PROCEEDINGS

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THE FIRST ANNUAL MEETING ON PUGET SOUND RESEARCH

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THE NEARSHORE ZONE

IS IT A GOOD INDICATOR OF THE SOUND'S HEALTH AND PRODUCTIVITY?

Ronald M. Thom,
University of Washington
Session Chair

Introductory Statement

Ronald M. Thom*

The nearshore zone of Puget Sound may be defined as the coastline which includes the intertidal zone, and upland and subtidal habitats immediately above and below the intertidal zone. Habitats within this narrow elevation range (i.e., +20 ft to -30 ft MLLW) function together in support of biological resources, and chemical and physical processes. The coastline of Puget Sound is more than 1,300 miles (2,100 km) long, and is comprised of river mouths, inlets and small and large bays. Many bays and estuaries are shallow and it is in these shallow, relatively protected habitats that aquaculture activities, particularly of oysters, are intense and fish, crabs, shellfish and birds abound. Habitats such as eelgrass meadows, marshes, kelp and seaweed beds support these living resources by providing food and refuge for juvenile and adult populations. Human use of nearshore habitats, and the resources they contain, is high in Puget Sound. The beaches are probably the area of most frequent and direct contact by the public with Puget Sound. Many sources of pollution and disturbance are located in the nearshore zone. For example, streams and rivers carrying nutrients and toxicants from upland areas empty into this zone. Sewage outfalls are located immediately offshore of many beaches. Marinas, with their associated pollutant sources, are always located in the nearshore zone. Because of the currents that eddy around points in the diverse coastline, water-borne pollutants may be trapped in poorly flushed embayments. The intertidal zone can form an area of concentration for floating pollutants, and is often referred to as the "bath tub ring" of the oceans. Even in view of these facts, the nearshore zone of Puget Sound has received relatively little scientific study. The purpose of this session, entitled *The Nearshore Zone: Is it a Good Indicator of the Sound's Health and Productivity*, is to focus the attention of regulatory and research communities on this productive, sensitive and little studied region of Puget Sound. The papers identify the scraps of understanding regarding potential and real pollution effects in this zone that have been gained from site-specific studies. The authors outline some steps managers can take to better understand the causes of these effects.

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Eutrophication and Oxygen Depletion in Budd Inlet

Charles D. Boatman*

Introduction

Fish kills and water quality violations associated with low dissolved oxygen levels in Budd Inlet prompted Ecology to initiate an investigation into the cause of the oxygen depletion. Algal blooms in the late summer and early fall, and their subsequent decline and decay, had been implicated as the major cause of the oxygen depletion in the inner Inlet. The presence of a secondary wastewater treatment discharge in the inner Inlet was known to be the major point source of nutrients in the Inlet. Therefore the major research question was what was the contribution of the wastewater discharge towards algal blooms in the inner Inlet, and how were the blooms linked to the low dissolved oxygen levels. In addition, if the blooms were associated with the low dissolved oxygen problem, and were linked to the discharge, to identify what measures may be taken to resolve the problem.

A laterally averaged two-dimensional computer model was developed to simulate the hydrodynamics, transport, and ecosystem dynamics of Budd Inlet. This included tidal mixing, inputs from point and non-point sources throughout the Inlet which varied over time, winds, sediment fluxes of oxygen and nutrients, and the algal/nutrient/D.O. ecosystem dynamics. Outfall alignments and discharge scenarios were run for the spring diatom bloom in May and the fall dynoflagellate bloom in September when low dissolved oxygen conditions were observed.

This papaer discusses results from model runs for a May diatom bloom and a late summer dinoflagellate bloom. Results from the September runs suggest that high respiration rates, coupled with daily vertical migration of the dynoflagellates, were the probable cause of the low dissolved oxygen in the bottom waters. To minimize the potential magnitude of the oxygen depletion it was recommended that nitrogen removal of at least 90 percent be implemented for effluent discharged into the Inlet.

Model Description

In choosing a dynamic model, it was necessary that a number of key factors could be accommodated by the model for application to Budd Inlet. At a minimum, the model must account for the

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mixing/dispersion dynamics such as river flow, density currents and wind driven currents which control the advective processes. It must also include the wind and tidal mixing which control the diffusive processes and be able to account for a constantly changing free surface due to the large tidal range within the Inlet. The water quality section of the model must describe the dynamics of the nutrient/algae/D.O. system and be able to include loadings of BOD, D.O. and nutrients which may occur anywhere within the Inlet and which may vary with time. It must allow for sources and sinks at the surface and the bottom. Overall, the model should not be too cumbersome or too costly so that it could be efficiently applied.

To satisfy these requirements, a laterally averaged two-dimensional finite-difference model was adapted for use in this study (for a more detailed model description see Boatman et al., 1986). This model included all of the key factors and was preferable over vertically averaged two-dimensional models which would not give the vertically stratified estuarine flow.

Biogeochemical processes are simulated within the water quality portion of the model using internal source/sink and reaction rate terms for each of the water quality constituents. Eight water quality constituents were used in the water quality module. They are: organic nitrogen (Org-N), ammonium (NH_4-N), nitrite (NO_2-N), nitrate (NO_3-N), Algae (Chl a), pheopigments, dissolved oxygen, and biochemical oxygen demand (BOD5). Three external source/sink terms: sediment oxygen demand (SOD), fecal pellets and atmospheric gas exchange were included. Each constituent has reaction rates, settling velocities, and interactions with other constituents, which are incorporated into the source/sink and reaction rate term. The major interactions are diagramed in Fig. 1.

Since oxygen utilization from the decay of algal blooms was an important issue, the SOD was included as an integral part of the dynamic model. Both SOD and benthic ammonium release were included as a real-time dynamic parts of the model using the settling rates of oxygen demanding organic material as a major input. The components used to represent the flux of oxygen demanding material were: algae, pheopigments, fecal pellets and BOD5. The first three components are a direct result of algal production. The fourth component, BOD5, was considered a separate source of oxygen demand and was treated as an external source in the model.

Model Results

For the purpose of discussion, the Inlet is divided into three longitudinal sections (see Fig. 2. for landmarks). The inner Inlet is defined as being south of Priest Point, with the central Inlet lying north of Priest Point and south of Gull Harbor. The outer Inlet is defined as the area north of Gull Harbor to the mouth of the Inlet which lies between Dover Point and Cooper Point.

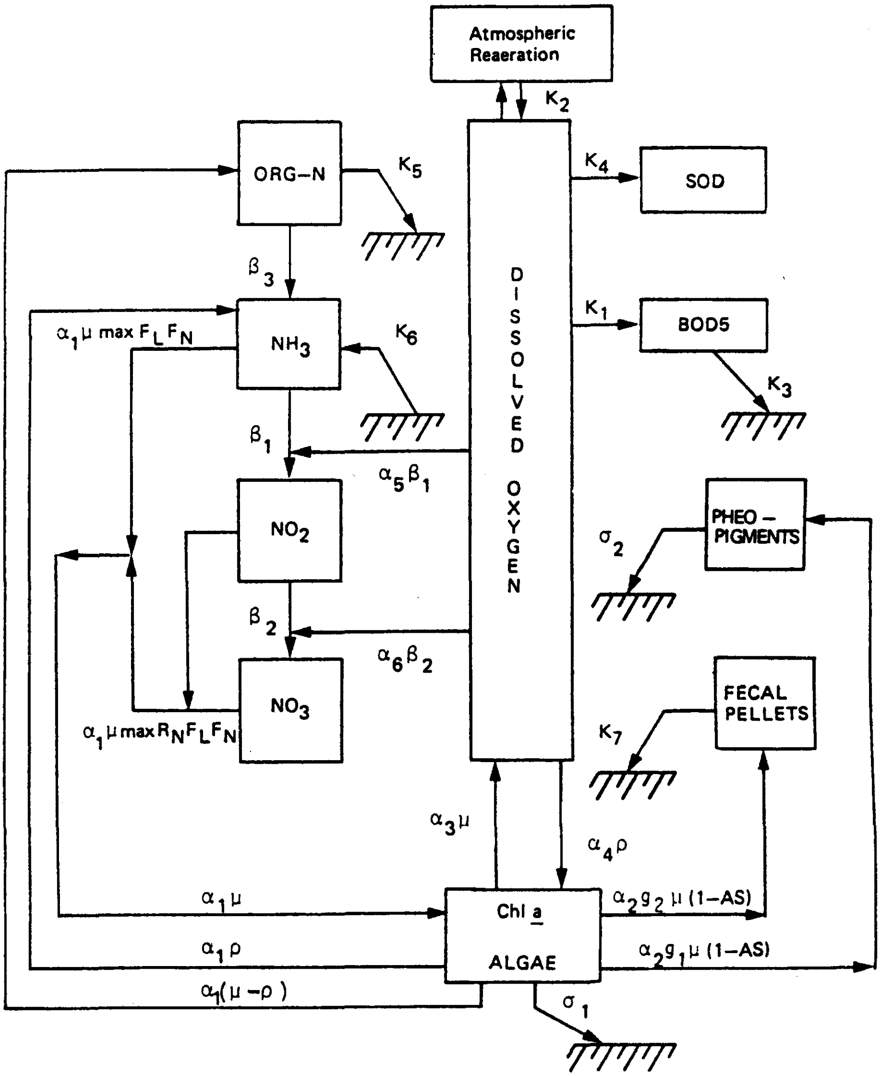

Fig. 1. Ecosystem/water quality module structure showing rate constants and coefficients used in the transfer of constituents. (see Boatman et al., 1985 for details).

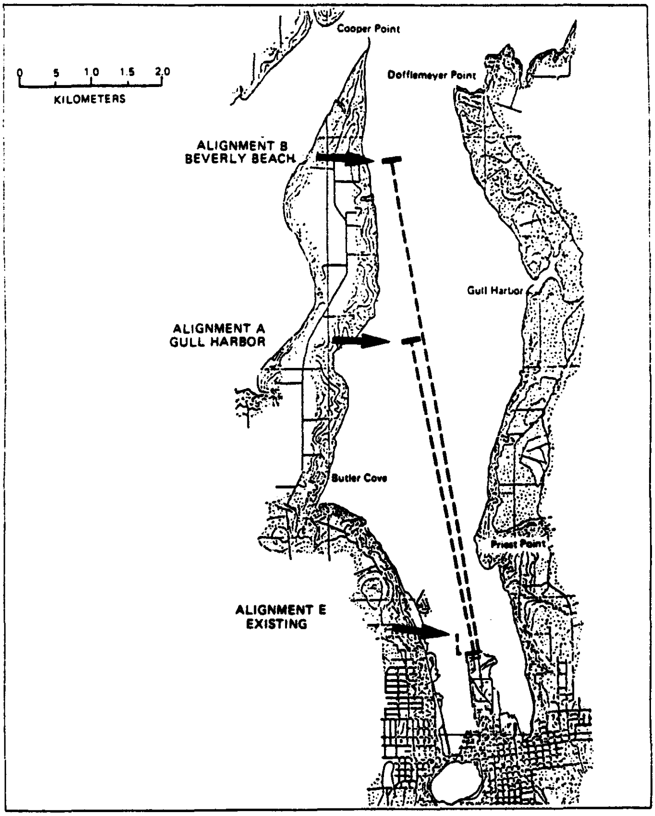

Fig. 2. Budd Inlet showing the modeled alternative outfall alignments E, A, and B.

May scenarios

Three outfall alignments, four flow rates and two alternative nutrient removal scenarios for the secondary wastewater discharge were used as input to the dynamic model. The three outfall alignments are shown in Fig. 2 and are labeled as Alignment B in the outer Inlet, Alignment A in the central Inlet, and Alignment E, which is the existing alignment, in the inner Inlet. Seven of the modeled scenarios are listed in the following table.

Scenario	Alignment	Modeled Flow rate	Nutrient Removal
1		No Discharge	
2	E	24.0 mgd	None
3	E	24.0 mgd	90% Removal
4	E	16.0 mgd	None
5	E	16.0 mgd	90% Removal
6	A	24.0 mgd	None
7	B	24.0 mgd	None

The 24.0 million gallons per day (mgd) flow rate corresponds to the estimated average dry weather flow (ADWF) rates for the year 2010. The 16.0 mgd flow rate approximates the presently permitted average wet weather flow (AWWF) rate of 16.3 mgd.

Comparisons of the results from the no discharge scenario 1 with scenario 2 at alignment E are shown in Fig. 3. This data indicates that the strength of the bloom is enhanced by the wastewater discharge. The sediment oxygen demand is also enhanced since SOD is related to algal production. However, the near-bottom dissolved oxygen actually increases during and after the bloom. This is due to rapid mixing and circulation in the Inlet which mixes the surface water down into the landward flowing deeper waters as seen in the simulated dispersion of a conservative tracer shown in Fig. 4. These results suggest that the photosynthetically produced oxygen in the surface water was being mixed into the bottom water more quickly than it could be consumed.

The strength of the bloom in Budd Inlet is not only controlled by the amount of discharge, but also by the location of the discharge in the Inlet. This is shown in Fig. 5. which is a summary of the relative enhancement of the May bloom in the inner, central and outer Inlet for each of the scenarios relative to the no discharge scenario. Fig. 5. shows that the strength of the bloom for the central Inlet (A) and the outer Inlet (B) alignments are somewhat less than for the inner Inlet alignment E. However, all of the alignments show an enhanced bloom relative to the no discharge scenario.

Fig. 5. clearly shows that next to complete discharge elimination, nutrient removal is the next best option in order to substantially reduce spring algal blooms. All scenarios without nutrient removal, regardless of placement within the Inlet, show a 30 to 50 percent increase of the strength of the algal bloom relative to the no discharge scenario for the inner Inlet.

<u>September scenario</u>

The oxygen and nutrient profiles from the September survey, which was the critical low dissolved oxygen period, were the most difficult to reproduce even with unrealistically high phytoplankton growth rates. The main difficulty was in reproducing the measured high levels of chlorophyll a and oxygen in the surface waters along with low levels of nitrate throughout the water column and low dissolved oxygen levels in the bottom waters of the inner Inlet.

At first it was thought that the low dissolved oxygen measurements in the bottom water were due to higher SOD rates. However, calibration runs designed to maximize benthic oxygen demand with high growth, settling and grazing rates could not produce bottom oxygen values lower than about 5 mg/L. The vigorous vertical mixing would not allow such large vertical oxygen gradients to be maintained.

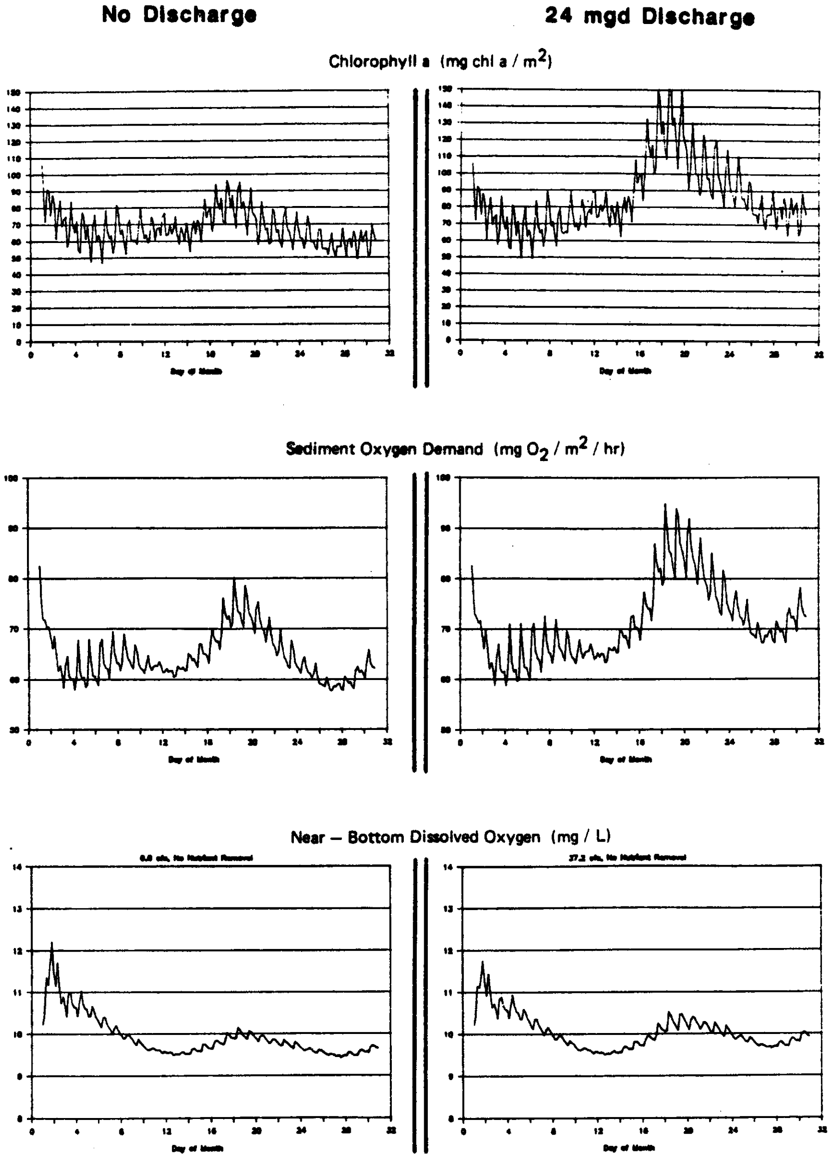

Fig. 3. Comparison of predicted phytoplankton production, SOD, and near-bottom oxygen concentration for the no-discharge scenario 1 and scenario 3 for the Inner Inlet.

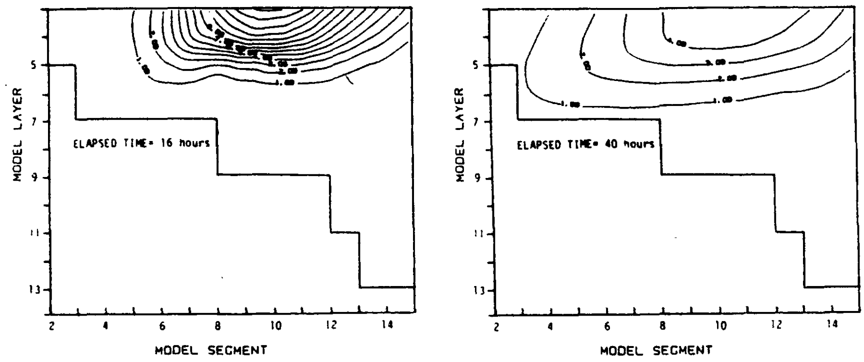

Fig. 4. Simulated Dispersion of a Conservative Tracer at 16 and 40 Hours after release in Budd Inlet.

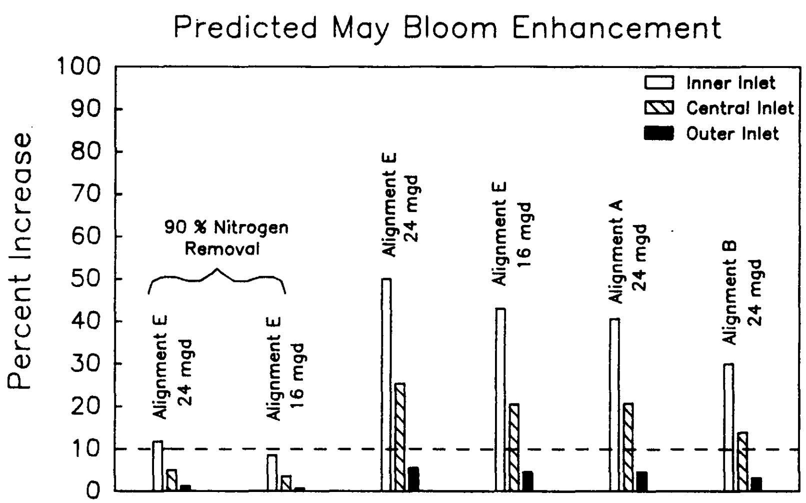

Fig. 5. Predicted bloom enhancement for May 1985 for seven discharge scenarios as a percent increase over the no-discharge scenario.

The final calibration of the dynamic model for September involved a revision of the water quality module to include algorithms which would simulate the phototaxic vertical migration and physiology of dinoflagellates, which were the dominant phytoplankton found during the September survey. Because they are active swimmers, the dinoflagellates have a much higher respiration rate than diatoms. It was believed that the active vertical migration of the dinoflagellates was the key process involved in maintaining the large vertical gradients of dissolved oxygen. In effect the dinoflagellates were acting as a biological "pump" taking oxygen out of the bottom water at night and producing it in the surface water during the day. As long as the vertical migration could be maintained and the rate of respiration in the bottom waters was greater than what could be replaced by vertical mixing, then large vertical gradients of oxygen could be maintained.

This revised water quality module gave qualitative results which were very encouraging. Surface water dissolved oxygen levels of between 15 and 20 mg/L and bottom water values of less than 3 mg/L could be maintained. Nitrate values were less than 2 ug/L throughout the water column in the inner inlet along with high chlorophyll a concentrations in the surface waters. These results could not be obtained without the simulated vertical migration and the physiological adaptations of the dinoflagellates. Fig. 6. compares water quality profiles from model runs for September conditions which show the contrasting results from dinoflagellate and diatom simulations using identical initial and boundry conditions.

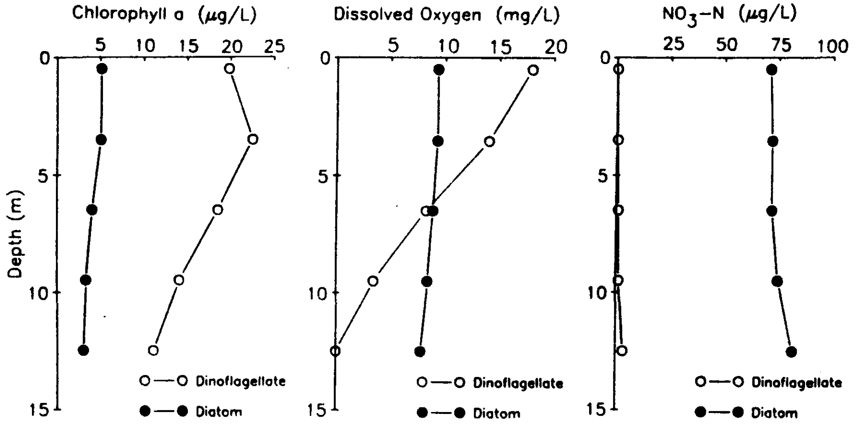

Fig. 6. Comparison of water quality profiles from dinoflagellate and diatom simulations for September 1984.

Since the occurrence of dinoflagellate blooms is part of the normal temporal succession of phytoplankton, the presence of dinoflagellate blooms in Budd Inlet is not as important to the low dissolved oxygen problem as are the strength and duration of the blooms. Based on our experience with the spring bloom in May, which showed 30 to 50 percent increase in the strength of algal bloom due to nutrient addition, it was inferred that a similar result would hold true for the dinoflagellate blooms.

The magnitude of the nutrient enhancement of the dinoflagellate bloom could in fact be greater than what was observed for the diatoms in May. Since the dinoflagellates can grow at low light levels and can migrate to reach their optimum light level, they would rarely be light limited. In addition, their ability to take up nutrients day and night implies that the magnitude of the bloom would be dependent on the supply of nutrients rather than the availabliity of light. This was not the case during the May diatom bloom when the duration of the bloom was controlled by the number of consecutive sunny days. This scenario is supported by a recent study of recurrent dinoflagellate blooms observed during the fall in a coastal lagoon on Vancouver Island (Robinson and Brown, 1983). The authors concluded that the ultimate magnitude of each years bloom was regulated by the availability of nitrate supplied in runoff.

Seasonally, dinoflagellate blooms in the central and inner Inlet may be encouraged by the lower flushing rates in the late summer and early fall, combined with the tendency of the bottom waters to accumulate in the inner Inlet due to the landward flowing estuarine circulation. Therefore there is a strong potential for a sustained dinoflagellate bloom enhanced by nutrient addition in Budd Inlet. The longer the bloom is sustained and the stronger it is, the larger the potential for oxygen depletion in the bottom waters, especially in the inner inlet. This agrees with a previously reported low D.O. period for August of 1977 in Budd Inlet which coincided with an extensive dinoflagellate bloom throughout the Inlet (Kruger, 1979).

