shoreline erosion is the source of sandy sediments in these transport lines.

Transport lines for fine-grained sediments in Dyes Inlet and Ostrich Bay showed one depositional environment: they originate in the center of Dyes Inlet basin

and their trends continue from where the deposition of sandy sediments terminate (Fig. 3b). Similar to the trends with sandy sediments, the fine-grained sediments also bifurcate into two gyres with a counterclockwise circulation in the south half of the basin, from which fine-grains are transported south into Ostrich Bay. Most of the trends for fine-grained sediment show more or less horizontal X-distributions, indicating that particles are far from their source and are so fine that size differentiation during deposition is minimal (McLaren et al., 1993).

These results indicate that the sources of sedimentary material to Ostrich Bay consist of fine-grained material transported through Port Washington Narrows and Dyes Inlet, and coarser material derived from shoreline erosion. The fine-grained sediments in Ostrich Bay and most of Dyes Inlet are formed under a “rain” of particles that do not move once deposited, and there is no transport of sediment north out of Ostrich Bay.

The distribution of sediment types in the Dyes Inlet system differs from the typical pattern for Puget Sound bays in that coarse sediments characterize the deepest parts of the main basin of the inlet, whereas mud is found in the shallower water. The converse is true for nearly all of Puget Sound (Evans-Hamilton, 1987). The pattern in Dyes Inlet results from the finer particles in the high energy discharge from Port Washington Narrows depositing in shallower areas farther down the transport path, including Ostrich Bay. The limited

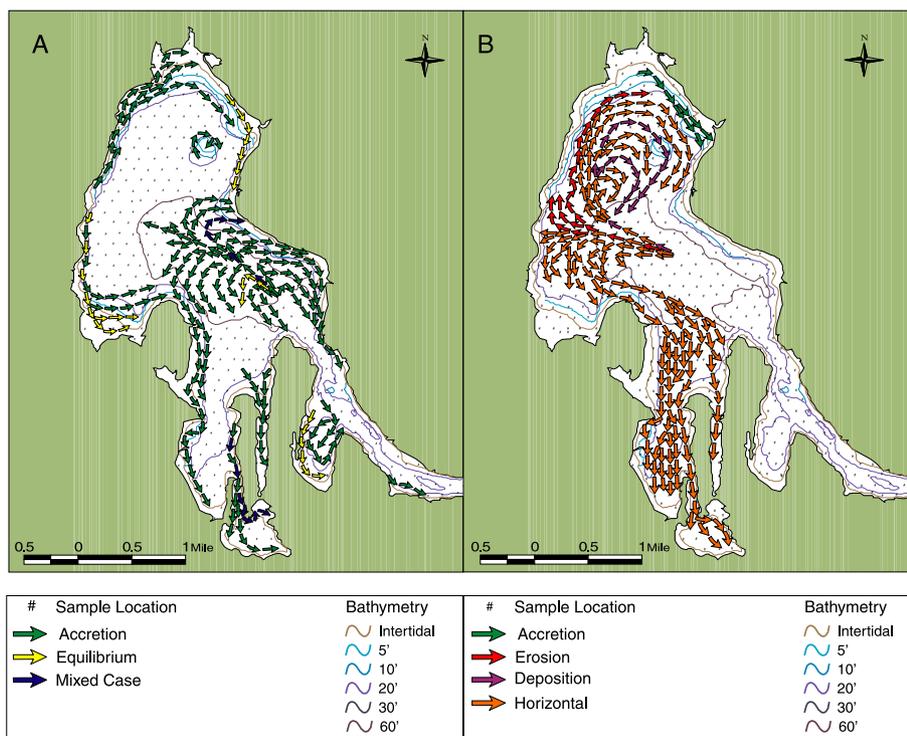


Fig. 3. Net sediment transport pathways for Dyes Inlet and Ostrich Bay sediments. (a) Transport pathways for sandy sediment; (b) transport pathways for fine sediment (clay plus silt fractions). Transport pathways were determined by STA.