The bottom water oxygen depletion created by dinoflagellate blooms in the late summer and fall is aggravated by fact that the water from Puget Sound which flushes the Inlet is on the average 2 mg/L lower in dissolved oxygen than in the Spring. This lower D.O. water is brought into Puget Sound by coastal upwelling through the Straits of Juan de Fuca during the summer months when northerly winds prevail off the coast. This water migrates south through Puget Sound and reaches Southern Puget Sound during the late summer. In addition, the bottom water temperature is also higher by three to four degrees in the late summer and fall which raises the SOD rates from 30 to 40 percent, enhancing oxygen depletion of the bottom waters.

It is very probable that the duration of the dinoflagellate blooms in Budd Inlet are controlled by a combination of meterological and hydrodynamic conditions which disrupt the vertical migration of the dinoflagellates. This is what was observed in the September, 1984 survey when strong vertical mixing and advection occurred due to heavy winds. This was also observed by others (Westley et al., 1973) in a September 1972 survey.

Altogether, the results indicate that even though conditions exist in the late summer and fall which contribute to lower dissolved oxygen values in the bottom water, these conditions alone cannot account for the observed values. The evidence strongly suggests that a reduction in nutrient addition will result in a reduction in the magnitude of the dinoflagellate bloom, and consequently a reduction in the magnitude of the oxygen depletion.

Conclusions and Recommendations

The probable cause of the low dissolved oxygen conditions in the late summer and early fall in Budd Inlet is the presence and persistence of dinoflagellate blooms. When bloom conditions prevail, there is strong evidence which suggests that the magnitude of the blooms are dependent on the supply of nitrogenous nutrients. Although we do not, at this time, have a quantitative dinoflagellate model for Budd Inlet, we can infer from our qualitative modeling results and our diatom model for May that the magnitude of the dinoflagellate bloom in the inner inlet could be enhanced from 30 to 50 percent by discharge configurations without nutrient removal regardless of outfall placement in the inlet. State water quality standards allow natural dissolved oxygen levels to be degraded by up to only 0.2 mg/L by man-caused activities, regardless of the water classification. Therefore nutrient removal should be implemented as soon as practicable in order to attempt to meet the State water quality standards.

Under the recommendations of state agencies, an algal bloom enhancement of 10 percent or less as the result of nutrient addition from the wastewater treatment plant was considered acceptable. The following recommendations would achieve this acceptable level and would minimize the potential magnitude of oxygen depletion in the late summer and early fall due to dinoflagellate blooms in Budd Inlet.

- For the present outfall location, maintain the permitted flow rate of 16.3 mgd (AWWF) and establish nutrient removal of at least 90 percent using best available technology. This could be accomplished on a seasonal basis from April through October.

- For any outfall location within the Inlet or any increase in permitted flow up to 22 mgd (AWWF), establish nutrient removal of at least 90 percent using best available technology. This could be accomplished on a seasonal basis from April through October.

Acknowledgements

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Nearshore Primary Productivity in Central Puget Sound: A Case for Nutrient Limitation in the Nearshore Systems of Puget Sound[1]

Ronald M. Thom,[2] Andrea E. Copping[3] and Richard G. Albright[4]

Introduction

Puget Sound is a fjordal estuary that contains a complex system of shallow embayments and deep reaches (max. depth ca. 280 m) bounded by shallow sills. Puget Sound receives nutrients from rivers, streams, sewage discharges and the ocean, and productivity in Puget Sound is generally regarded as being moderate to high relative to other estuarine systems (Strickland, 1983). It has been shown that, except for very short periods of time, inorganic nutrients are not limiting phytoplankton production in Puget Sound (Campbell et al., 1977). However, recent evidence indicates that excess nutrient inputs from agricultural and sewage discharge are resulting in phytoplankton blooms and eutrophic conditions in embayments in southern Puget Sound (URS, 1986).

Eutrophication has not been documented for central and northern Puget Sound, but odorous conditions on some beaches in these regions have recently (i.e., since 1980) been reported (Thom et al., 1984; Thom, 1985). In many of these cases, odors have been associated with piles of decaying seaweed.

Here we evaluate nutrient limitation and the potential for eutrophication in the nearshore zone in central Puget

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Sound. The nearshore zone is defined as the region from higher high water down to the lower depth limit of benthic vegetation (ca. -10 m MLLW). Benthic primary production is high and faunal resource use is extensive in this zone (Thom, 1987). Nutrient limitation is evaluated in terms of temporal dynamics of a potentially limiting nutrient (i.e., nitrate), nitrogen to phosphorous ratios (Ryther and Dunstan, 1971), and the dynamics of nearshore plant production and biomass. The findings are summarized in a conceptual model of the limiting or enriched conditions.

The most well studied cases of nearshore estuarine or marine eutrophication in the United States are from Nahant Bay (Quinlan, 1982) and Boston Harbor (Cotton, 1910), both located in Massachusetts. These areas have a long history (>80 years) of fouling by mats of macroalgae and intense odor. More recently, eutrophication in Kaneohe Bay, Hawaii has received considerable study (Smith et al., 1981). The decline in submerged aquatic vegetation in Chesapeake Bay is partially due to eutrophication (Orth and Moore, 1985). Lee and Olsen (1985) summarized the eutrophication problem in large coastal lagoons in Rhode Island, and developed management strategies to deal with the problem.

Study Sites

This work was largely carried out during the Seahurst Baseline studies sponsored by the Municipality of Metropolitan Seattle. Four beaches, located in the East Passage of central Puget Sound, were sampled for inorganic nutrients and benthic vegetation. Three of the beaches, i.e., Normandy, Seahurst and Tramp Harbor (Vashon Island), had dense accumulations of drift seaweed in summer months. The fourth beach, Aquarium (Vashon Island) did not have extensive drift seaweed accumulations. Fauntleroy Cove, an embayment showing recent evidence of eutrophication, was sampled intensively in the summer of 1985. The locations and descriptions of the beaches are given in Thom et al. (1984) and Thom (1985).

Methods

Nutrient and salinity samples were collected approximately 20 cm below the surface in hip deep water by hand. Inorganic nutrient samples (i.e., nitrate, nitrite, ammonia, phosphate) were stored frozen in the dark until analyzed. Salinity samples were held in citrate bottles. Duplicate water samples were taken biweekly, mid-morning during neap tide at each of the four Baseline study beaches in June 1982-March 1984. A total of 54 water samples were collected using the same methods at Fauntleroy Cove in July-August 1985. Samples were collected at low tide from streams located at Normandy Beach and Fauntleroy Cove where the streams entered onto the intertidal zone. The methodology for water analyses is given in Thom et al. (1984).

Standing stock and productivity of the benthic macrophytes (i.e., seaweeds and eelgrass) and sediment associated microalgae were sampled at 30-60 day intervals in June 1982-March 1984 at the Baseline study beaches. Phytoplankton standing stock was sampled daily during the same period at Seahurst beach only. Productivity of phytoplankton was measured at a station ca. 100m off Seahurst beach every 7-21 days from June 1982-December 1983. Data from 1983 only, the year with the most comprehensive data set, is presented below. Details of sampling methodology are given in Thom et al. (1984) and Anderson et al. (1984).

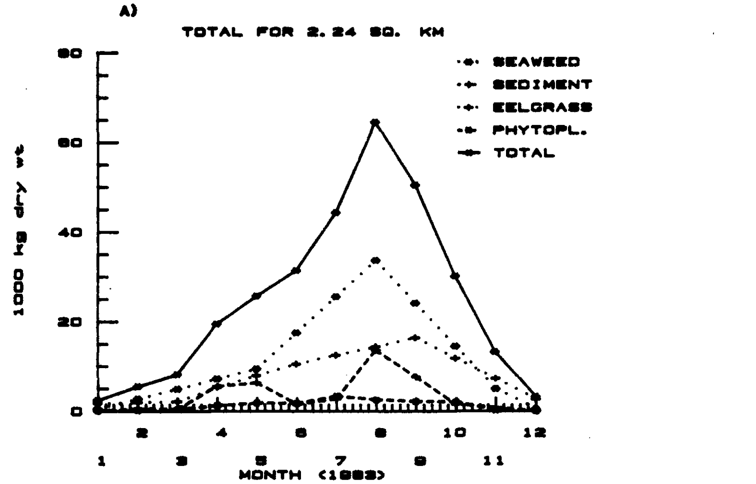
Results

Vegetation dynamics

Seaweed had the greatest standing stock among all autotrophs during the summer maximum, followed by eelgrass, sediment associated algae and phytoplankton (Fig. 1A). Net primary productivity for all nearshore autotrophs combined at Seahurst beach was greatest in spring (Fig. 1B), which was followed by a build up of biomass in summer (Fig. 1A). Mean monthly nitrate concentration at Seahurst beach showed a steady decline in spring from high winter concentrations during the period of maximum productivity, and a build-up during the period of declining productivity (autumn-winter) (Fig. 1B).

Nutrient dynamics

Nitrate concentrations varied with season at the four Baseline study beaches (Fig. 2). During late-spring through summer (May-September), nitrate was generally below 10 µg at/l, and was at times undetectable. Nitrate



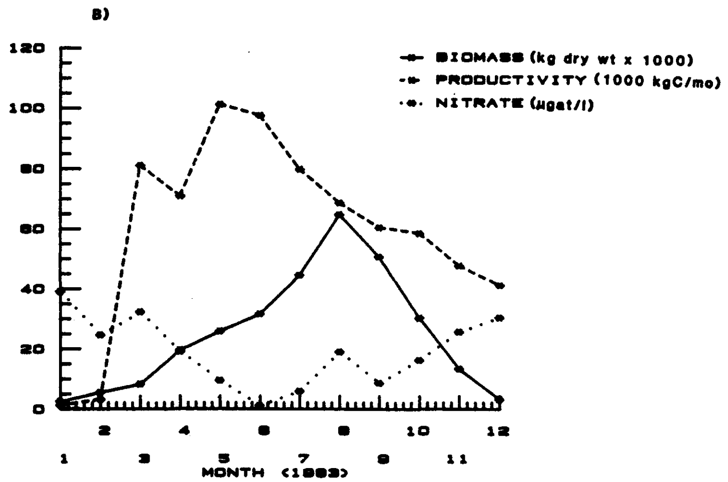


Figure 1. (A) Total standing stock of autotrophic components of the nearshore system in Seahurst bight; (B) total autotrophic biomass, net primary productivity and mean monthly nitrate concentration in the nearshore system in Seahurst bight.

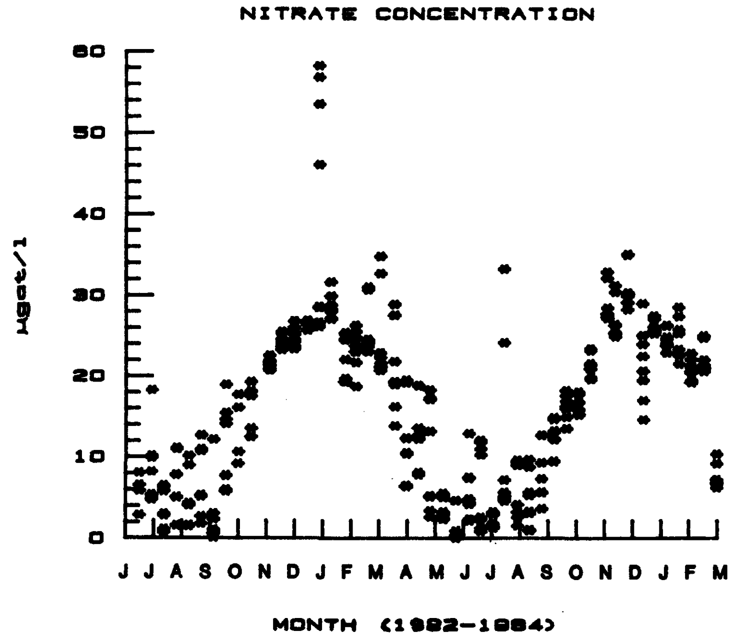


Figure 2. Nitrate concentrations in water samples from the East Passage of Puget Sound.

concentrations were highest and least variable in winter. Total inorganic nitrogen to phosphate ratio (N:P) followed a similar cycle (Fig. 3). The N:P ratio illustrated that nitrogen was being preferentially taken up during spring and summer. N:P was below the Redfield ratio (Redfield et al., 1963) of 16 in approximately 97% of the samples, which is a considered to be the threshold of nutrient limitation in algae. Furthermore, 69% of the values were below a more conservative ratio of 10 put forth by Ryther and Dunstan (1971) for marine coastal systems.

If oceanic water entering Puget Sound were the sole source of nitrate in ambient waters, a positive correlation would be seen between salinity and nitrate. The data from our

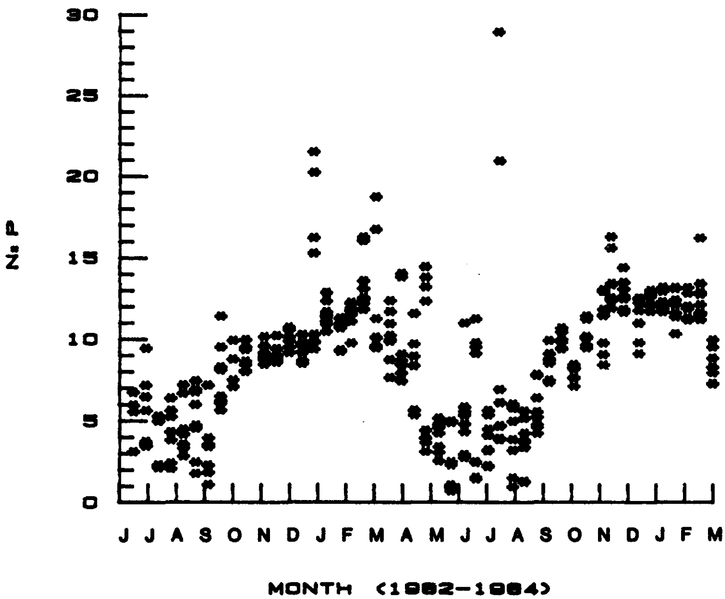


Figure 3. Total inorganic nitrogen to phosphate ratio (N:P) in water samples from four beaches in the East Passage of Puget Sound.

study sites show a seasonal relationship between nitrate and salinity, with some high nitrate levels associated with low salinity water during times of high freshwater input (winter and spring) (Fig. 4). Nitrate levels are highly variable during summer and fall when most input to the Sound is via the Strait of Juan de Fuca. Samples from the two small streams indicated that nitrate can be high in small streams relative to ambient concentrations in Puget Sound (Table 1).

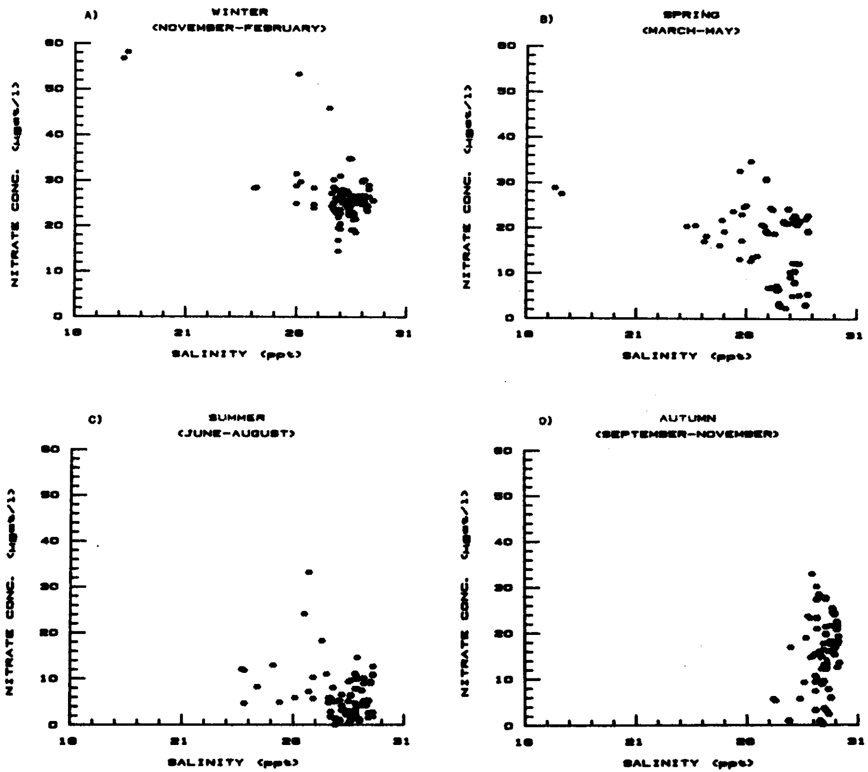


Figure 4. Salinity versus nitrate concentration in water from four beaches in the East Passage of Puget Sound collected in (A) winter, (B) spring, (C) summer and (D) autumn.

Table 1. Nitrate concentrations in streams adjacent to two of the study sites in central Puget Sound.

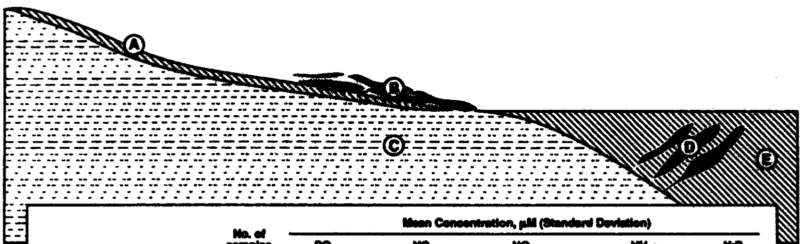
Stream Location	n	Nitrate (µg at/l) Mean	S.D.	Months
Fauntleroy Cove	6	47.6	5.1	August
Normandy Beach	4	145.5	18.1	December-January

Fauntleroy Cove model

Drift seaweed mats covered approximately 1.3 ha at Fauntleroy Cove, and reached a maximum standing stock of 1.7 kg dry wt/m^2 (Thom, 1985). More than 28 mT (wet wt) of seaweed were manually removed from the the area of greatest accumulation in a one month period in summer. Approximately 98 mT (wet wt) of seaweed loaded an ca. 1 ha area of the beach between 11 July and 4 September 1985 (Thom, 1985). Nitrate concentrations were very high near the area of greatest seaweed buildup (Fig. 5). Ammonia concentrations were exceptionally high in the seaweed mat and underlaying sediment water (Fig. 5), which indicated remineralization was occurring. Water flowing down the gradient of the beach had lower concentrations, and the lowest concentrations were recorded in samples collected directly within the actively growing green algal bed. Nitrate was in high concentration in samples from the stream, as compared to ambient water collected offshore (Fig. 5). Attached seaweed biomass in the immediate vicinity of the drift algal mat was in excess of eight times greater than background levels from other beaches in the East Passage (Thom, 1985). The attached plants at Fauntleroy were very large; one blade of the dominant green alga *Ulva fenestrata* measured 99 x 154cm in size.

Discussion

Depleted nitrate concentration during times of high net primary productivity and algal biomass coupled with N:P falling below growth limiting levels at the beaches provide strong evidence that nutrient limitation occurs under certain conditions in nearshore areas of central Puget Sound. It has been the general conclusion that nutrients are not limiting phytoplankton growth in the deep central portions of Puget Sound, which was recently



		No. of samples	Mean Concentration, µM (Standard Deviation)				
			PO_4	NO_3	NO_2	NH_4^+	N:P
A	Stream	6	2.5 (0.9)	47.6 (5.1)	0.2 (0.1)	1.9 (2.3)	21.5 (5.6)
B	Algal Mat	4	10.5 (3.3)	2.4 (2.8)	1.2 (2.0)	61.1 (29.2)	6.1 (0.6)
C	Sediment	5	23.9 (17.6)	6.7 (7.7)	2.9 (3.9)	191.5 (149.2)	10.6 (7.0)
D	Algal Bed	7	1.8 (0.3)	5.1 (4.9)	0.4 (0.2)	1.3 (0.4)	3.6 (2.2)
E	Ambient Puget Sound	9	2.0 (0.1)	15.1 (3.9)	0.6 (0.1)	1.1 (0.6)	8.5 (1.9)

FAUNTLEROY COVE INORGANIC NUTRIENT CONCENTRATIONS

Figure 5. Schematic drawing showing the nearshore system in Fauntleroy Cove and nutrient concentrations in various places in the system.

confirmed in bioassays using *Skeletonema costatum* (Welch and Messner, 1984). Nutrient limitation of phytoplankton has, however, been demonstrated for inlets in coastal British Columbia (Cochlan et al., 1986). Nutrient limitation in shallow estuaries is common, especially where benthic and water column photic zone coupling is strong (i.e., benthic processes strongly influence processes and chemistry of the total water column), and data on depressed N:P are generally cited to support this conclusion (e.g., Pilson, 1985; Lee and Olsen, 1985; Bishop et al., 1984; Quinlan, 1982). The nearshore zone in Puget Sound represents an area of relatively strong benthic-water column coupling. In areas where benthic plants are abundant, high levels of productivity during spring and summer is responsible for depleting inorganic nutrients that wash over the beaches at during flood tides.

The growth of seaweeds in Puget Sound is probably not light limited in spring and summer. Nutrients reach the seaweeds in pulses during flooding tides. Uptake of nutrients in pulses has been shown experimentally to significantly increase productivity and growth of several seaweed taxa (e.g.,Connolly and Drew, 1985; Fujita, 1985; Lapointe, 1985; Rosenberg and Ramus, 1984; Thomas and Harrison, 1985; Thomas et al., 1985). Under severe nutrient limited ambient conditions, frequency of nutrient pulses was quantitatively more important than concentration in regulating the growth of the red alga *Gracilaria tikvahiae* (Lapointe, 1985). Thomas et al. (1985) showed using a seaweed taxon (*Fucus distichus*) from the northwest that nutrient uptake rate was not saturated over a range of nitrogen concentrations much higher than is normally found in ambient waters. This feature, which appears to be common to marine macrophytes, but not phytoplankton, may be due to the thick multicellular growth form of seaweeds (Thomas et al., 1985). Non-saturated nitrogen uptake by seaweeds in dense stands under light-saturated conditions could rapidly and steadily deplete nutrients in the water that periodically covers the bed. This latter condition may explain why nitrogen concentration would be reduced relative to phosphorous, resulting in low N:P, even in cases where nitrogen input was high.

Scenario for Puget Sound

The conditions on the beach at Fauntleroy Cove indicate localized eutrophication. Our conceptual model (Fig. 5) of the mechanisms resulting in the condition follows that of Quinlan (1982). Stream water and oceanic water from Puget Sound are primary sources of nutrients to the system. An additional important source of nutrients is from remineralization in the decaying algal mat. Organic matter leaves the system via settlement to deeper areas, transport to adjacent beaches by longshore drift, and as dissolved and particulate matter in the water column. Drift seaweed biomass decreases rapidly in autumn in concordance with reduced attached algal biomass (Thom et al., 1984). Annual fluctuations in nitrogen input and variations in the dynamics of processes within the system will probably affect the annual nitrogen pool in the system. We suspect that the degree of nutrient trapping and potential for eutrophication is dependent upon the physical characteristics of the system. For an embayment with very limited flushing (i.e., Dabob Bay), Copping (1982) concluded that the system was essentially closed with regard to nitrogen cycling. Essentially, this same condition exists in Budd Inlet (URS, 1986). Under conditions where (1) organic matter is trapped in large quantities, (2) nutrient remineralization is possible, and (3) substrata is available nearby in the photic zone for seaweed attachment, the probability of eutrophication during summer is enhanced. This condition is exacerbated if nutrients from sources other than Puget Sound (e.g.,streams, sewage discharges) are available to the attached seaweeds.

Because of refluxing and cycling processes (Ebbesmeyer and Barnes, 1980), anthropogenic input of nutrients may be resulting in a net increase in the nutrient pool of Puget Sound. Collias and Lincoln (1977) calculated, based on available historical data, that the phosphate concentration in the main basin of Puget Sound increased 10% between data taken in 1932-1963 and data taken in 1974-1975. Although they state that this increase is within the annual fluctuations in phosphate, their data were from offshore sampling stations where we would expect a smaller increase relative to nearshore areas. Nitrogen data were not extensive enough to permit a comparative analysis. Collias and Lincoln calculated that approximately 1.9 (23 mT) and 2.5% (225 mT) of phosphate and nitrate input, respectively, to Puget Sound, remained within the system. Ebbesmeyer and Helseth (1977) found that phytoplankton biomass had not increased over a similar time period in the main basin. Data from deeper portions of Puget Sound may not be reflective of what is happening in nearshore areas. Based on data from other areas (e.g.,Kelley et al., 1985, Nowicki and Nixon, 1985, Boynton and Kemp, 1985), nutrient regeneration on beaches and release into the water column is a predominant and widespread process in shallow estuaries and coastal embayments. This process is probably important in embayments of Puget Sound. If nutrients are increasing in Puget Sound, it is likely that some will be trapped and increase in embayment subsystems. It is in these subsystems, where benthic-water column coupling is strongest, that the effects of nutrient build-up would predictably be detected first.

Eutrophication on beaches is important due to the likelihood of effects on human health and biological resources. We conclude that eutrophication is not a serious problem in Puget Sound except in a few local circumstances. However, a measurable mass of nutrients remain in the Sound, most freshwater nutrient sources are located close to shore, nutrient limitation may be a common feature in shallow areas, eutrophic conditions have been documented, and nutrient input will predictably increase in the near future as a result of increased population. These facts justify the need for concern. We recommend studies on the freshwater nutrient contributions to Puget Sound, nutrient requirements of benthic primary producers, dynamics of nutrient cycling, and the degree of nutrient trapping in embayments to further evaluate the model and scenario presented for Puget Sound.

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PSP Research: Implications for Puget Sound

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Introduction

The use of bivalve shellfish resources in Puget Sound is restricted by a natural phenomenon, the production of toxins by the dinoflagellate *Gonyaulax catenella* (*Protogonyaulax catenella*). Shellfish ingest this species and accumulate the toxins. When humans consume shellfish containing sufficient toxin, paralytic shellfish poisoning (PSP) can result. Recent research in eastern Canada has indicated that finfish resources in many areas may also be at risk from these toxins (White 1977, 1980). The purpose of the field and laboratory work reported in this paper was two-fold: (1) To determine conditions which control growth and accumulation of *G. catenella* and (2) to study the effect of the toxins on finfish in Puget Sound.

Methods

Frequent field sampling was conducted in Quartermaster Harbor in Vashon Island and adjacent channel waters during April through September, 1980-81 and 1983-85. Hydrographic parameters (temperature, salinity, Secchi depth, and nutrient concentrations) were measured. Field samples were processed in the laboratory to determine density and division rates of *G. catenella* and percent of *G. catenella* cells parasitized, and the toxin content of plankton tows and mussels. Methods are described by Nishitani et al. (1985).

The effect of the toxins on chum and pink salmon smolts, coho fry and Pacific herring juveniles was tested by feeding them copepods made toxic on a *G. catenella* diet. The fish were observed at intervals to determine effects on their behavior and survival. Methods are detailed by Erickson (1988).

Results

Vertical migration

Early in the field work a study was made to determine whether *G. catenella* undergoes diel (daily) vertical migration as do many related dinoflagellates. Counts of samples taken at 1m depth intervals and at frequent time intervals demonstrated that *G. catenella* migrated following this general pattern: At 0500 hours the band of greatest cell density was in the top 3m; during afternoon hours the depth of peak

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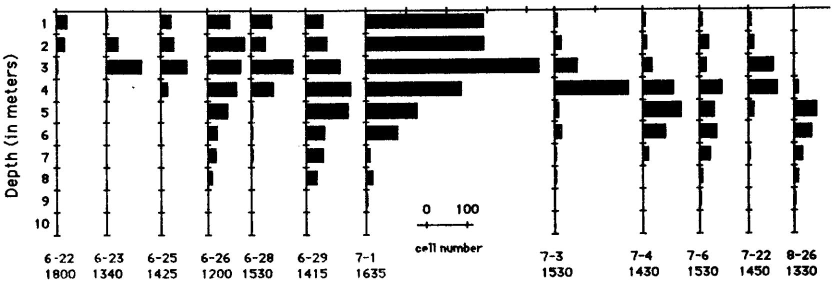

Figure 1. Vertical distribution of *G. catenella* populations in afternoon samples taken in Quartermaster Harbor, 1983. Length of horizontal bars indicates number of cells/ml.

density varied from 1 to 5 m (Figure 1); downward migration started at approximately 1800 hours and ended about midnight with the greatest density between 4 and 7m (sometimes 6-10m); migration upward started about 0100 and ended at approximately 0500. This general pattern was strongly influenced and even disrupted by very strong density gradients or wind- or tide-driven turbulence. Subsequent sampling methods were designed on the basis of this pattern. Knowledge of the pattern also provided insight into the length of time the cells were exposed to the different temperature and nutrient conditions above and below the pycnocline (Nishitani and Chew 1984 and unpublished data).

Factors influencing growth rate and density

Changes in *G. catenella* populations result from changes in the difference between the rate of growth (cell division) and the rate of removal of cells from the water column. At different times in this study the reduction of the *G. catenella* population was due to one or more of these factors: Removal of cells by tidal washout, encystment and settling out of cells, and consumption of motile cells by a variety of zooplankters, benthic invertebrates, and a dinoflagellate parasite.

Growth of *G. catenella* is slow, with fewer than 50% of cells dividing in 24 hours in field measurements, and a maximum of 1 doubling per day in laboratory cultures. Factors affecting growth rates were water temperature, which was strongly influenced by stratification, and nutrient availability. Increases in the population were noted after stratification had begun in the spring and the surface water temperature had risen to 13C. (The range of optimum water temperature for *G. catenella* growth was shown in the laboratory to be 13-18C, Norris and Chew 1975). The importance of favorable temperature was indicated in the field by the contrast between development of *G. catenella* populations at two stations, one in mixed channel water at the mouth of the bay, the other 3 miles inside the bay where relatively low tidal- and wind-driven turbulence permitted stratification and faster warming. The difference in *G. catenella* populations at the two stations was reflected in toxicity of mussels, those at the inner bay stations accumulating 80 µg toxin/100g shellfish meat several days to weeks before those at the channel station. 80 µg is the level of toxicity that has been set by the U.S. Food and Drug Administration as the standard for harvesting closure. A sparse cell density of only 10 cells/ml is sufficient to allow accumulation of 80 µg of toxins in mussels.

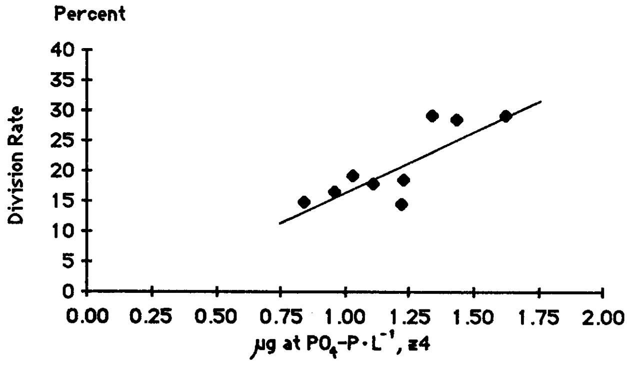

Figure 2. Division rates (% of cells dividing in 24 hours) at different PO4-P concentrations at 4m in Quartermaster Harbor, 1984. r = .816

At the inner bay station, following continued stratification and deepening of the top layer of 13C water, major increases in *G. catenella* populations ("blooms") occurred in late June of two years. During those bloom periods toxicity in mussels rose rapidly (as much as 1800 µg in a period of 5 days), to levels which could have caused death in persons consuming them. At the channel station similar, but lesser, surges in mussel toxicity occurred one to two weeks later, and in some cases followed a spring tide series accompanied by a decline of *G. catenella* density within the bay, suggesting export of the cells from the bay to the channel waters. High densities of *G. catenella* were found in channel waters when the water temperature was 13C or greater down to 9 m during later summer (Nishitani and Chew 1984 and unpublished data).

Reduction of growth rates, apparently due to low nutrient availability, was associated both with declines of blooms and with the suppression of bloom development. Low concentrations of either nitrogen ($NO_3 + NO_2 + NH_3$ - N) or phosphorus (PO_4-P) appeared to be an important causal factor in the decline of each of the three major blooms and the two mini-blooms observed during this study. In addition, during two early summer periods when temperature conditions were favorable, low nutrient availability appeared to be an important factor in precluding development of a major bloom. In some events nitrogen appeared to be the limiting nutrient; at other times, low phosphorus concentrations appeared to limit growth rates (Figure 2) (Nishitani et al. 1985 and unpublished data). These evaluations were based on N and P concentrations at 4m, a depth of long exposure time for *G. catenella*, according to migration studies.

The possibility of using the parasite, *Amoebophrya ceratii*, as a biological control agent was investigated and rejected after finding that it attacked several other species which at different times are dominant in phytoplankton assemblages. High host specificity would be an essential characteristic for a biological control agent in an estuarine environment.

Effect of toxins on finfish

Movement of the toxins to the next higher trophic level was demonstrated by the uptake of toxins by zooplankton (unpublished data). It is important to note that, during some blooms, *G. catenella* comprised approximately 95% of the phytoplankton crop available to zooplankters. The effect of the movement of the toxins to planktivorous fish was tested in the laboratory. These feeding experiments showed differences in tolerance for the toxins among the four species studied: Pacific herring and chum and pink salmon, all of which might be exposed to the toxins naturally, and coho salmon in the fresh water stage which would not be exposed to the toxins under natural conditions. The coho had a lethal reaction to very low dosages (LD50 = 39 μg STX eq/kg body weight). Pacific herring juveniles, the least tolerant of the other three species displayed disoriented behavior prior to death at relatively low dosages (LD50 = 102 μg STX eq/kg body wt). Compared with the Atlantic herring response (White 1981), the Pacific herring had similar disoriented behavior but a lower LD50. Smolts of chum salmon exhibited abnormal behavior (very passive and/or agitated) at moderate dosages (ED50=355 μg STX eq/kg body wt) but did not die at high dosages (9177 μg STX eq/kg body wt). Pink salmon showed no abnormal behavior and no mortality at the highest dosage tested (706 μg STX eq/kg body wt). Both the viscera and body tissues of chum and pink salmon and viscera only of herring were found to contain the toxins, which could be passed on to predators (Erickson 1988).

Discussion

The findings of these studies have implications for management and monitoring of Puget Sound. They also point to research needed for sound decision-making. In considering the implications for human use and management of Puget Sound, several facets of the problems caused by the occurrence of a natural poison in these waters need to be addressed: The resources at risk, the locations and level of risks, and the potential for human activities to increase the risks.

Resources at risk

The risks to human health are greatly reduced by the shellfish monitoring program conducted by the Washington State Office of Environmental Health and the Health Departments of counties with seawater shorelines. Since the monitoring program started in 1957 no PSP illnesses have resulted from consumption of shellfish commercially grown in Washington. Risks remain, however, because many recreational harvesters fail to recognize the seriousness of PSP and ignore closure warnings.

While the shellfish resources themselves have not been at risk (shellfish are only very seldom adversely affected by the toxins), the occurrence of PSP toxins in shellfish has placed severe restrictions on human use of these resources, particularly recreational harvesting. This has been especially pronounced since the mid-1970's when a pattern of more frequent occurrence of higher levels of toxicity began in Puget Sound. The 10 million dollar bivalve shellfish industry in Puget Sound, thus far, has experienced relatively minor losses from PSP harvesting closures. This has been due in large part to the fact that a high percentage of the industry is located in the Southern Basin. The possibility exists that finfish and other wildlife resources in Puget Sound could be adversely affected by the toxins. Quartermaster Harbor and other bays serve as nursery grounds for a variety of fish. The co-occurrence of dense *G. catenella* and herring or chum salmon smolts (and perhaps other susceptible species) could result in high mortality among the fish, due either to the lethal dosages received or to high levels of predation on fish receiving disabling but sublethal doses. Large kills of adult herring have occurred in eastern Canada (White 1977, 1980). Circumstantial evidence suggests PSP toxins may have been responsible for a large herring kill in British Columbia (Tester 1942). PSP toxins have been implicated in or proven to be the cause of sea birds deaths along the open coast of Washington (McKernan and Scheffer 1942) and in the Atlantic (Nisbet 1983). Humpback whale deaths off New England have been linked to PSP toxins ingested in planktivorous fish containing PSP toxins (Beach 1988). The fact that none of these effects of the toxins on finfish, birds and mammals has been documented in Puget Sound does not necessarily indicate that they are not occurring. It is possible that unexplained mass sea bird kills, periodic reductions in size of some finfish year classes or whale deaths in Puget Sound could have resulted from PSP toxins.

Location and levels of risk

Some level of toxicity in shellfish occurs each summer in all parts of Puget Sound, except for occasional years without toxicity in the Southern Basin and Hood Canal. Toxicity exceeding the closure standard has occurred in all parts of Puget Sound except the Southern Basin and Hood Canal. The possibility of future occurrences of PSP harvesting closures in those waters cannot be ruled out until we understand the factors regulating growth of *G. catenella* there.

To the extent that other bays are similar to Quartermaster Harbor and its adjacent channel waters, it is probable that bays will support earlier onset of blooms and more frequent blooms than will channels. It is also probable that low nutrient availability will plan a major role in controlling the density of *G. catenella* populations in those other bays.

The importance of warm water and stratification in bloom development indicates that the risk of unusually high levels of PSP toxins in shellfish could increase following periods of abnormally warm weather in spring or fall or periods of unusually high river runoff (which can cause deep stratification). The latter occurred in 1978 in the Whidbey Basin, which had no previous record of toxicity. At that time toxin levels in mussels rose to 30,000 μg/100g meat.

Potential effect of human activity on PSP problems

Of the major factors influencing density of *G. catenella*, human activities can effectively alter only one factor in relatively large bodies of water, namely, nutrient availability. Sources of nutrients derived from human activities would include sewage outfalls, septic tanks, marinas and agricultural runoff. Although much remains unknown about the utilization of N and P by this species, several important points about the effect of nutrient addition to Puget Sound can be made on the basis of this study.

It is unlikely that addition of nutrients by human activity will advance the timing of onset of PSP harvesting closures in early summer. Based on extrapolations from field and laboratory findings, the concentrations of N and P required to support a growth rate adequate to develop a density of 10 cells/ml (sufficient to require harvesting closure) are estimated to be extremely low and to be available in Puget Sound in early summer in all but the most extraordinary situations. However, in late summer when zooplankton predation rates are higher, a higher growth rate would be required to maintain that density of *G. catenella* cells. Under those circumstances, if nutrient depletion occurred, addition of nutrients by human activity might increase frequency and length of closure periods.

In this study, the timing of onset and the duration of dense populations (blooms) were controlled, in part, by nutrient availability. This raises the question of whether addition of N and P by human activity would increase the frequency and length of blooms. The result of bloom prolongation would be higher levels of toxicity in shellfish and other animals and, in some circumstances, greater export of *G. catenella* from bay to channel waters, resulting in increased toxicity in shellfish there.

The magnitude of the effect of adding nutrients to a water body by human activity will depend on both the quality of the added material and the characteristics of the receiving water. In order for added nutrients to stimulate growth of *G. catenella* to a density of bloom proportions (e.g., 100 cells/ml), at least the following five conditions would have to co-occur: (1) Limiting concentrations of N or P in the receiving water occurring when the temperature of the surface water layer was at least 13C. (2) The added nutrients reaching the top 10m layer. (3) An initial population of *G. catenella* existing in the receiving water. (4) The water parcel which received the added nutrients remaining intact, with minimal mixing, for sufficient time for growth to occur, several days to perhaps 3 weeks, depending on the initial density of *G. catenella* in the receiving water. (5) *G. catenella* being able to utilize the nutrients added.

The probability of these 5 conditions occurring simultaneously would depend on the individual probabilities of occurrence of the 5 conditions. This study indicated that the probability for each of the first four factors is higher in Quartermaster Harbor than in adjacent channel waters. It is expected that the same could hold true for other bays in which configurations, bathymmetry and protection from wind permit development of stratification and warm temperatures.

Research Needed

At least two questions need to be answered so that we may known the extent to which the addition of human and animal wastes can influence growth of *G. catenella.* (1) At what N:P ratio does P becoming limiting? Some of our field and laboratory work indicates that *G. catenella* may have a much higher requirement for P relative to N than is assumed for many other phytoplankters. If this is borne out by definitive laboratory experiments, this knowledge will help to clarify the potential response of *G. catenella* to the addition of different kinds of wastes to receiving waters with varying concentrations of N and P. (2) Which N- and P-compounds commonly added to Puget Sound, by point or non-point sources, can *G. catenella* utilize?

The answers to these two questions would help to provide a sound basis for decision-making regarding siting and management of activities which would add nutrients. In addition, this information might indicate whether *G. catenella* populations in the Southern Basin and Hood Canal are kept below closure levels by low nutrient availability. If this is found to be the case, the potential exists for human activities in those waters to elevate the risk of PSP problems.

Methods of Monitoring Dinoflagellates

For varying periods of time each year, in some parts of Puget Sound, dinoflagellates are the dominant phytoplankters, and thus form the major portion of the food resource of phytoplankton feeders. For this reason, a long-term study of the health of Puget Sound should monitor the populations of dinoflagellates and the physical and nutrient conditions which support their growth. Further need for such information stems from the fact that several species of dinoflagellates, in addition to *G. catenella*, are known to be noxious. *Dinophysis acuminata*, which occurs in Puget Sound, has caused diarrhetic shellfish poisoning in Europe (Kat 1985). In Puget Sound *Ceratium fusus* has caused mortalities of oysters (Cardwell et al. 1977) and pen-reared salmon and pandalid shrimp (Rensel 1976). *Gymnodinium sanguineum* (*G. splendens*) and *Prorocentrum gracile* were associated with low survival and poor development of oyster larvae (Cummins et al. 1976).

The type of routine monitoring commonly conducted, i.e., monthly at 0, 10 and 30m to measure pigment concentrations and hydrographic parameters yields data of limited value in assessing either relative abundance of dinoflagellates or the favorableness of conditions for their growth. The studies with *G. catenella* demonstrated repeatedly that a bloom can develop and disperse within two weeks, underscoring the importance of sampling more frequently than monthly. Because of the motility and the diel vertical migration of many dinoflagellate species, the distribution of their populations within the water column may vary from hour to hour. Thus, special sampling methods are required to obtain the most useful information. Either sampling at several depth intervals in the top 10m or continuous pumping through depth from 10m to the surface will provide better estimates of populations than sampling at 0 and 10m. Neither nutrient values at 0 and 10m nor their mean indicate nutrient availability at the intermediate depths in which *G. catenella* cells live through much of a 24 hour period. During this study the mean of inorganic N at 1 and 9m was comparable with that at 4m on only half of the sampling dates.

An added monitoring procedure which could give valuable information about the health of Puget Sound, would be the determination of the species composition of all phytoplankton. Persistent changes in species composition, exceeding interannual variations, could reflect changes in either climatic or hydrographic conditions. Species composition would be expected to be far more sensitive to changes in nutrient conditions than would pigment concentrations which are currently monitored to determine the total abundance of photosynthetic phytoplankton.

Recommendations

1. Research should be conducted to investigate nutrient utilization by *G. catenella*.
2. The potential effect of human activities on PSP problems should be considered in siting and management decision-making.
3. Modifications in water quality monitoring should be implemented to improve sampling of dinoflagellates and to provide species composition data.
4. Analyses of stomach contents for PSP toxins should be conducted during unexplained mass kills of fish, birds and mammals.

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DISTRIBUTION AND BIOLOGICAL EFFECTS OF SEA-SURFACE CONTAMINATION IN PUGET SOUND*

INTRODUCTION

The sea surface is a highly productive, metabolically active interface and a vital biological habitat for neuston--surface dwelling bacteria, microalgae, protozoa, copepods and larger organisms (Zaitsev, 1971). Also, numerous species of fish, including, cod, sole, flounder, hake, anchovy, mullet, flying fish, greenling, saury, rockfish, halibut, and many others have surface-dwelling egg or larval stages (Ahlstrom & Stevens, 1975; Brewer, 1981; Kendall & Clark, 1982; Zaitsev, 1971). Crab and lobster larvae concentrate in the surface film during midday as a result of positive phototaxis (Jacoby, 1982; Provenzano et al., 1983; Smyth, 1980). In Puget Sound, English sole (Parophrys vetulus) and sand sole (Psettichthys melanostictus) spawn between January and April, releasing trillions of eggs that collect on the water surface. The embryos float until hatching occurs, generally 6 to 7 days after fertilization (Budd, 1940 and personal observation).

Contaminants that have low water solubility or that associate with floatable particles concentrate in the surface microlayer (SMIC, upper 50 micrometers). Concentrations of potentially toxic PAHs, PCBs, and metals, orders of magnitude greater than U.S. Environmental Protection Agency water quality criteria standards, have been found in the SMIC of Puget Sound (Hardy et al., 1985 & 1986), Chesapeake Bay (Hardy, et al. 1987), and elsewhere (Hardy, 1982).

Despite the demonstrated importance of the sea surface as both a vital nursery area and a concentration point for contaminants, spatial distributions or temporal trends in surface contaminants, or their impacts on the reproduction of valuable marine species, remain largely unknown. Several recent studies have linked aquatic surface contamination with negative biological impacts. In Puget Sound (Hardy et al., 1988a&b), Southern California (Cross et al., 1988), and the North Sea (Kocan et al., 1988), fish eggs exposed to contaminated SMIC exhibited reduced viability.

Our studies were intended as a first step to determine, at a few sites and times: 1) the densities of neuston, including fish eggs; 2) the toxic effects of the SMIC in the laboratory and in situ; 3) the degree of association of toxicity with contaminated urban bay areas and with visible sea-surface films or slicks; 4) the magnitude of spatial and temporal variability in aquatic surface contamination; 5) the probable sources of contamination, and 6) the relationship between contaminant concentrations and the toxic effects on fish reproduction.

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METHODS

Sample Collection

Sea-surface microlayer samples were collected from 13 preselected stations in Puget Sound at four times between February and May, 1985 using the glass plate microlayer sampler (Harvey & Burzell, 1972; Hardy et al., 1985). The surface pressure (an indicator of the presence of surface-active organic films) was measured at a minimum of five points at each station using the spreading oil method (Adam, 1937). Stations sampled included relatively uncontaminated rural reference areas that are remote from areas of documented contamination; i.e., Sequim Bay and Central Sound, three urban bays--Elliott Bay near Seattle, Commencement Bay near Tacoma, and Port Angeles Harbor. Also, a station was located near Seattle's West Point sewage treatment plant discharge diffuser. Subsurface bulk water samples were collected only at three stations by opening a 3.5 liter glass jug at a depth of 0.2 m. Samples were immediately stored in the dark on ice and returned to the laboratory. Toxicity tests with unfiltered samples were initiated within 24 h of collection.

To determine the relationship, if any, between SMIC toxicity and the presence of surface slicks, 64 samples were collected and tested for toxicity during 1986 using a rotating Teflon drum sampler (Hardy et al. 1988c). Samples were from the same general sites as 1985, but were purposely collected either inside or outside areas of visible surface slick.

Toxicity Tests

Toxicity tests were conducted using either early stage sand sole embryos collected by neuston net from Sequim Bay or laboratory-fertilized sand sole eggs produced from gametes of one male and one female sand sole. In the laboratory, 4 to 10 replicate dishes containing about 15 eggs each and either 30 ml of microlayer or bulk water were incubated at 9°C for 6 days. At the end of the toxicity exposure period, dishes were examined and the number of normal live larvae recorded. Normal live larvae were completely free from the egg shell, showed active motility, and appeared to have normal morphology.