Table 1
Comparison of sediment metals concentrations in Ostrich Bay with nearby embayments

Metal	Mean concentration (mg/kg)				Ratios	
	Ostrich bay ^a	Dyes inlet ^a	Sinclair inlet ^b	Sequim bay ^c	Dyes inlet/ostrich bay	Sinclair inlet/ostrich bay
Arsenic	9.4	18.6	20.2	6.7	2.0	2.1
Mercury	0.35	0.72	1.3	0.06	2.1	3.7
Cadmium	0.89	1.2	1.6	1.0	1.4	1.8
Copper	47.5	81.0	238.5	45.8	1.7	5.0
Lead	40.6	70.2	195.7	8.3	1.7	4.8
Zinc	96.5	153.7	286.0	83.0	1.6	3.0
Silver	0.49	1.0	1.4	NA	2.0	2.9

Locations of sample areas in central Puget Sound are shown in Fig. 1; the reference bay is located in northern Puget Sound. Concentrations are arithmetic means. NA = not available.

^a Ostrich bay $n = 2$; dyes inlet $n = 3$. Data collected in 1988 (Battelle, 1989).

^b Sinclair Inlet $n = 6$. Data collected in 1984 (Battelle, 1986).

^c Reference area, $n = 4$. Data collected in 1984 (Battelle, 1986).

confines of Dyes Inlet itself forces the flow from the narrows into the two transport gyres depicted in Fig. 3.

A comparison of metals concentrations in fine-grained sediments of Ostrich Bay, Dyes Inlet, and Sinclair Inlet during the mid-1980s is shown in Table 1. Numeric ratios of Dyes Inlet metals concentrations to Ostrich Bay concentrations ranged from 1.4 to 2.1. The relatively narrow range of ratios is consistent with the finding of the STA that the fine-grained sediments in Ostrich Bay and Dyes Inlet likely come from the same source, with Ostrich Bay sediments being diluted up to two-fold by local uncontaminated sources. Comparison of sediment metals concentrations from nearby Sinclair Inlet with Ostrich Bay showed a wider range of ratios from 2.0 to 6.4 (Table 1), reflective of the disproportionately elevated concentrations of copper and lead in Sinclair Inlet sediment compared to Ostrich Bay or Dyes Inlet (Battelle, 1989).

3.3. Sediment quality

Echinoderm larvae total effective mortalities at all fine-grained sediment stations of Ostrich Bay and Dyes Inlet (>50% fines) were elevated over reference area responses, and also exceeded the regulatory toxicity-based sediment quality standard of 38.9% response (Table 2). In contrast to the echinoderm larvae test, amphipod survivals for Ostrich Bay and Dyes Inlet stations were not significantly different from reference area stations.

Results of the STA indicate that the fine-grained material that settles in Ostrich Bay is an extension of the material that covers the southern area of Dyes Inlet near the mouth of the bay. Southern Dyes Inlet sediment can thus be considered representative of depositing material that transports through the mouth and into Ostrich Bay. Because the sediment in southern Dyes Inlet has not been impacted by Navy activities at Ostrich Bay, and sediment in Ostrich Bay does not transport northward into southern Dyes Inlet, the sediment of southern Dyes

Table 2
Sediment toxicity in the Dyes Inlet system

Sediment area	n	Amphipod (percent mortality)	Echinoderm larvae (percent effective mortality)
Northern Dyes Inlet	2	20 ± 4.2	47 ± 25
Southern Dyes Inlet ^a	2	23 ± 20	42 ± 10
Ostrich Bay	6	19 ± 2.9	47 ± 10
Carr Inlet (reference area)	4	21 ± 5.7	8.9 ± 2.7

Data are mean ± standard deviation.

^a The area identified as “Southern Dyes Inlet” is located in the southern gyre of Dyes Inlet at the mouth of Ostrich Bay, and is considered representative of fine-grained sedimentary material that deposits in Ostrich Bay (see Fig. 3b).

Inlet can be considered a surrogate for the incoming fine-grained sedimentary material that deposits in Ostrich Bay. As shown in Table 2, the average toxicity of fine-grained sediment in Ostrich Bay, based on the echinoderm larvae bioassay, is no different from the toxicity in sediment from southern or northern Dyes Inlet. Since the primary source of these sediments is outside the Dyes Inlet system, it can be inferred that impacted, and apparently toxic, fine-grained sedimentary material enters the Dyes Inlet system from Port Washington Narrows and deposits throughout the Inlet and Ostrich Bay.