Toxicity tests were also conducted in situ. The field exposure system consisted of duplicate polyethylene cups (150 ml) held in a 0.09-m^2 styrofoam floatation board. The bottoms of the cups were open for water exchange through a 0.5-mm mesh nylon screen. The apparatus was submerged below the water surface and slowly brought to the surface to trap the surface film within each dish. Exposure systems were tethered to buoys in Sequim Bay (reference) and in urban bay areas with suspected contamination. Eggs from one female sand sole were fertilized with sperm from one male. At about 2 h postfertilization, approximately 50 embryos were added to each dish, allowed to incubate for 6 days in situ, retrieved, and examined for normal live-larval hatching success.

Chemical Analysis

The SMIC samples collected during 1985 and 1986 for toxicity tests were also analyzed for metal and organic contaminants. Details of the analytical methodology have been reported elsewhere (Hardy et al. 1988b). All 1985 samples were chemically analyzed. For 1986 samples, only 20 selected samples were analyzed. Samples were selected to cover a range of previously measured toxicity to sole embryos. In addition, for comparison to the microlayer, bulkwater samples from Sequim Bay, Port Angeles Harbor and West Point were analyzed.

RESULTS

Biological

The percent of normal live-larval hatch (LLH) at the end of a 6-day exposure period ranged from 0 to 96% (Table 1). Samples were analyzed by ANOVA and ranked by a Newman-Keul's multiple range test. Significantly toxic ($p < 0.05$) samples were all from urban bay sites. These included Stations 1 and 2 (Elliott Bay), 4 and 10 (Commencement Bay), and 13 (Port Angeles Harbor). Only microlayer samples from the urban bays showed significant toxicity. No toxicity was found in bulk water samples (0.2 m depth) from either urban or rural bay stations. The greatest survival, 86 to 96%, occurred in samples from the Central Sound (station 7) and Sequim Bay (station 8) sites. Out of 12 microlayer samples tested from urban bays, five showed significant toxicity (i.e., 55% or less LLH). In both the urban bays (but not the rural sites), normal LLH decreased with increasing quantities of surface active organic films according to:

2) $Y = -3.65X + 112$

where, Y = % normal live larvae

X = mean surface pressure (dynes cm^{-1}) at the urban bay station sampled

$N = 32$, $r = 0.6505$, and $p = < 0.001$

Fertilized sole eggs, placed in exposure chambers in situ and allowed to float in contact with the SMIC during embryogenesis, displayed marked reductions in hatching success in the urban bays compared to the reference stations. In 1985 normal live larval hatch in Port Angeles Harbor and Commencement Bay was only 4% and 42%, respectively, of that at the reference site (Sequim Bay). In Commencement Bay all larvae showed morphological abnormalities, primarily kyphosis (bent spine). In 1986, a Commencement Bay exposure resulted in only 55% normal live larvae and two exposures in Elliott Bay, performed a week apart, gave 79% and 78% as many normal live larvae as Sequim Bay. These results compare well to those found in the laboratory tests.

TABLE 1

Summary of Microlayer Contaminant Concentrations and Toxicity for Puget Sound (X = not measured; <DL = absent or below detection)

Location	Sample No.	Date-Station	Aromatics ($ug\ l^{-1}$)	Saturates ($ug\ l^{-1}$)	Pesticides ($ng\ l^{-1}$)	PCB ($ng\ l^{-1}$)	Total Metals ($ug\ l^{-1}$)	Normal Larvae (% normal)
	1	860314-I1	65	900	<DL	401	107	0
	2	860314-I2	298	<DL	2.7	45	88	76
	3	860314-K1	8031	236	9.0	<DL	3801	20
	4	860314-K2	17	<DL	<DL	311	3462	4
Elliott	5	860314-L2	<DL	74	<DL	<DL	58	100
Bay	6	850305-1	166	X	<DL	1201	683	0
	7	850305-2	81	X	<DL	<DL	226	53
	8	850305-3	13	X	2.8	1941	X	63
	9	850311-1	13	X	<DL	<DL	4753	0
	10	850311-3	<DL	X	<DL	<DL	28	80
	11	860319-A1	232	2057	43.8	254	247	0
	12	860319-C1	<DL	587	<DL	56	73	22
	13	860319-C2	16	847	<DL	61	87	7
	14	860319-D2	3	1073	<DL	443	317	5
Commencement	15	860319-E2	<DL	27	<DL	<DL	19	100
Bay	16	860319-F2	<DL	55	<DL	<DL	101	0
	17	860319-G1	<DL	423	<DL	90	280	0
	18	860319-H2	<DL	51	<DL	<DL	36	85
	19	850312-10	1123	X	0.5	2429	1725	0
	20	850312-11	<DL	X	0.2	<DL	9	78
	21	850306-4	170	X	4.1	3894	1357	55
	22	850312-4	<DL	X	<DL	<DL	80	84
	23	860220-A	<DL	12	<DL	<DL	96	98
	24	860306-A1	<DL	4	<DL	<DL	24	94
Sequim	25	860306-B2	1	24	<DL	<DL	101	92
Bay	26	860318-B2	<DL	3	<DL	<DL	15	100
	27	850303-9	17	X	<DL	<DL	66	74
	28	850303-8	<DL	X	<DL	<DL	69	86
	29	850310-8	<DL	X	0.1	<DL	25	96
	30	850303-bulkwater	<DL	X	<DL	<DL	1	81
	31	860314-C1	132	139	<DL	28	160	98
Central	32	860314-F1	2	2	<DL	<DL	25	98
Sound	33	860314-G2	1	2	<DL	<DL	11	100
	34	850305-7	12	X	0.1	<DL	34	88
	35	850311-7	<DL	X	0.1	<DL	81	89
Port Angeles	36	850212-13	591	X	<DL	X	471	36
West Point	37	850509-12	54	X	<DL	X	411	X

Chemical

Concentrations of potentially toxic metals in the microlayer of Puget Sound were extremely high at many stations (Table 1). Mean concentrations for total SMIC metals (1985 data) were 53, 58, 793, and 1420 g liter^{-1} for Sequim Bay, central Puget Sound, Commencement Bay and Elliott Bay, respectively . Eleven of the samples analyzed from the urban bay microlayers had high concentrations of several metals. The rank order of sites based on total metal concentrations was Elliott Bay > Commencement Bay > Port Angeles and West Point > Central Sound > Sequim Bay > bulk seawater from Sequim Bay (Table 1).

Saturated hydrocarbons (C-8 through C-34) were analyzed only in the 1986 samples. Discussion of individual concentrations at each station is beyond the scope of this paper, but high concentrations were found in many samples. Maximum concentrations of total particulate n-alkanes including phytane and pristane occurred in the following order (by site): Commencement Bay > Elliott Bay > Port Angeles > central Sound and West Point > Sequim Bay (Table 1).

Potentially carcinogenic, mutagenic and teratogenic PAHs occurred in the majority (57%) of the microlayer samples from Puget Sound. Extremely high concentrations were found at several stations in Elliott Bay and Commencement Bay. In general, concentrations were in the following order: Elliott Bay and Commencement Bay > Port Angeles, West Point and central Puget Sound > Sequim Bay. Bulkwater samples were analyzed from Port Angeles Harbor (Station 13), West Point (Station 12), and Sequim Bay (Station 8); none contained any detectable aromatic hydrocarbon contaminants.

Concentrations of pesticides were detected in 28% of the SMIC samples analyzed. Highest total pesticide concentrations were in Elliott Bay and Commencement Bay (Table 1).

DISCUSSION AND CONCLUSIONS

We found that toxicity was strongly correlated with the presence of slicks. In urban areas, toxic hydrophobic contaminants concentrate in natural films (Hardy, 1982). Wind and current patterns collapse the films into thicker visible slicks. Such slicks are not restricted to urban bays, but appear to move from place to place. Contaminated surface films could be carried by wind and surface currents, deposit in intertidal beach areas, contaminate shellfish, and impact other species, such as herring, that deposit eggs intertidally.

Even a cursory examination of Table 1 suggests possibile reasons for toxcity in some individual samples. In Sample 9, a low percentage (0%) of normal larvae occurred when exposed to a sample containing a very high metal content (especially Cu). Particles could be observed in this microlayer sample, and were most likely derived from sandblasting activity in the industrial shipyards of Harbor Island. Toxicity occurred in Samples 1 and 11 with high saturate concentrations and in Samples 6 and 8 with high PCB concentrations. However, most toxic samples contained a complex mixture of contaminants.

Both stepwise and principal component analyses suggested a strong inverse relationship between the presence of contaminants in the sea-surface microlayer and the hatching success of neustonic fish eggs (see Hardy et al. 1988b for details). Each of the measured contaminants appears to contribute to the overall decrease in the percentage of normal live larvae.

SMIC contamination in Puget Sound originates from a variety of sources. High Pb concentrations near the urban centers suggest a major source of surface contamination in these areas from gasoline combustion. Ratios of individual compounds (Prahl et al. 1984) suggest that the high levels of PAHs in many of our samples result from runoff or direct deposition of fossil fuel combustion products. Carbon preference ratios (Barrick et al. 1980) at some sites indicates the presence of uncombusted petroleum hydrocarbons. The presence of silver, a product of photographic processing, suggests inputs of domestic sewage, especially near West Point. Thus, the water surface appears to be contaminated with a complex mixture originating from a large variety of sources.

The significance of surface contamination in Puget Sound lies in the biological importance of the microlayer. The surface of Puget Sound is a nursery for the early life stages of many valuable species such as Dungeness crab and sole. Although eggs and larvae may be greater than 50 m (the depth of the microlayer) in size they often float or come into contact with the microlayer. Also, as Liss (1975) suggested, solubilization of airborne particles at the sea surface will lead to their rapid introduction into marine food chains via the high concentrations of microorganisms found in the microlayer.

Benzo(a)pyrene (BaP) is strongly carcinogenic. Hose et al. (1982) found a 30% increase in mortality of floating sand sole embryos exposed to 0.1 g liter^{-1} BaP. BaP concentrations between 3 and 123 g liter^{-1} occurred in seven of our samples. Metal concentrations in many of the SMIC samples exceed EPA water quality criteria by orders of magnitude. PCB concentrations in some samples were as much as 130 times greater than the Environmental Protection Agency Water Quality Criteria of 30 ng liter^{-1} (Table 1). Contamination of the water surface is widespread in Puget Sound. It is not restricted to urban bays, but occurs also in central Puget Sound. Toxicity to floating fish eggs results from a complex mixture of anthropogenic contaminants arising from a variety of sources. Overall, these results raise considerable concern regarding the environmental quality of the sea surface as a habitat for the developmental stages of flatfish and other species. Our results strengthen the argument that maintenance of a healthy marine environment will require development of quality standards and monitoring programs, not only for the water and the bottom sediments, but also for the sea surface.

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Shoreline Stranding of Floatable Materials in Seahurst Bay

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Introduction

Puget Sound is essentially a large, narrow bathtub, whose nearly vertical sides are the surrounding land (Fig. 1). Floatable materials (e.g., logs, oil and grease, styrofoam, and the contaminating toxicants and microorganisms that are associated with them) are present throughout Puget Sound (Fig. 2). These floatable materials can (but are unlikely to) escape from Puget Sound by passing through the narrow openings at Admiralty Inlet and Deception Pass. As a result, floatable materials from many sources have a high potential to eventually strand themselves on beaches in a "bathtub ring" effect. Our concern is that these stranded materials may degrade the shoreline environment, influencing the organisms as well as the people that use and enjoy the resources of our intertidal environments.

The objectives of our research, sponsored by the Municipality of Metropolitan Seattle (METRO), were to carefully examine an intertidal shoreline of Puget Sound and evaluate the level of chemical contamination of the water and sediment as well as the bacterial contamination of shoreline water, sediment, and shellfish. Chemical contamination was evaluated by examining the quantity of oil and grease materials contained in surface waters, contained in water within the crevices of the shoreline sands (sediment-water), and attached to shoreline sediment. Bacterial contamination was evaluated by determining the quantity of several types of indicator bacteria in the tissues of shellfish, on sediment, and in the sediment-water and surface water. These values were then compared to appropriate state or federal standards.

Methods

Our study area was Seahurst Bay, located on the eastern shoreline of south central Puget Sound. This embayment, including Seahurst Park, is a popular shellfishing and swimming site. The surrounding area has a moderate level of home development near the shoreline, two small creeks (Salmon and Seahurst) which discharge into the bay through the intertidal shoreline, an urban run-off channel at the foot of Seola Beach Drive (near the northern end of the bight), and a small, predominantly domestic, primary treated

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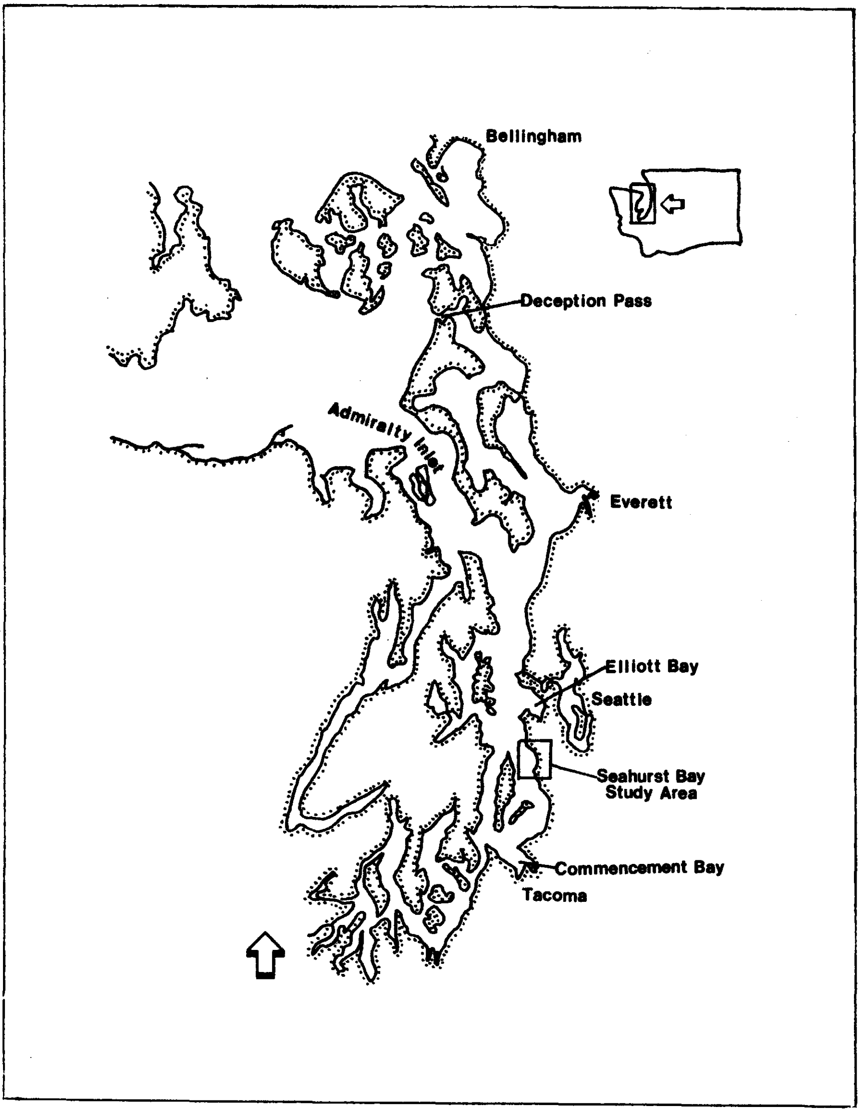

FIGURE 1. Puget Sound, Washington.

<u>FIGURE 2</u>. Floatable Materials Present in Puget Sound.

sewage effluent discharge located at Salmon Creek. This plant discharges into the center of the embayment, nearly 266 m from the shoreline at a water depth of approximately 73 m (Fig. 3).

Samples of surface water at four locations were carefully collected by slowly submerging the lip of a glass bottle beneath the water's surface. Samples of sediment-water were collected similarly from the surface of water which rose and filled holes that had been dug in the shoreline sediments at approximate intervals of 100 m over a shoreline distance of approximately 1400 m. Three-inch diameter sediment cores were collected at approximate intervals of 50 m over that same 1400 m of shoreline. Three individual shellfish (Clinocardium nuttallii) were collected every 50 m along the shoreline for a distance of approximately 800 m. All sample collections were taken from an approximate tidal height of 0 feet between 15 June and 8 August 1983. Water and sediment samples for chemistry or bacteriology were placed in glassware specially cleaned for this project by the chemistry and bacteriology laboratories at the Municipality of Metropolitan Seattle. The shellfish samples were placed in plastic bags without the addition of water. All samples were placed in coolers during the collection and on ice after return to the transportation vehicle. Samples were then transferred to METRO's Laboratory where they were processed for the various measurements.

Pentane extractable materials (oil and grease) were measured in each of the surface water, sediment-water, and sediment samples. Fecal coliform bacteria were measured in sediment-water and shellfish samples while enterococci bacteria were measured in sediment-water, and Clostridium perfringens bacteria in sediment samples. The chemical measurements were made by METRO personnel under the direction of Dr. Raleigh Farlow (currently QA/QC Officer for Region 10 EPA) while the bacterial measurements were made by Mr. Steve Fischnaller (currently at METRO's bacteriology laboratory) at the University of Washington's bacteriology laboratory under direction of Dr. Jack Matches. Other anecdotal observations included characterization of the color of sediment within the cores and the sediment odor type and strength.

Results

Surface water concentrations of pentane extractable materials in observable slicks ranged from 300 to 1500 $\mu g/l$, were highest approximately 1200 m from the northern portion of Seahurst Bay, and showed consistent decreases toward the northern portion of the bay (Fig. 4a). Pentane extractable materials in Seahurst Bay sediment cores ranged from 10 to 70 $\mu g/g$ (dry) and were highest near the Seola Beach Drive urban run-off site, near Salmon and Seahurst Creeks, and midway between Seola Beach Drive and Salmon Creek (Fig. 4b). Sediment-water concentrations of pentane extractable materials ranged from 200 to 1600 $\mu g/l$ and except for one station were consistently higher in the area between Seola Beach Drive urban run-offsites and the Seahurst Creek sites than at sites to the north of the Seola Beach Drive sites (Fig. 4c).

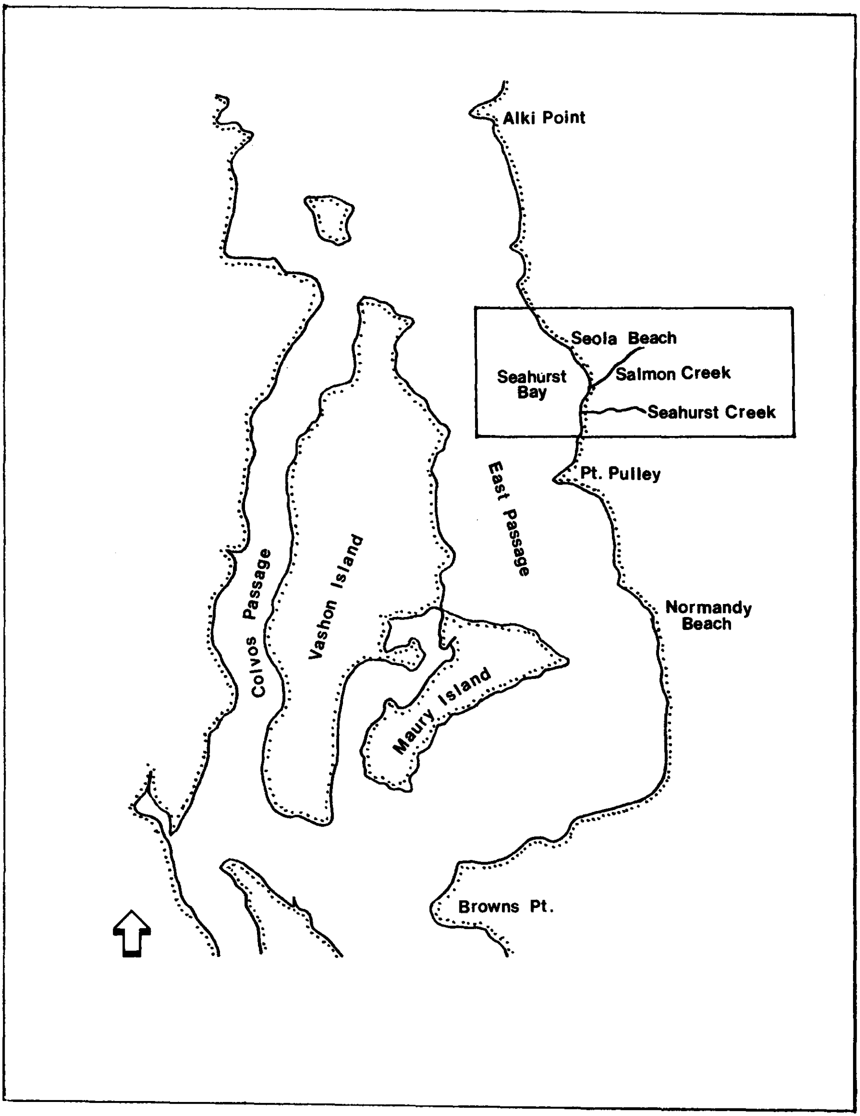

FIGURE 3. Seahurst Bay Study Area.

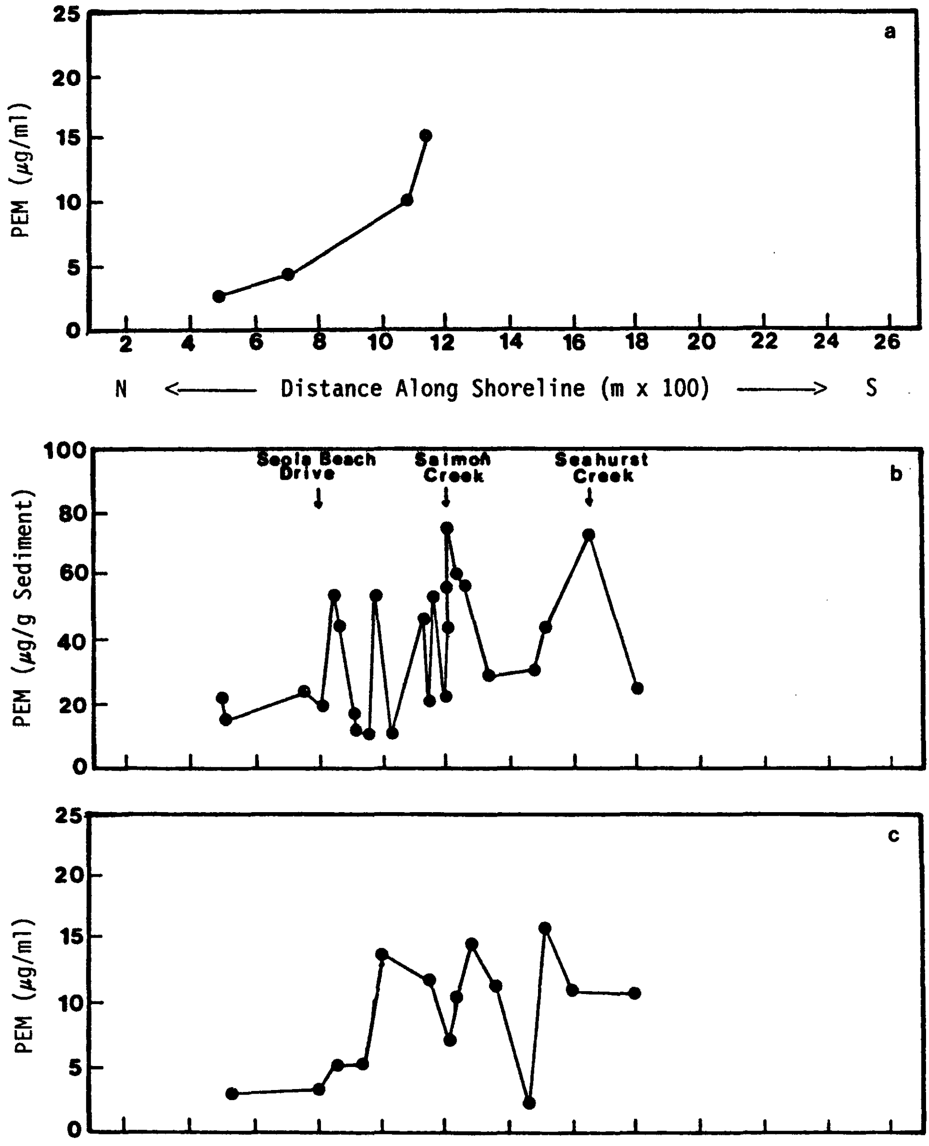

FIGURE 4. Concentrations of Pentane Extractable Materials in a) Nearshore Surface Slicks, b) Sediments, and c) Sediment-Water Samples Collected June 15 through July 12, 1983.

Concentrations of fecal coliform bacteria in sediment-water ranged from less than 10 to nearly 500 MPN/100 ml and were highest 1300 m from the northern portion of Seahurst Bay (Fig. 5a). Fecal coliform bacteria in the shellfish tissues ranged from approximately 10 to more than 16,000 MPN/100 g of tissue (wet) and had highest concentrations in the same samples at the 1300 m station (Fig. 5b). Enterococci bacteria ranged from less than detectable to more than 20/100 ml of sediment-water, again with the highest concentration at the 1300 m station (Fig. 6a). Clostridium perfringens spores in sediment ranged from less than 100 to over 4000 per 100 g of sediment (dry) with the highest concentration of bacterial spores at the 1300 m station (Fig. 6b).

Anecdotal observations included blackened, anaerobic sediments at the surface of beaches in regions of the urban run-off and areas surrounding the creek discharges, as well as centered in the Bay. These areas of blackened sediment contained a thin (<1mm) greenish layer at the surface, darkened, anaerobic sediments beneath that layer to depths of more than 60 cm and undarkened, non-anaerobic sediment beneath that darkened layer. Other areas of the beach, at higher tide levels or in regions without these potential shoreline inputs had blackened sediments at times but the blackened sediment was present in the lower parts of a sediment core and not at the surface. All of these observations were made on beaches with relatively coarse sand sediment.

Discussion

Oil and Grease

Elevations of oil and grease materials in sediments, surface water and sediment-water were observed in Seahurst Bay. The distribution of those elevated concentrations indicated an association with the urban run-off sites near Seola Beach Drive, near Salmon and Seahurst Creeks which may also contain some level of urban run-off, and in an area between the Seola Beach and Salmon Creek sites which apparently is isolated from potential urban run-off contamination. Materials associated with urban runoff appear to be the likely source of oil and grease contamination at Seola Beach Drive, Salmon and Seahurst Creeks.

Baterial

Contamination of shellfish resources, sediment, and sediment-water by bacterial indicators of sewage (fecal coliform, enterococci and Clostridium perfringens spores) showed a different distributional pattern from the oil and grease contamination. The maximum bacterial contamination level was located between Salmon and Seahurst Creeks at a site that showed moderate levels of oil and grease concentration. All of the bacterial measurements showed

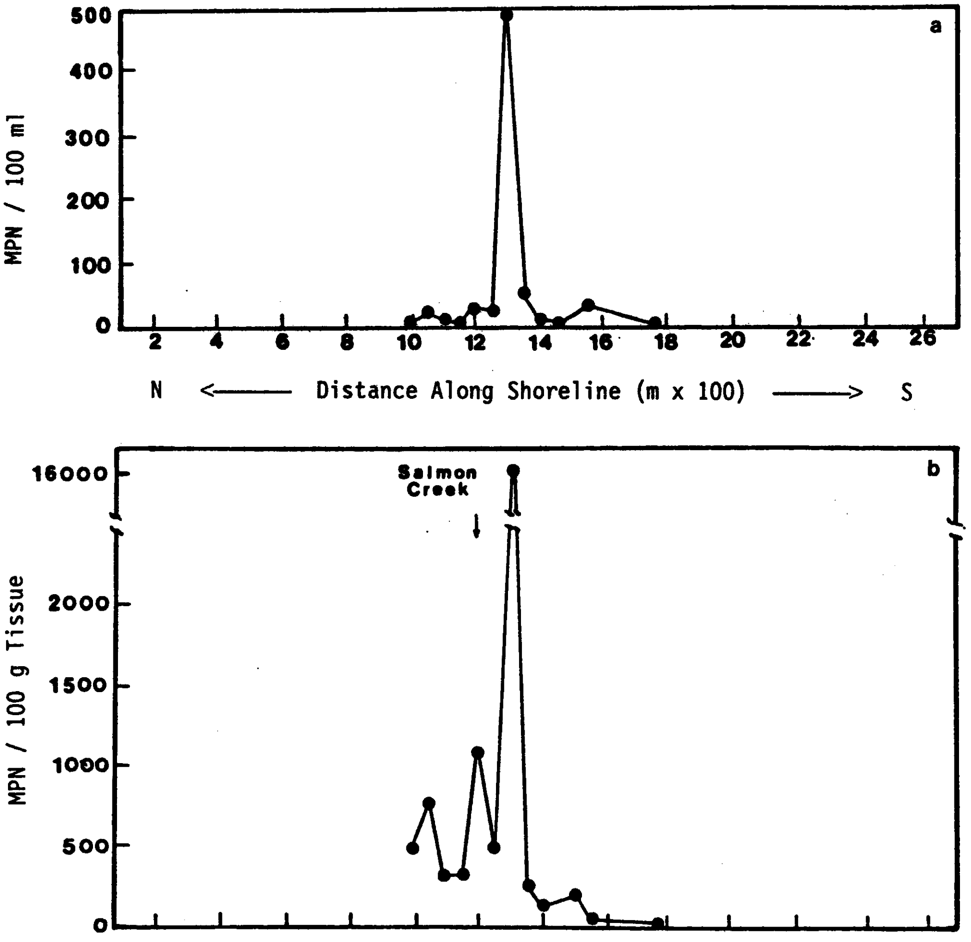

FIGURE 5. Concentrations of Fecal Coliform Bacteria in a) Sediment-Water and b) Shellfish Samples Collected on August 8, 1983.

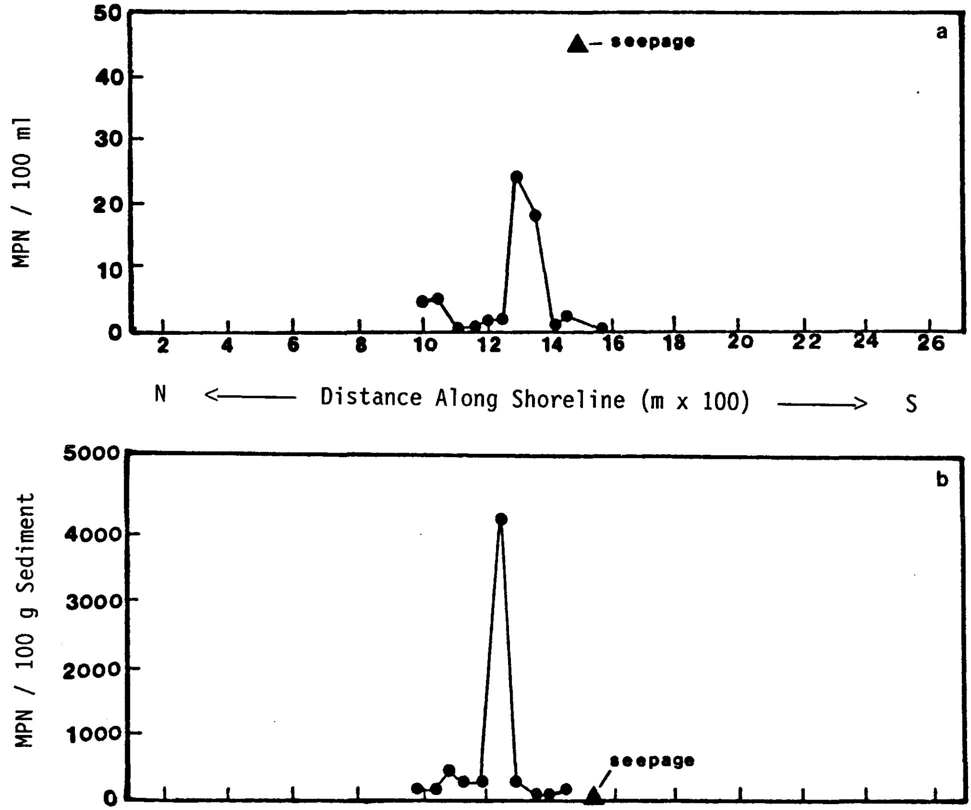

FIGURE 6. Concentrations of a) Enterococci Bacteria in Sediment-Water and b) Clostridium perfrigens in Sediment Samples Collected on August 8, 1983.

maximum concentrations at the same station, 1300 m from the northern border of Seahurst Bay, which is also 100 m south of Salmon Creek and 300 m north of Seahurst Creek. Unlike the case with the source of elevated oil and grease observations, this contamination does not appear to be tied to urban run-off as seen with the elevated oil and grease observations.

Field Observations

Anecdotal observations that were common between these sites were the vertical distribution of color and odor within the columns of sediment. At each of these sites the blackened, anaerobic near-surface layers overlying the deeper, cleaner sands is abnormal for coarse sand shorelines. These shorelines are typically clean throughout the sediment column. This type of layering is even opposite to the normal beach sediment profiles in finer grained sediments. Typically, sediment in these finer grained shorelines become darker and more anaerobic deeper within the sediment column. The typical profile results from progressively less oxygen as water percolates down into the sediment.

Some aspect of the environment has been modified to create a very different sediment profiling. We hypothesize that floatable organic materials, potentially from many sources, strand on the surface of the sediment during low tide at concentrations beyond the assimilative capacity of the shoreline environment. This stranding allows some fraction of the stranded materials to percolate through the surface to deeper depths. In its progress through the sediment column, oxygen is removed at more rapid rates near the surface. The sediment becomes anaerobic near the surface of the sediment column where the majority of the organic materials are retained. It is possible that the elevated oil and grease materials observed at these sites are the components causing the blackening of the top of the shoreline sediment column.

Significance of Oil and Grease Contamination

The significance of the elevation in oil and grease contamination of water and sediments and the indicator bacterial contamination in sediments and in the tissues of shellfish along these shorelines will be compared to existing federal and state criteria.

Oil and Grease Criteria from EPA 440/5-86-001 (May 1986)- Quality Criteria for Water 1986.

The Criteria are as follows:

> "For Domestic water supply: virtually free from oil and grease, Particularly from the tastes and odors that emanate from petroleum products.

For Aquatic Life:

(1) 0.01 of the lowest continuous flow 96-hour LC50 to several important freshwater and marine species, each having a demonstrated high susceptibility to oils and petrochemicals.

(2) Levels of oils or petrochemicals in the sediment which cause deleterious effects to the biota should not be allowed.

(3) Surface waters shall be virtually free from floating non-petroleum oils of vegetable or animal origin, as well as petroleum derived oils."

A major difficulty in establishing a criteria for oil and grease materials is that this material is not a specific chemical, but rather thousands of organic compounds with widely varying chemical, physical and toxicological processes (EPA 1986). A gross generalization allows placing oil and grease materials into categories based upon their origin from either petroleum related materials or oils from animal or vegetable origin. The source of the oil and grease materials in Seahurst Bay is unknown but probably consists of mixtures of petroleum compounds from roadside run-off as well as oils of animal and vegetable origin that are natural marine products as well as from activities of man.

Oil and grease from various petroleum related origins show wide range in toxicology (EPA 1986). However, even with this variation, acute 96-h LC50 determinations indicate that petroleum derived oil and grease concentrations of 100 $\mu g/L$ will harm the most sensitive tested organisms (EPA 1986). Sublethal toxicity of petroleum derived oil and grease materials are known to occur at levels as low as 10$\mu g/L$ (EPA 1986). It has also been documented that concentrations of petroleum hydrocarbon as low as 1 $\mu g/L$ are deleterious to marine organisms (Nassarius obsoletus) by interfering with its ability to detect its food source (Jacobsen and Boylan 1973). Salmon can detect petroleum hydrocarbons at concentrations as low as 10^{-7} $\mu g/L$, but the chemosensory reponse to hydrocarbons begins to degrade at 1 $\mu g/L$ (Pearson, Woodruff, and Johnsen 1987). Oils and grease from animal or vegetable origin are generally considered to be non-toxic to humans or aquatic life. The observed concentrations of oil and grease materials in surface water and in the sediment water samples during this reconnaissance ranges from approximately 200 to more than 15,000 $\mu g/l$. If the oil and grease materials were derived from petroleum origin there should be significant biological effects to the organisms that live in the water and in association with the sediment-water along these shorelines. Strict biological assessment of these shorelines was beyond the scope of these reconnaissance surveys and was not performed.

Significance of Bacterial Contamination

USFDA controls the interstate transport and commercial harvesting of shellfish while the State of Washington indirectly controls shellfish production and harvesting through the establishment of regulations that protect general water use through the establishment of water quality criteria classifications. Commercial shellfish harvesting operations are not allowed by the USFDA if the rearing waters have fecal coliform concentrations whose geometric mean is greater than 14 MPN/100mL or if the shellfish meat has concentrations of more than 230 MPN/100 g of tissue (wet).

The state of Washington further limits shell fishing through protection of certain classifications of water. Waters of the State of Washington that are classified as AA or A must assure that clams, oysters and mussels rearing, spawning and harvesting uses are protected. Class B water must assure protection of rearing and spawning while class C water is not required to protect clam, oyster or mussel rearing, spawning or harvesting. These requirements are met in Class AA or A water, if the geometric mean of fecal coliform concentrations is less than 14 MPN/100mL of water. This restriction is relaxed to fecal coliform counts of 100 MPN/100 mL in Class B water that assures protection of rearing and spawning potential of shellfish but not harvesting. Further relaxation of standards for Class C water does not protect the rearing, spawning or harvesting of shellfish. Water classified as C has geometric mean fecal coliform concentrations of 200 MPN/100mL. Seahurst Bay is classified as AA.

Fecal coliform counts in shellfish from Seahurst Bay at all but two stations were higher than allowed by the USFDA for commercial harvesting of shellfish. Fecal coliform counts in sediment-water exceeded the USFDA limit for commercial shellfish harvesting. All but one station in this bay met the requirements for Class B for the State of Washington. The concentrations at that one station would classify the water as Class C. Federal requirements do not allow commercial harvesting of shellfish, while the water quality is consistent with placing Seahurst Bay into a State Water Classification that does not assure the protection of shellfish harvesting. Either set of legislation indicates that the shellfish resources and the water in Seahurst Bay is not of sufficiently high quality to protect harvesting.

Conclusions

The intertidal shoreline environment of Seahurst Bay was contaminated by floatable oil and grease materials as well as bacterial contamination. It is likely that the blackening of sediments and production of anaerobic odors is a result of loading from floatable organic materials that strand on the shorelines at concentrations that exceed assimilative capacity of the shorelines for that material. These materials may be the oil and grease of natural and man-made orgin being released into the embayment through urban run-off collection systems. The bacterial contamination of shellfish resources are also significant, but appear to be related to another unidentified source. That source is influencing the shoreline at the 1300 m station.

The observed effects, potentially related to the stranding of floatable materials on the shorelines of Seahurst Bay, should be viewed as a warning. This small embayment is relatively isolated from the major contributors of contamination to Puget Sound's Elliott and Commencement Bays and as a result should be relatively clean. They are not and it is our hypothesis that floatable materials, especially those that are not as obvious as oil slicks, may be effecting the health of shoreline environments. Presently, the scientific evidence is largely anecdotal but we feel that an evaluation of shoreline contamination over a much wider area than a single embayment needs to be performed in Puget Sound.

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Epibenthic Harpacticoid Copepods as Indicators of Wetland Fitness[1]

Jeffery R. Cordell and Charles A. Simenstad[2]

Abstract

The occurrence and assemblage structure of epibenthic harpacticoid copepods may be highly diagnostic of wetland habitat quality in Pacific Northwest estuaries. This question was addressed in an examination of data from 56 various collections which were made in a variety of habitats in eight regions of Puget Sound and coastal Wasington estuaries. Data included density (no. m^{-2}) estimates of 81 taxa of prominent harpacticoids collected in late winter and early spring over the last ten years. Numerical classification (cluster and nodal constancy) analyses indicated eleven taxa assemblages distributed among thirteen site groups. Assemblages were generally associated among four zoogeographic regions: (1) central and (2) northern Puget Sound, (3) the Snohomish River estuary, and (4) Grays Harbor and the Columbia River estuary. Factors contributing to assemblage structure among these regions and their different habitats were perceived to be natural physical and biological characteristics such as degree of freshwater influence, sediment structure, and macrophytic and microphytic plant composition and standing stock. Differences in the prominence of certain assemblages, and component taxa within assemblages, suggested effects of anthropogenic stresses. While these analyses were not designed to indicate faunal assemblage responses to pollution or other stresses, the results indicate that epibenthic harpacticoid copepods may be viable indicators of habitat quality, especially relative to the fitness of wetland habitats for production of fish prey resources.

Introduction

The vast majority of classical studies which have investigated factors that control community structure in marine benthic organisms have utilized macroinvertebrates, i.e., those larger than 1 mm. But for marine meiobenthos, and particularly for harpacticoid copepods, empirical data on community structuring mechanisms is generally lacking (Hicks and Coull, 1983). During the last ten years, we have engaged in a number of studies in various locales and

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wetland habitats in Puget Sound and coastal estuaries of Washington to document assemblages of epibenthic harpacticoid copepods as prey of small nearshore fishes. Because fish can be highly prey selective, and their food resource availability must be understood at the level at which the fish discriminate, our studies have been necessarily highly taxa- and life-history intensive (i.e., identification to individual life-history stage and species level). In aggregate, these studies have indicated that epibenthic harpacticoid copepods are spatially and temporally heterogeneous in their abundance and population structure. While most of the original sampling designs did not overtly address factors which contributed to differences in harpacticoid assemblage structure, it was hypothesized that a synthesis and reassessment of these datasets would reveal associations or patterns which might suggest environmental factors contributing to assemblage structure. We were able to consolidate and compare these data only because the samples were obtained systematically using a uniform methodology and processed in the laboratory using the same protocol, both critical to any integrative analysis of this type.

Methods

For this exercise, we utilized selected epibenthic samples collected during a variety of studies in Washington State between 1980 and 1987. These collections were located in seven general localities (Fig. 1). All samples were collected by the authors except those from Everett Harbor, which were supplied by Dames & Moore, Inc; however, all copepods were verified by the senior author. At most of these localities, several discrete habitats were sampled, based on either tidal elevation or physical or biotic characteristics (Table 1). A total of fifty-six sites (Table 1) and eighty-one harpacticoid taxa/groups (Table 2) were utilized for the cluster analysis. Datasets were selected based on the following criteria: (1) the samples were collected between March and early June; and (2) harpacticoid copepods had been identified to species or lowest taxonomic level possible. Epibenthic organisms were sampled with one of several similar quantitative suction pump devices which draw water and associated organisms from inside a closed, ported cylinder through a collection sieve (Fig. 2). Harpacticoids from the samples were sorted and enumerated under a dissecting microscope. When necessary for positive identification, adult individuals were dissected and diagnostic appendages were mounted on a microscope slide in a streak of 50:50 water:glycerin solution. All data were recorded directly onto National Oceanic Data Center (NODC)-type computer forms, which utilize the 10-digit NODC taxonomic code. Tabulation and basic statistical analyses of data were performed using the computer package SUPERPLANKTON, which was specifically developed for epibenthic and zooplankton data in NODC

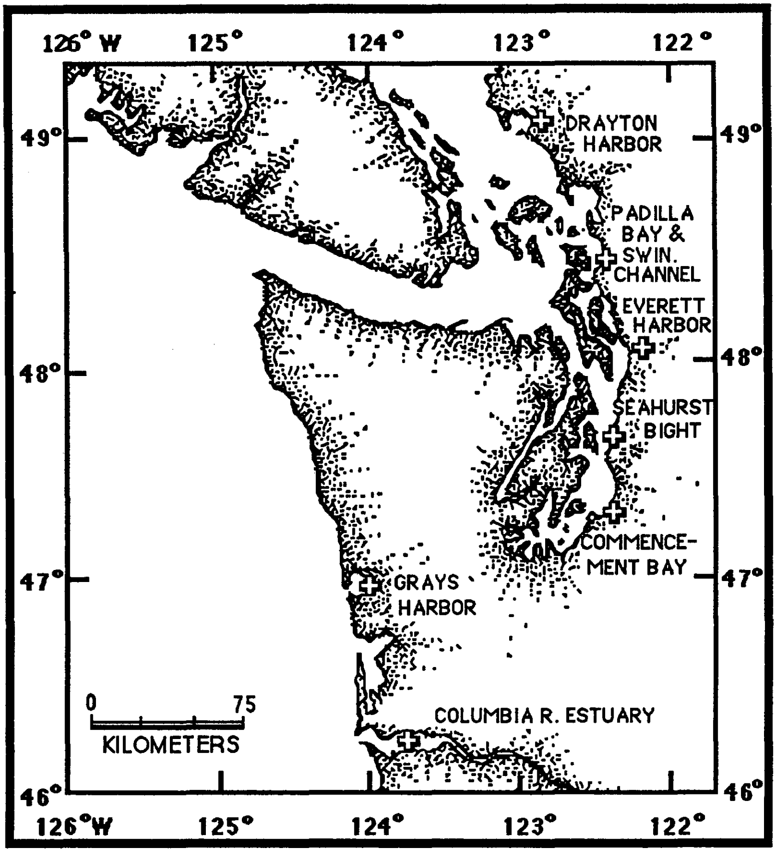

Figure 1. General regions of epibenthic harpacticoid copepod sampling (1980-1987) in the Pacific Northwest used for assemblage analysis.

format and reports basic statistics, including densities of individual taxa/life history stages and total epibenthic organisms in areal terms as numbers m^{-2}. Assemblage structure was examined quantitatively using agglomerative hierarchical classification (clustering) of the density data using the Bray-Curtis dissimilarity measure and a flexible ($\beta = -0.25$) fusion strategy. Collections (individual sample sites) constituted the entities and individual taxa densities the attributes. Similarities among sampling sites were determined by clustering the density data matrix after cube root transformation and standardization by division by taxa means; taxa assemblages were clustered by inverting the cube-root transformed data and standardized by division by taxa maxima. Clusters were objectively defined at the 0.65 dissimilarity level. To 8

Table 1. Pacific Northwest epibenthic collection sites utilized for cluster analysis.

Region/Characteristics		No. of sites	Habitats	Elevations (ft)
Drayton Harbor	Enclosed bay; low freshwater discharge	5	Marina, mudflat, eelgrass	+9.0 to +2.
Padilla Bay	Large embayment; low freshwater discharge	4	Marsh, mudflat, two *Zostera* spp.	+7.2 to +0.5
Swinomish Channel	Lagoon and associated tidal channels	3	Mudflat, tidal channel	+1.0 to 0
Everett Harbor	Estuary with large freshwater discharge; highly industrialized	13	Sand, detritus, mudflat	+3.0 to -6.0
Seahurst Bight	Fjord with no direct freshwater input	10	Cobble, sand, eelgrass	~+3.0 to ~-6.0
Commencement Bay	Estuary with moderate freshwater discharge; highly industrialized	1	Artificial gravel beach	+2.0 to -2.0
Grays Harbor	Estuary with large freshwater input; some industrial disturbance	9	Mud, sand, gravel	+2.0 to 0
Columbia River Estuary	Estuary with extreme freshwater input	11	Embayment, sand/ mudflats, channel slopes, mid-river sandflats	+2.0 to -2.0

Table 2. Number of taxa/groups from each family of Pacific Northwest harpacticoid copepods utilized for cluster analysis.

Family	No. of taxa/groups
Longipediidae	1
Canuellidae	1
Ectinosomatidae*	1
Darcythompsoniidae*	1
Tachidiidae	5
Harpacticidae	7
Tisbidae	2
Tegastidae	1
Thalestridae	8
Parastenheliidae	2
Diosaccidae	20
Ameiridae	5
Canthocamptidae	4
Cyllindropsyllidae	2
Cletodidae	7
Laophontidae	14

*Not identified beyond family level.