Ordnance compounds in sediments of Ostrich Bay were not detected at detection limits of 100 µg/kg during 1997. Ordnance compounds were also undetectable in Ostrich Bay sediment in a more recent study (Carr and Nipper, 1999). Mercury, however, was detected in fine-grained sediments of both Ostrich Bay and Dyes Inlet during 1997 at concentrations that exceeded the state regulatory AET-based sediment quality criterion (0.42 mg/kg), at ranges of 0.54–1.4 and 0.48–0.74 mg/kg, respectively. Average mercury concentrations in sediments of Ostrich Bay, Dyes Inlet, and the reference area of Carr Inlet during 1997 are depicted in Fig. 4. Regression analysis found a significant but weak correlation of

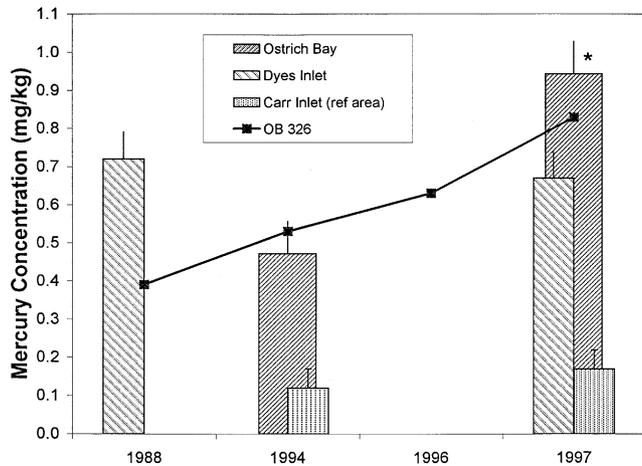


Fig. 4. Temporal comparisons of sediment mercury concentrations. Data are mean and standard deviation. Data sources and sample sizes as in text. * = significantly greater than 1994 Ostrich Bay mean concentration at $p < 0.01$.

mercury concentrations in fine-grained sediments in Ostrich Bay with echinoderm larvae toxicity, at $R^2 = 0.67$. It is unknown how much of a role mercury has played in the bioassay responses of Ostrich Bay, particularly since toxicity more strongly correlated with percent fraction of fine grains ($R^2 = 0.79$), to which mercury concentrations also strongly correlated. No other chemicals were elevated above state regulatory criteria, and regressions for toxicity were not significant with unionized ammonia in sediment porewater or in overlying water of the bioassays, or for total sulfides in porewater or bioassay water.

More recent studies by the US Geological Survey (Carr et al., 1998) found that sediment samples from multiple stations in Ostrich Bay were toxic in echinoderm larvae tests. Concentrations of chlorinated pesticides (e.g. total DDT), polychlorinated biphenyls, polycyclic aromatic hydrocarbons, and some metals (e.g. Cu, Pb, Hg) were found to exceed threshold-effects level (TEL) or effects range-low (ERL) values in sediments of Ostrich Bay and the mouth of the bay (Carr and Nipper, 1999), although exceedances of the Washington state AET-based sediment criteria were not observed. However, Toxicity Identification Evaluations (TIE) of whole sediment and sediment porewater from the most toxic stations of Ostrich Bay were unable to identify a chemical group responsible for the observed toxicity (Carr et al., 2001). None of the detected chemicals, or suite of volatile and semi-volatile compounds, butyl tins, ordnance compounds, or mercury were found to be contributors to sediment toxicity or at levels exceeding toxicity thresholds in spiked sediment samples. The final TIE studies concluded that toxicity of Ostrich Bay sediment appeared to be related to an unmeasured organic or organo-metallic compound (Carr and Nipper, 2000).

Because mercury concentrations consistently exceeded sediment quality criteria throughout the Dyes Inlet system, and mercury apparently has been continually transporting into Ostrich Bay, temporal trends in concentrations were evaluated. Multi-station long-term monitoring data for mercury are unavailable, but a comparison of mercury data collected from Ostrich Bay in 1997 with data collected in 1994 shows a significant 2-fold greater concentration (Fig. 4). More telling, at a single mid-basin station in Ostrich Bay (OB 326) that was matched by location with stations from the reconnaissance survey of 1988, the 1994 remedial investigation, and the sediment core collection in 1996, surficial sediment mercury concentrations appear to steadily increase between 1988 and 1997 (Fig. 4).