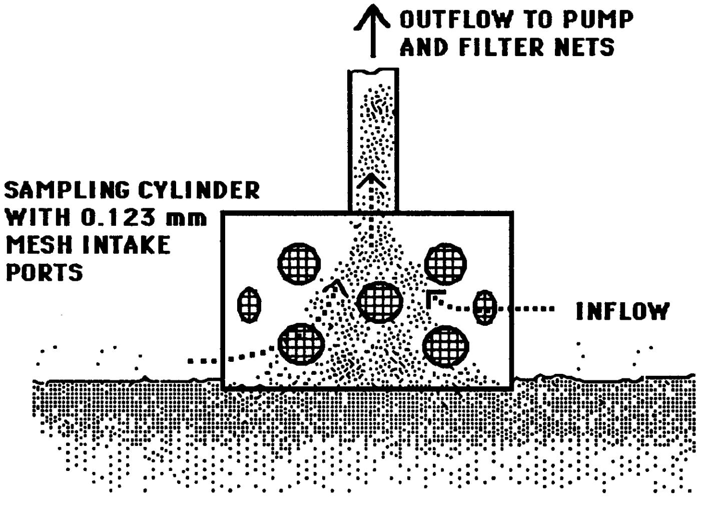

Figure 2. Schematic of basic epibenthos sampling apparatus used to collect harpacticoid copepods in intertidal and shallow subtidal habitats in Puget Sound and coastal Washington estuaries, 1980-1987.

interpret the coincidence among site and taxa clusters, a two-way nodal constancy plot was constructed from the cluster dendrograms, where constancy was quantified as the number of occurrences of each taxa in each cluster. More detailed descriptions of cluster and nodal constancy can be found in Boesch (1973), Everitt (1980) and Lambert and Williams (1962).

Results and Discussion

Hierarchical classification of the harpacticoid copepod density data distinguished eleven taxa clusters and thirteen site clusters; however, in examining the site-taxa coincidence by nodal analysis, these clusters were combined further to form eight harpacticoid assemblages and four geographical areas at a slightly higher level of dissimilarity (~1.0-1.2), which reflected more logical ecological and zoogeographical groupings (Fig. 3). By this exercise, a number of associations of harpacticoid assemblages with habitats in different geographical locations in the Pacific Northwest could be identified with some confidence. Knowledge of the habitat and regional characteristics suggest to us that many of the observed differences in assemblage structure can be attributed to variations in natural factors such as the proximity and magnitude of freshwater discharge, sediment structure, exposure to wind and wave action, presence and standing stock of epibenthic algae and macrophytes, etc. But, since differences and shifts in community structure and dominance can indicate environmental stress such as toxic pollution (the indicator species concept), these results also suggest one potentially useful approach to use harpacticoids as indicators of nearshore habitat quality in the Pacific Northwest. For example, a group of harpacticoids occurring only at a newly constructed artificial gravel beach in Commencement Bay (Table 3, cluster H), may be indicative of a habitat in a disturbed or early successional state.

Certain "indicator" species of harpacticoids have been found uniquely associated with poor environmental conditions (e.g., extremely organically enriched and anoxic conditions; Marcotte and Coull 1974; Moore and Pearson 1983). We have also found some of these harpacticoids (*Bulbamphiascus imus* and *Mesochra lilljeborgi*) to be the only harpacticoid taxa occurring at a severely degraded site in the Tacoma waterway, Commencement Bay (these samples were not used in this study because they were collected in the winter). The fact that indicator species dominated assemblages did not occur in the present data may be because none of the studies included in the analysis were specifically designed to describe an environmentally degraded habitat. Epibenthic harpacticoid sampling

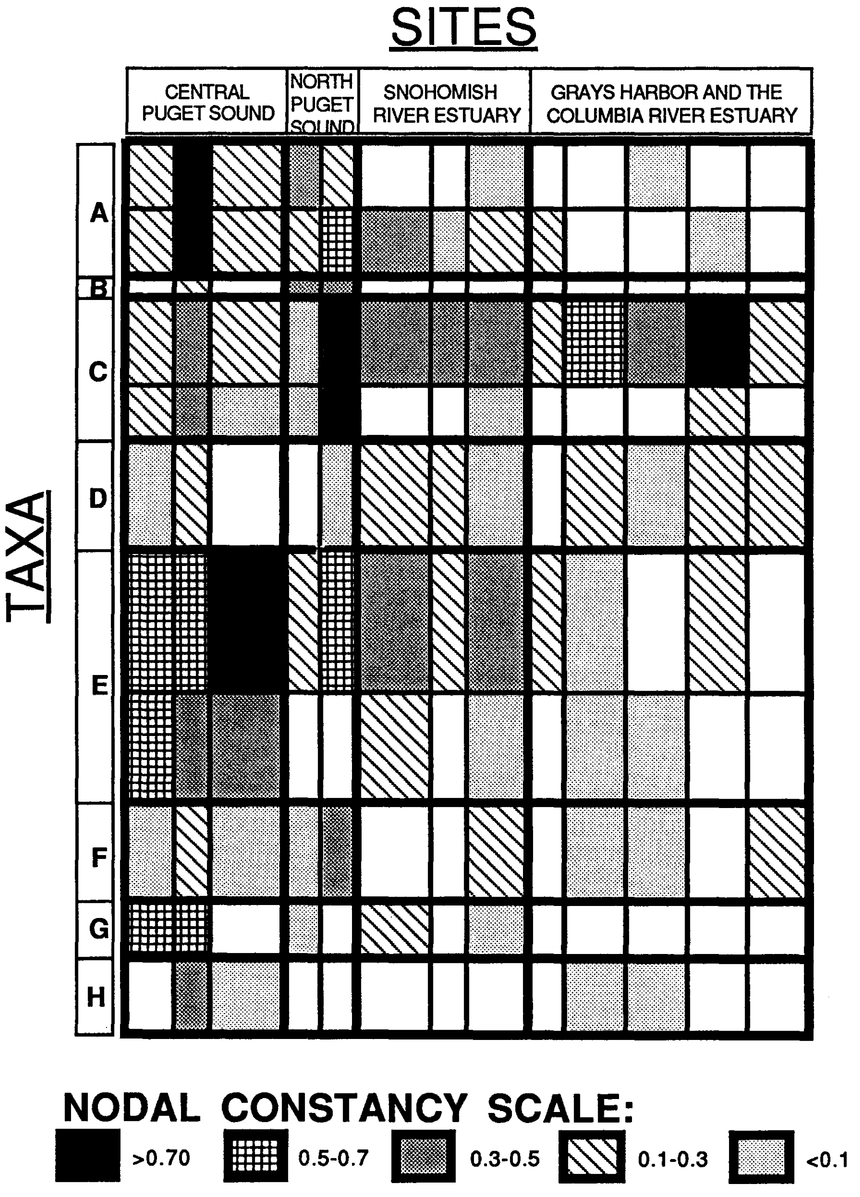

Figure 3. Coincidence between site and taxa clusters for harpacticoid copepods in the Pacific Northwest. Rectangles enclosed in lighter lines represent clusters at the 0.65 dissimilarity level; those enclosed by darker lines represent clusters at approximately the 1.0 dissimilarity level.

Table 3. Epibenthic harpacticoid copepod clusters associated with regions and habitats of Puget Sound and coastal Washington.

Cluster	Prominent region/habitat	Number of taxa	Representative taxa
A	Phytal-associated, Puget Sound	12	*Zaus* spp. *Harpacticus uniremis*
B	Unique to Drayton Harbor	2	*Ameira parvuloides* *Heterolaophonte capillata*
C	Sheltered littoral sand/mud flats in northern Puget Sound, coastal freshwater-dominated estuaries	13	*Scottolana canadensis* Ectinosomatidae *Microarthridion littorale* *Tachidius* spp. *Leimia vaga*
D	Oligohaline coastal estuaries; uncommon in Puget Sound	10	*Attheyella* sp. *Bryocamptus* sp. *Harpacticus arcticus* *H.* sp. A
E	Widespread throughout Puget Sound; rare in coastal estuaries	23	*Huntemannia jadensis* *Tisbe* spp. *Mesochra* spp. *Dactylopodia crassipes* *Diosaccus spinatus* *Amphiascoides* spp.
F	Unique to Drayton Harbor	9	*Nitocra spinipes* *Heterolaophonte hamondi*
G	Common only to Central Puget Sound; phytal and/or sand dwellers	4	*Amphiascopsis cinctus*
H	Unique to artificial beach, Commencement Bay	7	*Parastenhelia spinosa* *Pseudonychocamptus spinifer*

specifically designed to compare anthropogenically-stressed or disturbed habitats would presumably provide a more definitive test of this approach.

A harpacticoid assemblage structure analysis of wetland habitats in Puget Sound can also assess the "fitness" of the habitat in terms of its ability to support organisms which utilize that habitat but do not reside permanently in it. It is increasingly evident that nearshore assemblages of harpacticoid copepods are important prey resources for a variety of juvenile and other small fish, and that often only a small number of taxa in the assemblage are utilized by the fish (Hicks and Coull 1983; Gee

Gee 1987; Simenstad et al., in prep.). In the Pacific Northwest, harpacticoid copepods in the genera *Harpacticus, Tisbe*, and *Zaus* comprise a primary prey resource for juvenile pink and chum salmon during their critical early outmigration from fresh to marine waters, and for a number of other small or juvenile fish when they pass through nearshore habitats (Cordell, 1986; Simenstad et. al, in prep; Cordell and Simenstad, in prep.). Therefore, loss or degradation of nearshore habitats which support such ecologically-important harpacticoid species may indirectly affect the economy of the region by decreasing early life-history survival of commercially important species.

Reduction in habitat fitness may also be indicated by exclusion of particularly sensitive taxa or lower diversity in associations which contain such taxa. For instance, harpacticoid clusters which are characterized by the presence of *Harpacticus, Zaus,* and *Tisbe* (Table 3, clusters A and E) have a strong association with central and northern Puget Sound, but weaken in the Everett Harbor area, and almost completely drop out in the Columbia River estuary and Grays Harbor. While this drop in taxa cluster-regional association specifically in Everett Harbor may be due to environmental degradation, it may also be generally due in these regions to other physical factors, such as salinity or substrate structure. Again, more directed studies would have to be designed to separate harpacticoid assemblage differences due to such natural environmental "stresses" from those associated with habitat degradation. Such studies will require microhabitat-level, seasonal, and species-specific definition of the harpacticoid assemblages. Unfortunately, the large body of environmental baseline and monitoring information has concentrated on macrofauna and largely ignored meiofauna such as harpacticoids. Because they are abundant in nearshore environments, may be sensitive to environmental stress, and are important prey for consumers at higher trophic levels, we believe that epibenthic harpacticoid copepods may lend themselves well as biological indicators, and should be considered more thoroughly in future monitoring studies.

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Summary Statement

Ronald M. Thom*

Papers presented in the session on *The Nearshore Zone: Is it a Good Indicator of the Sound's Health and Productivity* highlighted both the importance and complexity of nearshore systems in Puget Sound. Papers by Boatman, Thom et al., and Nishitani illustrated the intense degree of coupling between nutrients and primary production that occurs in the zone. The papers showed that nutrient depletion is a common feature of nearshore systems in Puget Sound, in contrast to deeper areas which are rarely nutrient limited. Boatman developed the first comprehensive model of production and nutrient dynamics for nearshore systems in Puget Sound. He used the model to explain oxygen depletion resultant from dinoflagellate blooms occurring in late summer. In addition, the model predicted the degree of sewage treatment required to reduce the probablity of sustained oxygen depletion. Thom et al. showed that nutrient limitation occurs in macrophyte dominated systems. Streams may be contributing significant amounts of nutrients to nearshore systems during the dynamic spring-early summer period of primary production. Nishitani summarized several years of data on the toxic dinoflagellate *Gonyaulax catenella*, and showed that phosphate concentration may play an important role in producing blooms. Nishitani presented explanations, largely nutrient based, for the apparent lack of blooms in southern Puget Sound. Hardy and Antrim showed that conventional pollutants and toxic organic compounds are concentrated in surface slicks in Puget Sound, in comparison to very low concentrations in bulk water samples collected immediately below the surface. Concentrations of these contaminants in the microlayer can exceed EPA standards, and are toxic to floating sand sole eggs. Word and Ebbesmeyer presented a mechanism through which microlayer contamination reaches beaches in a process referred to as the "bath tub ring" effect. They illstrated the effects of reconcentration of surface slicks on a beach, previouly referred to as clean, by showing highly elevated organic matter and bacteria concentrations. In the final paper, Cordell and Simenstad recommended epibenthic harpacticoid

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copepods as an ecologically meaningful and highly sensitive group of organisms for determining the environmental quality of nearshore habitats. These organisms are the principal prey resource of juvenile salmon, and are sensitive to rather small changes in the benthic conditions. They summarized 10 years of data at 54 habitats in their paper, which represents the most comprehensive summary of its kind for the northwest.

The nearshore zone is a complex and highly dynamic system, with relatively strong benthic-water column coupling. Currents are poorly known in these sytems due to the complexity inherent in a tidally influenced, heterogeneously scalloped shoreline. All of the authors concluded that the nearshore zone was poorly understood due to a lack of historical focus by researchers and agencies. And finally, in response to the question posed by the session title, these systems appear to be the area within which to find the most easily detected indications of the health and productivity of Puget Sound.

HABITAT MODIFICATION

WHAT IS THE STATUS OF PUGET SOUND HABITATS AND CAN WE CREATE HABITATS TO MITIGATE LOSS?

Charles A. Simenstad,
University of Washington
Session Chair

Introductory Statement

Charles A. Simenstad*

More than 70% of the wetland habitats in the major estuarine deltas of Puget Sound have been destroyed since the mid-1800's. Therefore, a critical mandate of the **PSWQA Planning Act** was to recommend "protecting, preserving and, where possible, restoring wetlands, wildlife habitat, and shellfish beds throughout Puget Sound." This session focuses specifically on wetland habitats, not because other habitats are less important to Puget Sound as an ecosystem, but because wetlands are the more sensitive ecotones, or "tension" zones between the Sound and the human communities which have developed around it. In addition, these other habitats are being addressed more extensively in the other sessions during this meeting. We have organized the **Habitat Modification** session to focus on five basic questions: (1) What habitats have we lost? (2) What is the significance to the ecosystem? (3) What are the resource agencies doing to evaluate the extent, sources, and consequences of wetland habitat loss and modification? (4) Are there methods to mediate or reverse this loss? and, (5) How complete is the science required by wetland legislation, and the implementation thereof, in reversing these trends?

Accordingly, the papers we have selected describe the spectrum from basic ecological research to management policy: Anthropogenic and natural dynamics in estuarine wetland change; linkages between wetland production and estuarine consumers; wetland management strategies implemented by state resource agencies; several approaches to wetland mitigation; and, appropriately as the anchoring presentation, an assessment of the efficacy of the Clean Water Act, Section 404 Permit Process in ensuring compliance of mitigation intent and policy.

They represent the breadth, as well as the dearth, of wetland habitat science in Puget Sound and bear out that these wetland ecotones are, indeed, under tension.

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CHANGES IN DUWAMISH RIVER ESTUARY HABITAT OVER THE PAST 125 YEARS

GEORGE BLOMBERG, CHARLES SIMENSTAD, AND PAUL HICKEY*

INTRODUCTION

Changes in the topography, physical processes, and biological communities in estuaries commonly occur over a time span of hundreds to thousands of years. However, changes in the Duwamish River estuary, as with many urban and developed estuaries in Puget Sound, have been greatly accelerated by human influence over the past century.

This preliminary study identifies estuary habitats and resources which have been diminished, altered, or lost in the recent past. Historical documents and charts, beginning with the period 1854 to 1860, are used to depict incremental changes in selected estuarine habitat types, particularly wetlands. This data is combined with information describing alterations in the flow and discharge of tributaries to the estuary. Additional historical analysis documents changes in land use in shoreland and floodplain areas of the estuary.

General conclusions are drawn concerning the effect of lost or diminished estuarine resources on functions and processes, the biological values of the estuary, important to fish and wildlife using the system. Conclusions include notes on strategies for recovery of estuarine features lost to development and planning for mitigation of development use impacts in the Duwamish estuary based on historical analysis.

APPROACH

Assessment of historical ecosystem changes requires gathering information on the estuary as it was in the past, and comparison of that information with recent data. Data describing changes in the Duwamish River estuary are limited and it was necessary to extract and interpret information from historical data compiled for other purposes. A number of assumptions (described in the following section) concerning treatment of data were necessary to allow derivation of the information presented in this study.

The Duwamish River estuary has been inhabited for thousands of years; however, the effects of human activity were negligible until the arrival of large numbers of new settlers in the late 1800s. Changes in the estuarine environment over the past one hundred and twenty-five years have been much more significant than the effects of previous human activity, and far exceed the naturally-caused changes expected in estuarine ecosystems. Therefore, our analysis of the historical data base began with the mid 1800s.

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METHODS AND MATERIALS

Historical Data Base

Historical documents examined for this study included old maps and navigation charts, photographs dating from the late 1800s, and published data and information.

The availability of historical information determined the scope of this study. Beginning with the earliest accurate map information and historic data describing the estuary's drainage basin we compiled a series of four summaries describing the estuary over three forty year intervals, 1854-1860, 1908, 1936-1940, and 1985-1986. Evaluation of changes in the estuarine system was then based on changes observed in the intervals separating these "estuarine snapshots" and on cumulative changes recorded between 1854-1860 and the present.

Analysis of changes in estuarine habitat types relied on the methods used by Thomas (1983) in his analysis of historical changes in the Columbia River estuary. Information identifying past abundances of estuarine habitat types was interpreted from maps published by the U.S. Geological Survey detailing Elliott Bay and the Duwamish River in 1854 (USGS 1980) and 1908 (USGS 1908). These maps are an accurate representation of the early estuary. However, these sources did not include detail sufficient to distinguish habitat types in the mid and upstream reaches of the estuary. Map information describing the downstream portions of the estuary was applied to upstream reaches to estimate areas of tidal marsh and tidal swamp habitat. This method for projecting the historic abundance of habitat areas was verified with old photographs where possible. Recent USGS quadrangle maps, together with aerial photographs from 1936, 1946, 1960, 1974, and 1986, were used to describe estuarine habitat and shoreland conditions in 1936-1940 and 1985-1986.

Information describing alterations in the course and discharge volumes of tributaries to the estuary, and the area of the estuary's watershed, was compiled from numerous published records and studies (USGS 1972, 1985, and Washington Department of Fisheries 1975). Available discharge and total water flow data for the estuary was hindcast to estimate discharge conditions for the period 1854-1860, prior to extensive watershed alteration. Estimates for discharge volumes in 1936, 1960, and 1986 were interpolated or taken directly from published data, as appropriate. Estimates of the watershed area of the estuary prior to 1911 and for the present drainage area of the estuary were computed from USGS and Washington Department of Fisheries sources.

Information on sources of pollution and contamination in the estuary was obtained from published state and federal water quality analyses (Washington Pollution Control Commission 1955; USGS 1972; EPA 1985; and, METRO 1985).

Definitions

Because of the limited resolution of historical information, five broadly defined estuarine habitats were identified, as follows:

(1) Medium depth water: From approximately minus fifteen feet mean lower low water (MLLW) to approximately 0 feet MLLW.
(2) Shallows and flats: From MLLW to the waterward edge of tidal marsh or swamp vegetation or the approximate mean higher high water (MHHW) elevation (the shoreline indicated in Figures One through Four).
(3) Tidal marshes: Areas where emergent vegetation and low shrubs were present, extending from approximately eight feet above MLLW to two to three feet above MHHW.
(4) Tidal swamps: Wetlands comprised of shrub and forest vegetation and extending landward to the line of non-aquatic vegetation. Tidal swamps are generally inundated by spring tides, yet may extend waterward in some areas to MHHW.
(5) Riparian shoreline: Areas of natural shoreline with marsh or swamp habitat types present or areas where a natural buffer between upland and aquatic area habitats is present.

The following categories were used to describe the displacement of estuarine habitat with development uses:

(1) Developed floodplain and shorelands: Locations where estuarine habitat areas have been filled to create uplands and where shoreland areas have been built and committed to intensive development use.
(2) Developed shoreline: Areas where former estuarine riparian habitat has been displaced by developed shoreline, including pier structures, bulkheads, and riprap shorelines.
(3) New developed shoreline: New shoreline with development characteristics created from fill in former estuarine shallows and flats.
(4) Deep water: Water area with a depth greater than fifteen feet below MLLW. This habitat type was not a feature of the undeveloped estuary and was created by dredging of former estuarine shallows and flats for the purpose of navigational access.

RESULTS

Undeveloped Estuary (1854-1869)

Early records of the Duwamish River estuary indicate a river meandering through significant areas of tidal wetlands to enter Elliott Bay via three main distributary channels (USGS 1980). The meandering channel included approximately 440 acres of medium depth aquatic area, with a total shoreline length of approximately 93,000 feet (Figure One and Table One). Approximately 1270 acres of tidal marshes and 1230 acres of tidal swamp are estimated to have been present in the estuary. A significant feature of the downstream portion of the estuary was a broad expanse of unvegetated intertidal flats and shallows (approximately 1450 acres) at the mouth of the estuary bordering the south margin of Elliott Bay.

Discharge of freshwater through the estuary was estimated to range between approximately 2,500 cfs and almost 9,000 cfs (Figure Eight). The watershed area of the estuary included lakes Sammamish and Washington, and the Cedar, Black, Green, and White rivers. The combined area of the river systems historically tributary to the Duwamish Estuary was approximately 1640 square miles (Figure Nine).

1908

By 1908, the early effects of settlement in the estuarine floodplain were rapidly becoming evident (Figure Two). Significant reductions in the area of shallows and flats (26 percent), tidal marshes (17 percent), and tidal swamps (52 percent) had occurred relative to the period 1854-1860. Shallows and flats were filled with materials generated by dredging for the purpose of navigational access and with material removed from uplands east and north of the estuary during urban regrade activities (Benoit 1979). Note that approximately 1210 acres of estuarine wetlands and floodplain area had been filled or converted to development uses. These development activities added approximately 4,000 linear feet of new developed shoreline to the estuary in area formerly comprised of estuarine shallows and flats (Figure Seven).

Little information is available to characterize the effects of development on water quality in the estuary during this period. However, early industrial uses and activities were generally resource-based and it is unlikely that significant amounts of complex contaminants were introduced to the estuary at this time.

1936-1940

Figure Three approximates the Duwamish Estuary during the period 1936-1940. The Duwamish Waterway was completed in 1917, straightening and widening the former channel of the river. This navigational improvement resulted in a small loss in medium depth water area (five percent) compared with the area of the historic river channel. Note that excavation of the navigation channel created deep water habitat (approximately 240 acres, measured south of the historic MLLW contour) at the mouth of the estuary where none existed formerly. New deep water habitat was excavated largely at the expense of shallows and flats area, reducing the abundance of this habitat by approximately eighty-eight percent compared with 1908. A significant amount of shallows and flats were also filled between 1908 and 1936-1940 due to regrade activities. Tidal marshes were further reduced (84 percent) compared to 1908 and tidal swamp habitat was completely absent by 1936-1940.

Developed floodplain and shorelands in 1936-1940 increased to approximately 3750 acres (310 percent greater than 1908). This development activity reduced estuarine riparian shoreline by approximately 58 percent compared with 1908, while increasing the linear footage of developed shoreline by approximately 1200 percent.

Significant changes in the watershed of the estuary began in the period 1900 to 1911, with water diversions to the cities of Seattle and Tacoma. In 1911 the flow of the White River was completely diverted to the Puyallup River drainage, into Commencement Bay. In 1916, as a result of construction of the Lake Washington Ship Canal, the flow of the Black River was eliminated and the Cedar River was diverted into Lake Washington. Cumulatively, these changes reduced the estuary's watershed to approximately 483 square miles, a seventy-one percent reduction (Figure Nine).

With increased urban development in the floodplain and shorelands of the estuary, and due to diversification in industrial activities, the estuary was receiving significant amounts of waste materials. In the period 1936-1940, eight direct sewer outfalls are noted in the estuary, with four combined sewer overflows (Washington Pollution Control Commission, 1955). In addition, twelve sites in estuarine shoreland locations were discharging toxic waste materials directly to the estuary, including the following compounds: cyanides, chromates, acid pickling liquor, caustic liquids, synthetic resins, formaldehyde compounds, phenols, pentachlorophenols, acetylene sludges, and arsenic compounds.

The Present Estuary 1985-1986

The Duwamish River estuary as we know it today has changed significantly compared with historic conditions (Figure Four). Since 1936-1940, shallows and flats have decreased another twenty-five percent and tidal marsh areas have decreased eighty-eight percent. Continued filling along the margins of the straightened channel has resulted in the loss of approximately 30 acres of medium depth habitat (approximately eight percent less area than present in 1936-1940). Conversion of estuarine shoreline from vegetated to developed characteristics has also resulted in approximately 50 percent more developed shoreline than estimated for 1936-1940.

Developed shorelines and estuarine floodplain now total approximately 5,200 acres, a 39 percent increase from 1936-1940. Developed shoreline measures approximately 53,000 linear feet. Note that continued filling of estuarine aquatic area to expand uplands for development use resulted in a twelve percent reduction in the deep water habitat created by dredging of the north portion of the Duwamish Waterway and the East and West Waterways (i.e., new deep water habitat was refilled, decreasing this habitat type by approximately 30 acres).

The watershed area of the estuary remained at approximately 483 square miles and the discharge through the estuary has maintained approximately the same hydrologic cycle and amplitude.

Beginning with early water pollution control efforts in the 1950s, direct contaminant and industrial discharges to the estuary were significantly reduced and untreated sewage discharges were limited to sewer overflows during storm events. In place of eight direct sewer outfalls noted in 1936-1940, twelve major combined sewer overflows now represent potential discharges of untreated waste to the estuary. A significant change in drainage of developed areas surrounding the estuary included construction of extensive storm drain systems, collecting runoff from urban areas and discharging the drainage at point sources. Approximately thirteen major storm drains enter the estuary. In addition, numerous smaller storm drain systems drain developed former floodplain and shoreland areas. Although industrial activities are more actively controlled, a significant number of sites continue as sources of contamination to the estuary via direct contact or discharge to the estuary or indirectly by affecting groundwater in areas continuous with the estuary (EPA 1985 and METRO 1985) (refer to Figure Four).

DISCUSSION

Existing conditions in the Duwamish River estuary represent the cumulative result of three principal development activities over a period of 125 years. First, dredging and fill in the immediate area of the estuary have physically replaced the features of the estuary, substituting a uniform deep channel bounded by intensively developed uplands for the former complex system of estuarine mid-depth channels, intertidal flats, and fringing tidal marshes and swamps. It is estimated that less than two percent of the former area of shallows and flats and tidal marshes remain, while tidal swamp habitat has been completely eliminated.

Second, the watershed of the estuary has been significantly reduced and the volume and temporal distribution of water discharging to the estuary has been greatly diminished compared with past flows. Approximately 30 percent of the former drainage basin of the estuary remains. The monthly profile of water discharge represents volumes approximately 70 to 75 percent less than estimated for the unaltered watershed.

Third, the estuary is now bordered by approximately 5,200 acres of uplands, formerly estuarine wetlands and shorelands, supporting industrial development and other uses.

These extensive changes in the estuary have altered the water quality in the Duwamish and affect the estuary's ability to support fish and wildlife resources. For example, the combination of a deepened channel, reduced watershed area and freshwater discharge, and pollutant loading has resulted in decreased levels of dissolved oxygen in the estuary during the late summer and early fall months. These oxygen "sags" were first identified in the 1930s, and dissolved oxygen levels below five ppm were recorded during the 1950s (Washington Pollution Control Commission 1955). It is not known if low levels of dissolved oxygen continue to adversely affect fish and wildlife in the estuary; however, the potential for the altered estuary to accumulate significant loadings of dissolved contaminants during periods of low fresh water discharge remains. A recent fish kill associated with high levels of dissolved materials took place in the fall of 1986 (WDOE 1986).

In addition to the effects of decreased freshwater discharge on estuarine water quality, physical changes in tributaries to the estuary have altered anadromous fish populations using the system. In 1860, the Duwamish basin contained approximately 1640 square miles, including approximately 1900 linear miles of rivers and streams. The majority of these streams were accessible to and used by five species of Pacific salmon and trout:

chinook, pink, coho, chum, and steelhead (Suckley and Cooper 1860, Everman and Meek 1998, Wilson 1974, Grette and Salo 1976, and Turner 1976). In addition, sockeye were documented in Lake Washington (Rathbun 1900) and in the Cedar River (Everman and Meek 1896) and were probably harvested by native fishermen in the estuary (Butler 1987). The present 480 square mile watershed includes approximately 640 river miles; but only approximately 125 miles are accessible to anadromous fish, due to the City of Tacoma water diversion dam constructed in 1911. This represents a ninety-three percent reduction in the streams and rivers available to migratory fish. Chinook, coho, and chum salmon are still found in the present Duwamish-Green River system in sizeable numbers, largely the result of fish hatchery releases by the Muckleshoot Indian Tribe and the Washington Department of Fisheries. Steelhead are present as well, while pink and sockeye salmon are found incidentally.

Therefore, the capacity of the Duwamish River estuary to support resident and migratory fish and wildlife populations has been significantly altered due to losses in river basin capacity and estuarine habitat. Clearly, the potential for natural production of anadromous fish is limited due to the loss of spawning habitat in rivers and streams no longer tributary to the estuary or where physical obstructions have been placed. The significant reduction in estuarine intertidal wetlands has dramatically diminished the production of most of the invertebrate food organisms important to juvenile salmonids migrating through the estuary. In addition, loss of the complex mosaic of marshes, flats, and channels has undoubtedly diminished available refugia; and circulation changes have modified the stretches of the estuary used for physiological transition. The overall outcome is probably shorter residence times and lower growth in the estuary, resulting in a net lower survival of the Duwamish salmon populations compared to less developed estuaries and watersheds. For other fish species and wildlife, including waterfowl and mammals, the estuary no longer contains wetland habitats important for feeding, reproduction, and refuge and the water quality of the estuarine system contrasts dramatically with the historic estuary, reducing the abundance of these organisms in the area.

These direct alterations to the estuary are significant; and some restoration of wetland habitat function is necessary if fish and wildlife utilization is to be enhanced. Indirect consequences of these changes--though less obvious--also constrain the potential effectiveness of future enhancement efforts. For example, the extent of salinity intrusion and the location of the estuary's turbidity maximum, or "null zone", have shifted up-estuary as a result of the drastic reduction in riverine discharge. Since both characteristics of the estuary's circulation are closely linked to the distribution and maintenance of wetland communities, the efficacy of habitat restoration or enhancement for fish and wildlife may depend on our understanding of their role in the historic estuarine ecosystem.

CONCLUSION

As indicated by this historical profile of the Duwamish River estuary, significant alteration has taken place. The estuary remains an important site for development uses, and additional losses of fish and wildlife habitat are expected to continue. It is essential that the effects of future development uses and activities on estuarine fish and wildlife resources be offset by appropriate mitigation actions. The preliminary data presented here suggest that more detailed understanding of the historic conditions in the estuary may provide guidance in determining appropriate actions for compensation of future estuarine habitat losses. Precise information identifying the relative importance of lost estuarine resources, and their associated biological functions and processes, must serve as a template for location and construction of fish and wildlife enhancement and restoration actions as well as specific mitigation projects.

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TABLE ONE

DUWAMISH ESTUARY HABITAT CHANGES
1854 to 1986

Year and Percent Change

Estuarine Habitat Types	1854	1908	1940	1986	Total Percent Change
area in acres					
Medium depth water	440	410(-7%)	390(-5%)	360(-8%)	minus 18%
Shallows and flats	1450	1080(-26%)	130(-88%)	25(-81%)	minus 98%
Tidal marshes	1170	970(-17%)	160(-84%)	20(-88%)	minus 98%
Tidal swamps	1230	590(-52%)	0	0	minus 100%
length in feet					
Riparian shoreline	93,000	90000(-3%)	38000(-58%)	19000(-50%)	minus 80%
Development conditions					
area in acres					
Deep water	--	--	240[3]	210(-12%)[3]	
Developed shorelands and floodplain	0	1210	3750(310%)	5220(+39%)	plus 430%
length in feet					
Developed shoreline	0	4000	47000(+1175%)	53000(+12%)	plus 1430%
New developed shoreline[2]	--	21000	28000(+33%)	28000	

1 measured in historic channel area of estuary, upstream of river mouth

2 new shoreline resulting from fill in former estuarine shallows and flats

3 new deep water habitat created by dredging of former estuarine shallows and flats

Note: The present federally authorized Duwamish Waterway (extending approximately five miles south from north Harbor Island) includes approximately 6000 linear feet of vertical bulkhead, 17,500 feet of pier structures, and approximately 25,000 feet of exposed riprap shoreline. (ACOE January 1983)

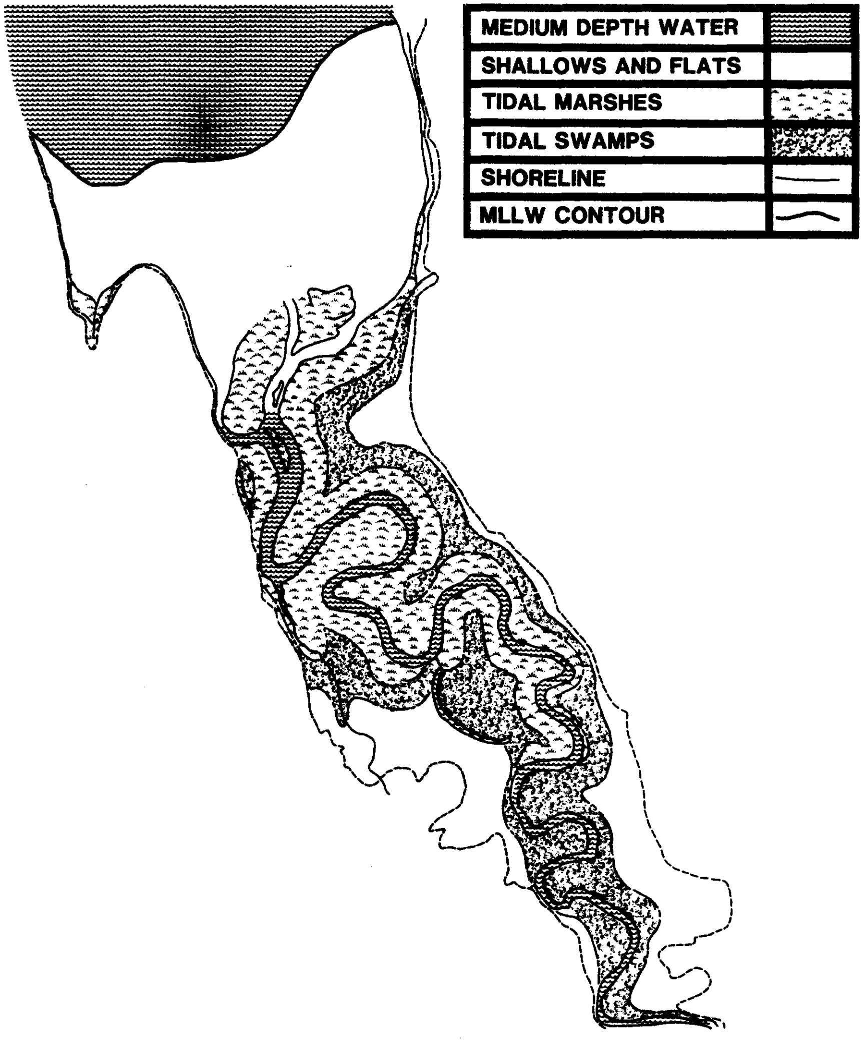

FIGURE ONE: DUWAMISH RIVER ESTUARY, 1854

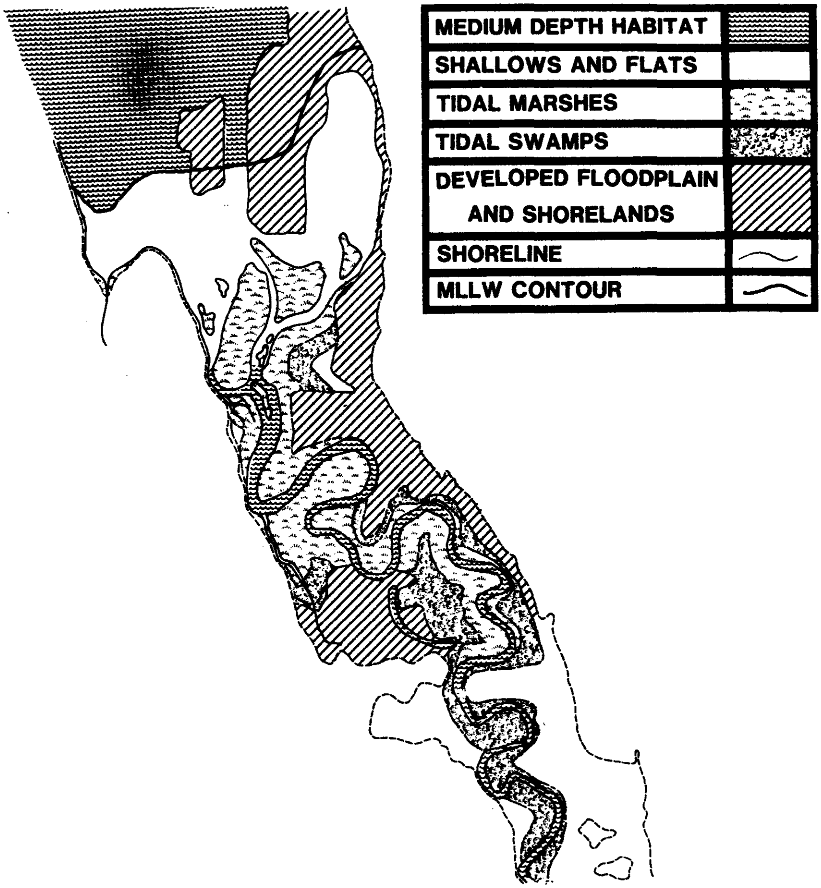

FIGURE TWO: DUWAMISH RIVER ESTUARY, 1908

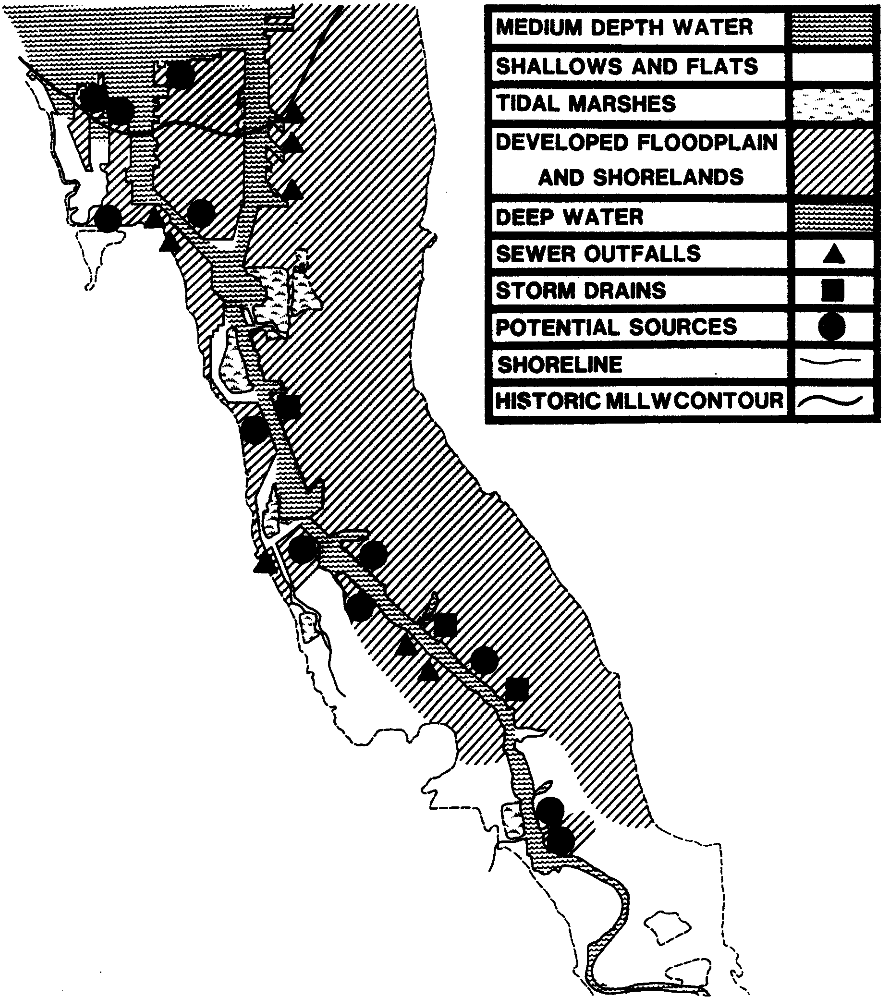

FIGURE THREE: DUWAMISH RIVER ESTUARY, 1940

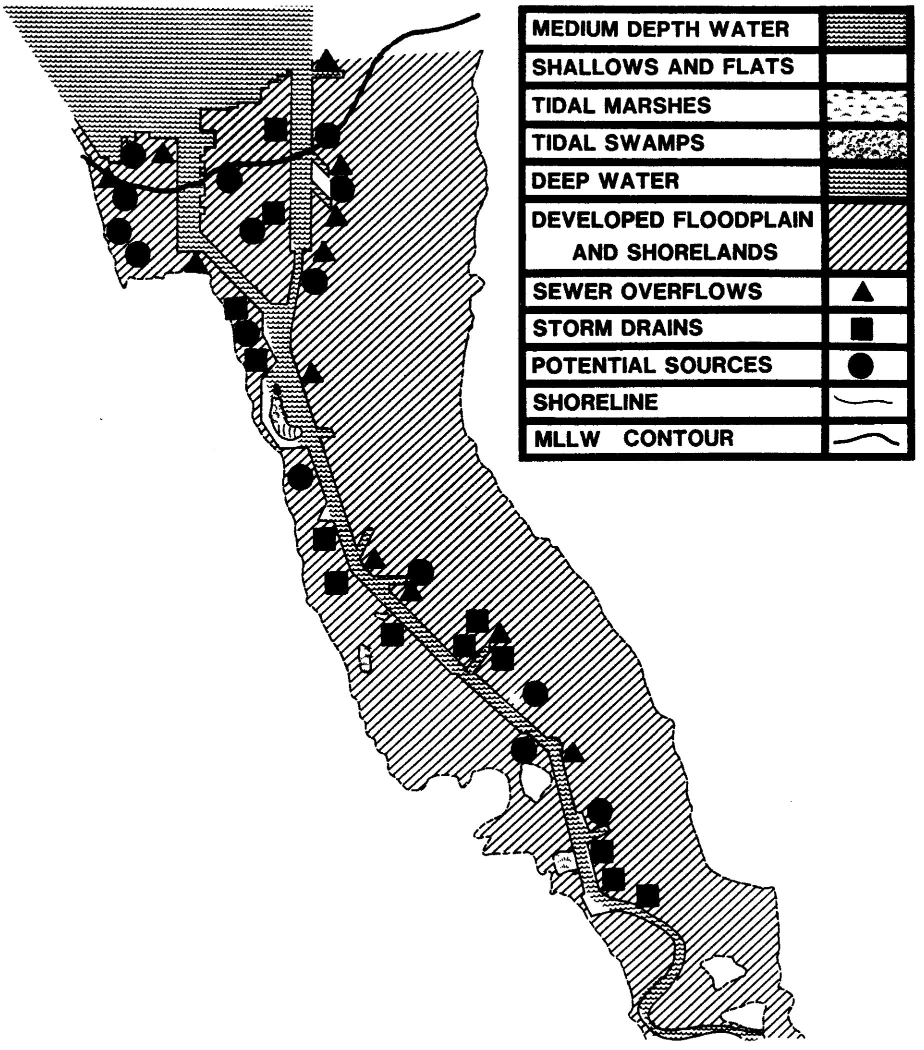

FIGURE FOUR: DUWAMISH RIVER ESTUARY, 1985

HISTORICAL CHANGES
IN THE DUWAMISH RIVER ESTUARY, 1854-1973

INCREMENTAL CHANGES

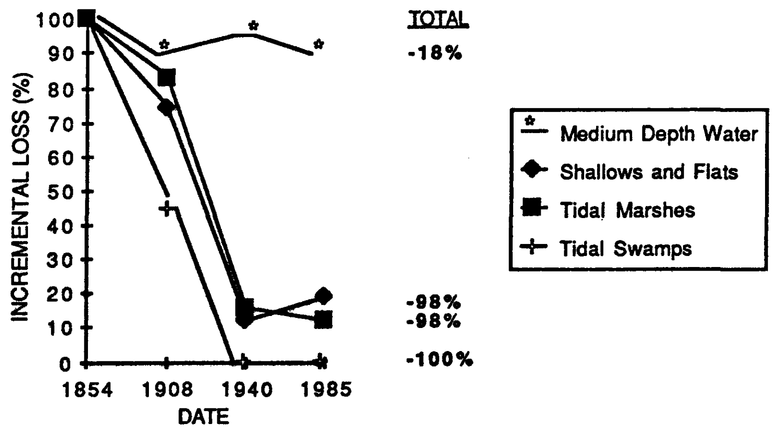

FIGURE FIVE

HISTORICAL CHANGES
IN THE DUWAMISH RIVER ESTUARY, 1854-1973

HABITAT AREAL EXTENT

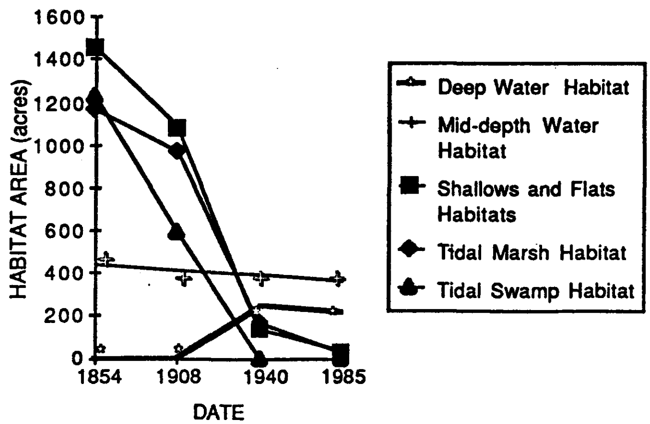

FIGURE SIX

HISTORICAL CHANGES IN THE DUWAMISH RIVER ESTUARY, 1854-1973

SHORELINE COMPOSITION

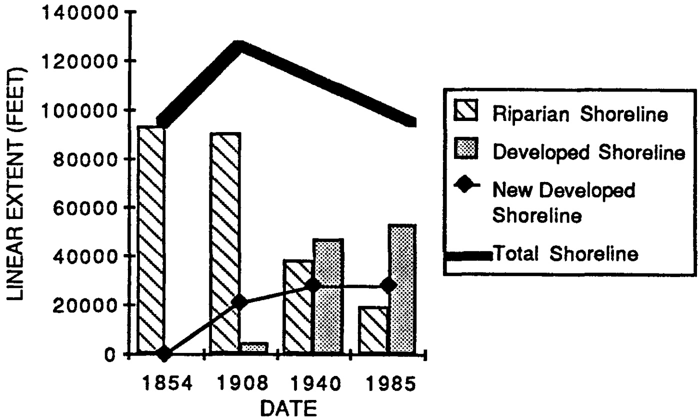

FIGURE SEVEN

HISTORICAL CHANGES IN DISCHARGE OF THE DUWAMISH RIVER 1860-1986

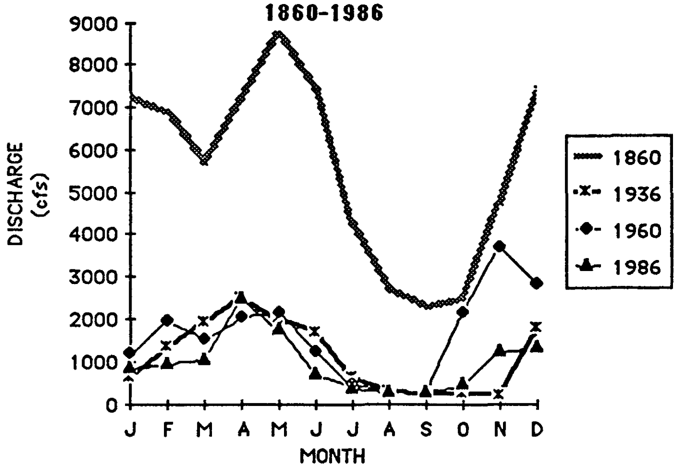

FIGURE EIGHT

HISTORIC CHANGES IN DUWAMISH RIVER WATERSHED (sq.mi.)