Although Ostrich Bay sediment mercury concentrations appear to have increased since 1988, a similar trend is not apparent in Dyes Inlet nor in the reference area of Carr Inlet. Dyes Inlet was not sampled during 1994, but comparison of data from the three 1988 stations that were matched by location with 1997 stations suggests that mercury concentrations in fine-grained sediment of Dyes Inlet remained unchanged between 1988 and 1997 (Fig. 4). Similarly, mercury concentrations at the reference area sediment stations also remained unchanged from 1994 to 1997, although the grain sizes of the 1997 samples, at an average 20% fines, were coarser than 1994 sediments (>50% fines), which could mask changes in fine-grained sediment mercury concentrations.

The temporal trend analysis suggests that mercury-laden fine-grained material may continue to deposit in Ostrich Bay. As indicated above, the fine-grained deposits appear to originate largely from outside of Ostrich Bay and Dyes Inlet. The ultimate sources of the impacted sedimentary material and the mercury found in surficial deposits of Ostrich Bay are unknown. Searches for potential sources are ongoing, and include operating and abandoned outfall discharges to Port Washington Narrows, and sediments of nearby embayments, such as Sinclair Inlet, Eagle Harbor and Elliott Bay (see Fig. 1), that are contaminated with mercury at concentrations greater than those found in Dyes Inlet (Battelle, 1989; Malins et al., 1984; Lefkovitz et al., 1997). However, transport mechanisms for sediments from these embayments have not been fully evaluated in the context of contributing sources to nearby depositional environments.

3.4. Re-evaluation of sediment remediation

Results of the studies described herein led to a re-evaluation of remediation approaches for Ostrich Bay sediments. Additional sedimentation modeling was performed to determine the influence of depositing sediment on the effectiveness of natural recovery and

sediment covers to achieve the state cleanup level within 10 years. Modeling was based on the measured sedimentation rate of 0.8 cm/yr, and assumed that bioturbation mixed the top 10-cm layer, that background depositing sediment was at the level of toxicity measured with southern Dyes Inlet sediment (Table 2), and that the reference area sediment toxicity was 10%. The model further assumed that the bioassays respond proportionally to diluting effects of mixing and that the responses are in the linear range of a dose-response curve (i.e. two standard deviations around the 50% response).

Results of the natural recovery modeling using the 1997 toxicity data showed that if background sedimentation were at reference area sediment toxicity levels, then burial and mixing under natural sedimentation would result in Ostrich Bay sediment achieving regulatory cleanup criteria at most stations of the bay (EA, 1998). However, with the toxicity of background sedimentation exceeding the state cleanup level, as determined from the surrogate source sediment in southern Dyes Inlet, natural recovery is rendered ineffective as a remediation option. Bioturbation mixing of a 10-cm cover of dredged material that tests at the reference area toxicity level with underlying 10-cm of sediment under current conditions would result in the cover material eventually exceeding the cleanup criteria within 10 years as impacted sedimentary material mixed with the cover. Similarly, a sediment cap of 1.5–3 feet of dredged material placed over the most toxic sediment of Ostrich Bay (which tested at 66.7% total effective mortality in the echinoderm larvae bioassay) would eventually mix with the depositing impacted sedimentary material and exceed the state cleanup level within 40 years.

The conclusion of the reevaluation was that none of the remedial alternatives would meet the federal or state criteria of permanence or long-term effectiveness under present depositional conditions. Following this realization, alternative plans for managing contaminated Ostrich Bay sediment were put into place. Recommendations have been made to regulatory agencies that the current impacts to Ostrich Bay sediment are not due solely to releases of ordnance compounds or other chemicals from the onshore Navy activities, but are greatly influenced by incoming impacted sedimentary material. However, the source of the toxicity in Ostrich Bay sediment remains elusive. Bioassay and chemical monitoring should be performed on sediment from southern Dyes Inlet as surrogate for incoming depositing material, or on suspended particulates in incoming tides, to determine when the toxicity of incoming deposits has decreased. Once that determination has been made, natural recovery and depositional modeling can identify likely scenarios for effective remediation. At our present stage of understanding the sediment dynamics of the Dyes Inlet system, any decision on management of contaminated sediment in Ostrich Bay

must consider the quality of incoming sedimentary material.

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