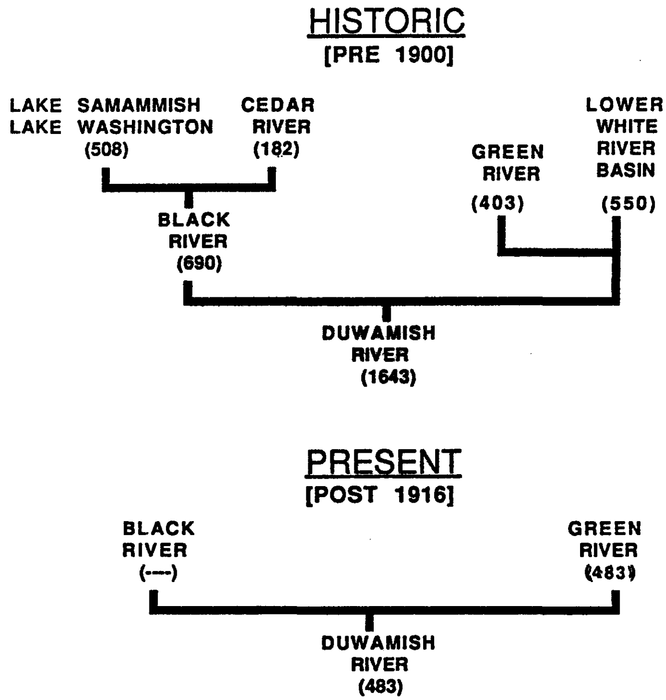

FIGURE NINE

Estuarine marsh dynamics in the Puget Trough - implications for habitat management.

Ian Hutchinson*

1. Introduction

Intertidal wetlands, especially those in deltaic settings, are highly dynamic habitats. Floods and ocean storms, shifts in the location of channel bars or barrier spits, and changes in the size and position of delta distributaries can all bring about dramatic changes in the extent and nature of coastal wetlands. We know very little about the character or rates of such change in coastal wetlands in the Pacific Northwest, yet these processes have profound consequences for wetland conservation and creation projects in the region. In this paper I plan to review what is known, and discuss how this information might be incorporated into guidelines for wetland acquisition, conservation and creation priorities. Section 2 of the paper examines the present extent and character of estuarine wetlands in the Puget Trough (geographically equivalent to Puget Sound and the Strait of Georgia); Section 3 summarizes the effects of European settlement on the area and nature of estuarine wetlands in this region; Section 4 examines the effects of delta progradation, aggradation and sea-level change on the habitat resource; and Section 5 considers the implications of this information for the management of coastal wetlands in the region.

2. Present Extent and Nature of Coastal Wetlands in the Puget Trough

A compilation of published data on the extent of estuarine marsh habitat indicates that there are at present some 75 km^2 of this habitat type in the Puget Trough. This total is almost equally divided between the Strait of Georgia and Puget Sound sub-regions (Table 1). In 1985 I began a study of geographic variation in the vegetation of these estuarine wetlands, selecting as a sample 17 of the larger deltaic wetlands that had not undergone severe disturbance. A plant community classification based upon the relative abundance of 80 vascular plant species in 905 randomly-sampled quadrats formed the basis for a typology of individual deltas (Hutchinson 1988). The typology divided the vegetation of the estuarine marshes of the Puget Trough into three categories: saline marshes (dominated by halophytes and Carex lyngbyei), brackish marshes (with a significant admixture of Scirpus species), and a fjord-head marsh (characterised by extensive monospecific stands of Carex lyngbyei) (Fig. 1A). Saline marshes are characteristic of deltas formed by (1) small rivers with winter peak flows and declining discharges during the growing season, and (2) strong wave action or tidal flushing offshore (Fig. 1B). Brackish marshes are associated with (1) the foreshores of the deltas of larger rivers with a primary or secondary discharge peak in early summer or (2) smaller rivers emptying into sheltered embayments. The confined fjord-head setting of the Squamish delta and protracted high summer flows of this river create an environment that is unique amongst the set of deltas (Fig. 1B).

Almost 80% of the extant marsh area in the Puget Trough can be categorized as brackish marsh, 18% as saline marsh, and 2% as fjord-head marsh (Fig. 2). The degree to which

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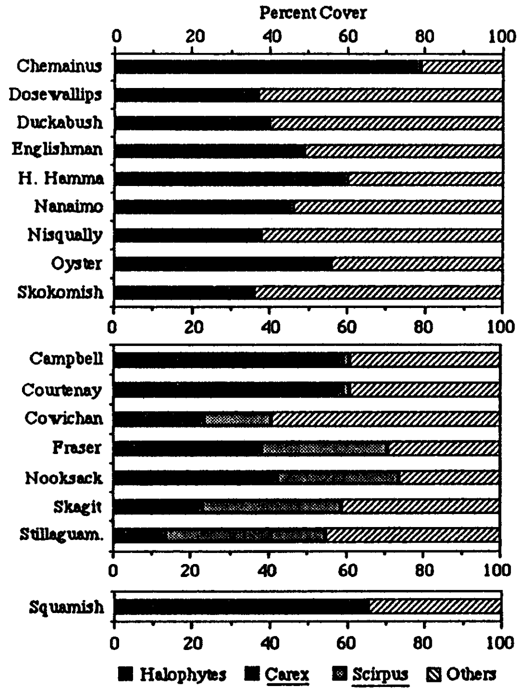
1A.



1B.

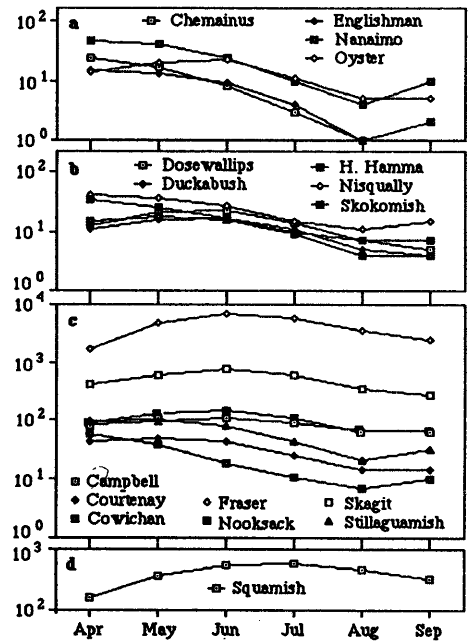


Fig. 1A. Percent abundance of dominant ecological groups and taxa of emergent plant species in (upper) saline, (middle) brackish and (lower) fjord-head estuarine marshes of the Puget Trough. Data: Hutchinson (1988).

Fig. 1B. Mean monthly discharge of contributing rivers at the mouth for (a) saline marshes in the Strait of Georgia, and (b) Puget Sound; (c) brackish marshes, and (d) fjord-head marsh areas. Units are $m^3 s^{-1}$.

these marsh types are functionally different remains unknown. Within and between-marsh type variation in: (1) total primary production; (2) volume, timing, and quality of organic detritus export to estuarine foodchains; and (3) exogenous organic matter deposition, is still primarily a matter of speculation.

Table 1. Areal extent (in ha) of intertidal marshes in the Puget Trough at the time of European settlement and at the present time. All of the marsh areas greater than 50 ha in extent at the time of European settlement are listed.

	Pre-Settlement	Present-Day	Percent Change	Source
Georgia Strait				
Baynes Sound	120	117	-2	Prentice and Boyd 1988
Burrard Inlet	150	9	-94	Environment Canada 1975
Chemainus	160	121	-24	Prentice and Boyd 1988
Courtenay	87	74	-18	Prentice and Boyd 1988
Cowichan	191	101	-47	Prentice and Boyd 1988
Englishman	54	47	-13	Prentice and Boyd 1988
Fraser	3186	2810	-12	Hutchinson et al. 1988
Nanaimo	275	130	-53	Prentice and Boyd 1988
Squamish	120	117	-2	Hutchinson et al. 1988
Other Areas	110?	110	?	Hutchinson et al. 1988
Total	4453	3636	-18	
Puget Sound				
Duwamish	260	3	-99	Bortleson et al. 1980
Lummi	580	30	-92	Bortleson et al. 1980
Puyallup	1000	0	-100	Bortleson et al. 1980
Nisqually	570	410	-28	Bortleson et al. 1980
Nooksack	450	490	+9	Bortleson et al. 1980
Samish	190	4	-98	Bortleson et al. 1980
Skagit	1600	1200	-25	Bortleson et al. 1980
Skokomish	210	140	-33	Bortleson et al. 1980
Snohomish	3900	1000	-74	Bortleson et al. 1980
Stillaguamish	300	360	+20	Bortleson et al. 1980
Other Areas	300?	250	?	Hutchinson (unpub.)
Total	9360	3887	-58	
Grand Total	13790	7523	-45	

3. Historical Extent and Nature of Coastal Wetlands in the Puget Trough

The ecological communities of the lowlands of the Puget Trough have undergone considerable disruption and disturbance in the last century. Disturbance has been particularly severe in deltaic areas, which, after dyking and draining, offered a productive base for agriculture, and, more recently, suitable sites for industrial and urban development. The intertidal location of much deltaic marsh protected it to some degree from agricultural alienation; estuarine and riparian meadows and shrublands were more likely to be turned into farmland.

Historical cartographic sources are seldom precise enough to allow reliable estimation of the extent of intertidal communities. Consequently the estimates of the areal extent of estuarine marsh in the Puget Trough prior to the time of European settlement given in Table 1 must be treated with caution. However, although the accuracy of the estimate of areal extent in the mid- to late-nineteenth century (138 km^2) may be questionable, it is apparent that a considerable loss of estuarine marsh habitat has occurred in the region since that time, and that this loss has been concentrated in the estuaries of Puget Sound.

Those marshes that have remained little disturbed since the time of European settlement have probably not changed in overall community structure since that time, although regulation of river flow may have had an effect on community composition in some instances. In those cases where intertidal marshes have been obliterated by economic development (e.g. Puyallup, Duwamish River deltas) the character of the pre-settlement marsh may be reconstructed on the basis of the discharge regime of the contributing river. For example, the mean annual hydrograph of the Puyallup River is very similar to that of the Nooksack River, and the marshes of the Puyallup River, like those of the Nooksack, are therefore considered to have had a predominantly brackish character. In contrast, the annual hydrograph of the Duwamish River is similar to that of the Nisqually River, the deltaic marshes of which are predominantly saline. A reconstruction of the areas of estuarine marsh habitat in the mid-nineteenth century indicates that saline marshes were more prominent at this time than at present; ca. 35% of the total area likely belonged to this category (Fig. 2).

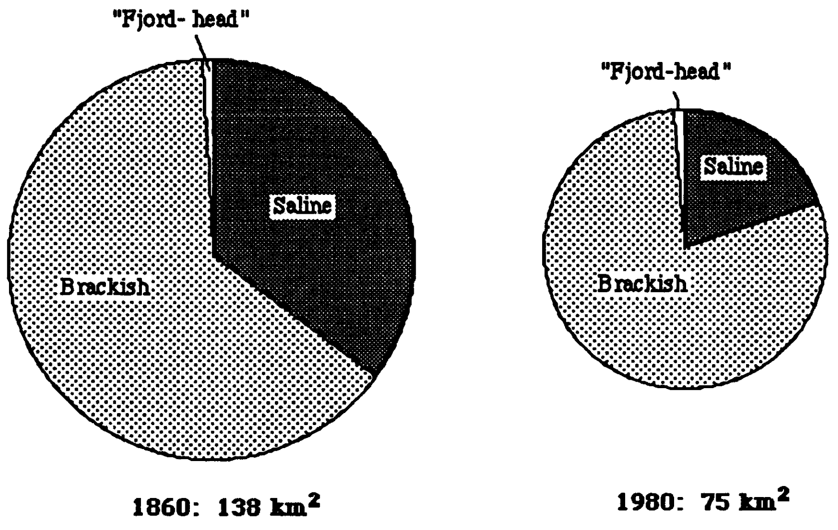


Fig. 2. Relative extent of estuarine marsh habitats in the Puget Trough, 1860 - 1980.

Most of the loss of saline marsh habitat is attributable to dyking and draining of a few extensive areas of saltmarsh on the abandoned or relatively inactive deltaic foreshores of Boundary, Lummi, Samish and Padilla Bays. In contrast, the amount of dyking and draining that took place in the small, saline marshes of active deltas was relatively slight. In the brackish delta marshes a considerable amount of alienation of wetland habitat (estuarine meadow and high marsh) took place, not only in the larger systems, but also in deltas of intermediate size (Prentice and Boyd 1988).

4. Allogenic Change

These intertidal marsh habitats developed on a coastline that was deglacierized some 10,000 years ago. Isostatic and eustatic adjustments in sea-level for the several millenia after deglaciation, combined with rapid building of deltas in the immediate post-glacial period, likely led to large-scale changes in the location and extent of estuarine wetlands in the Puget Trough during this period. Tectonic activity associated with the offshore plate boundary has produced small-scale changes in sea-level on the outer coasts of Oregon, Washington and British Columbia for at least the last 2,000 years (Atwater 1987), but in the Puget Trough sea-level has been essentially stable (Clague et al. 1982), or rising slowly (Eronen et al. 1987) for the last 5-6,000 years .

The intertidal wetlands that were present in the Puget Trough at the time of European settlement, were the product of evolutionary processes that had been operating over this period of stable sea-levels. Such wetlands, although likely in equilibrium with the physical environment of the proximate estuary, would not be in ecological stasis. Changes in the ecological communities of such marshes would be a product of the combined effects of normative processes such as the accretion of sediment on the deltaic platform, and disruptive events such as channel shifting. Both processes would produce changes in the competitive status of established and invading plants, leading to long-term allogenic successional change.

The rate of successional change in a particular estuarine environment prior to European settlement can be assessed from estimates of rates of deltaic progradation in the geological literature. Land clearing and logging produced dramatic increases in sedimentation and marsh establishment in Oregon estuaries in the late nineteenth and twentieth centuries (Johannesen 1961,1964), and it is likely that similar changes took place in at least some of the estuaries of the Puget Trough. In the Fraser River delta, for example, Williams (1986) documented a delta-front progradation rate of 1.6 m a^{-1} for the last 5000 years, which is substantially less than the rate of marsh progradation in the last 50 years (Table 2). Flow regulation and channelization of the rivers of the region in the post-settlement period may also have had substantial effects on rates of marsh advance and community structure.

Table 2. Progradation Rates of selected Puget Trough deltas in the historical period.

Delta	Data Source	Data Period	Total Advance (mean±s.e., m)	Mean Annual Rate (m)
a) Saline.				
Nanaimo	Airphotos	1932-1977	14±5	0.3
Nisqually	Maps	1878-1973	21±13	0.2
Skokomish	Maps	1884-1952	-6±17	-0.01
b) Brackish				
Fraser	Airphotos	1932-1980	266±70	5.6
Nooksack	Maps	1888-1972	1003±110	11.9
Stillaguamish	Maps	1886-1973	298±50	3.4
c) Fjord-head				
Squamish	Airphotos	1932-1964	80±26	2.5

Historical cartographic and aerial photographic evidence indicates that there has been considerable local variation in marsh progradation rate in the post-settlement period, with the rate dependent on the size and sediment supply of the contributing river (Table 2). Saline marshes are restricted to deltas in which river flow (and hence sediment output) is limited. The saline marshes in the sample built out much more slowly than their brackish counterparts. The most dramatic case of marsh progradation in the Puget Trough is represented by the Nooksack River delta. A major channel avulsion in the mid-nineteenth century diverted the Nooksack River from Lummi Bay into the northern end of Bellingham Bay, and the delta front has been building seaward at a rate of almost 12 m a^{-1} since that time.

Deltaic aggradation usually proceeds concurrently with progradation. In 1984 I placed 340 wooden stakes at random in the marshes and neighbouring mudflats of several deltas in the Strait of Georgia. The length of the stakes has been measured at regular intervals since then, and the results indicate that the rate of aggradation, as might be expected, is concordant with the size and sediment load of the local river (Fig. 3). The rate of sediment accretion on the saline marsh communities of the Nanaimo River delta is negligible (0.1 ± 0.4 mm a^{-1}). The equivalent values for the other deltas are: Cowichan estuary (2.1±0.4 mm a^{-1}), Squamish River delta (8 ± 2 mm a^{-1} cf. Pomeroy and Stockner 1976), and Fraser River delta (18 ± 2 mm a^{-1}). As far as I am aware, there are no long-term estimates of sediment accretion rates in the Puget Trough marshes which could be used to test the validity of extrapolating these short-term results over much longer periods.

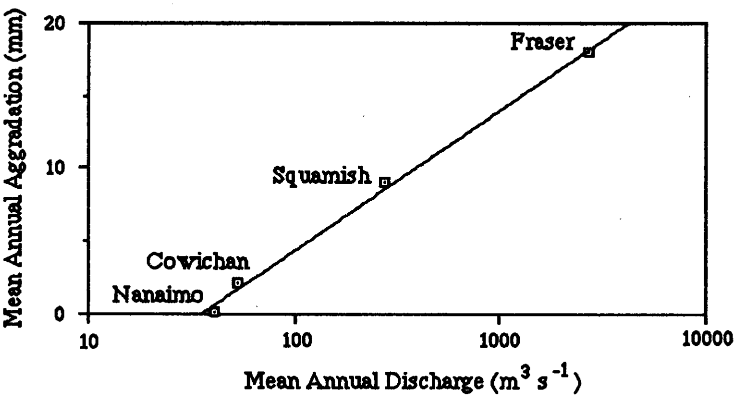


Figure 3. Mean annual rate of aggradation in estuarine marsh habitats as a function of the mean annual discharge of the contributing river.

At this rate of accretion the foreshore marshes of the Fraser River and Squamish River deltas are capable of proceeding from a pioneer community at ca. 2.5 m above chart datum to the upper marsh limit (ca. 5.0 m a.c.d.) within 150-300 years. In contrast, the low sedimentation rate on the Nanaimo River delta marshes means that the low marsh areas that are currently forming will require several millenia to aggrade to the upper marsh limit. This scenario is based on the assumption that local sea-levels are stable over this interval. A more likely scenario, however, is that global sea-levels will rise substantially in the near future (from 1 - 1.5 mm a^{-1} at present to 2 - 8 mm a^{-1} by the year 2050) as a result of CO_2-induced climatic change (Office of Energy Research 1985). Marshes with low sedimentation rates will likely be unable to keep pace with the rising sea-level, and will be destroyed by erosion. The future character of ecological communities in estuarine marshes of the region will therefore depend to a great extent on the balance between deltaic sedimentation rate and the rate of sea-level rise.

Conclusions and Recommendations

Deltas are built into receiving basins of variable morphology and wave climate by rivers that can differ substantially in terms of their flow volumes and flow regimes. The resultant estuarine intertidal environment is highly variable between deltas, and also highly dynamic. These spatial and temporal variations obviously have profound implications for estuarine marsh conservation, rehabilitation and creation projects. Rather than conclude this paper with a series of directives based upon an analysis of the historical and future trajectories of the estuarine marshes of the Puget Trough, I consider that it might be more productive to pose a series of questions to those of you who are directly concerned with developing policies for marsh habitat conservation in the region.

1) To what extent should the aim of conservation be to maintain the present balance of estuarine marsh habitat types, or to attempt to reconstruct the range and character of habitats present in the region prior to European settlement?

2) .To what extent should conservation aim to maximise the "functional characteristics" of estuarine marshes? As brackish marshes are possibly more productive, and perhaps also more supportive of estuarine foodchains than saline marshes, should conservation - rehabilitation efforts give priority to the maintenance of brackish marsh habitat?

3) Rates of natural establishment and marsh progradation are much greater in the case of brackish marshes (especially those associated with the estuaries of the larger rivers in the Puget Trough) than their saline counterparts. Should a conservation priority therefore be to protect and construct saline marsh habitat at the expense of the more 'robust' brackish marsh?

4) Brackish marsh substrates aggrade much more rapidly than do saline marshes, with concomitantly rapid changes in plant community composition. Some brackish marsh conservation areas may therefore only have a limited productive lifetime compared to that of neighbouring saline marshes. Should this influence conservation or rehabilitation policy?

5) Given the slow progradation and aggradation rates of saline marshes, the areas of this habitat type in the Puget Trough likely represent the accumulated product of several millenia of development. Should the conservation of the remnant (and perhaps relict) saline marsh areas therefore be a major priority?

6) The major losses of estuarine marsh habitat in the Puget Trough in the last 100 years are attributable to agricultural and urban-industrial development of this coastline. Major losses in the next century may be an indirect product of anthropogenic climatic change. Should we be developing policies to mitigate against future losses from this cause, and can we effectively protect estuarine marsh habitats from them if they do occur?

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Mussel Growth and Estuarine Habitat Quality[1]

Mary H. Ruckelshaus, Robert C. Wissmar
and Charles A. Simenstad

Abstract

Understanding food web dynamics can offer insights into ecosystem functioning and facilitate predictions of ecosystem responses to perturbation. This study addresses the question of how biological and physical habitat characteristics affect mussel growth in the Padilla Bay estuary in northern Puget Sound. Mussels (*Mytilus edulis*) were placed in cages in habitats dominated by different primary producers and physical characteristics. Growth rates of caged mussels were highest at the mouth of the estuary and lowest in a freshwater slough. However, concentrations of food sources (chlorophyll *a* and particulate organic carbon and nitrogen) showed an inverse relationship with growth rates. Physical stresses associated with the habitats appeared to depress growth rates despite high food concentrations. Decreases in growth rates showed a strong positive correlation with decreases in salinity, concentration of inorganic matter in the seston and submergence time. Additionally, natural abundances of stable carbon isotopes suggested that different primary food sources for mussels in different habitats may have been partly responsible for variable growth rates. If food sources and physical conditions critical to consumer performance can be determined, prioritization of estuarine habitat management goals may be facilitated.

Introduction

Wetlands perform a variety of functions that can effect habitats well beyond their own boundaries. General functions provided by wetlands include: 1) moderation of surface and groundwater flow; 2) regulation of sediment erosional and depositional processes; 3) performance of physical and chemical processes affecting water quality and nutrient cycling; and 4) generation of plant matter and provision of food and habitat for animals (Strickland,1986). Gaining a better understanding of wetland functions and how they can be measured may facilitate

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predictions of the potential impacts of perturbations to such habitats.

In this paper, we address function #4 in reporting the results of a study in which the sources and fates of organic carbon in an estuary were explored. Organic carbon produced by plants represents an important food source for estuarine organisms. Determination of the origins of carbon in consumer diets in different habitats can give us an idea of the importance of plant production in those habitats. Historical approaches to tracing the distribution and fate of organic carbon in estuaries have included calculation of carbon budgets and/or stable carbon isotope ($\delta^{13}C$) analyses (Teal, 1962; Fry and Sherr, 1984). However, the results from such analyses can be conflicting. For example, a study of the origins and fates of organic carbon in Hood Canal, Washington, concluded that although marine phytoplankton accounted for the greatest percentage of the total annual net carbon production, consumers within the system derived most of their carbon from eelgrass, epiphytes, and macroalgae (Simenstad and Wissmar, 1985). These results indicate that estimates of food quantity based on carbon budget data do not necessarily represent the ultimate availability of food to consumers. Furthermore, neither carbon budgets nor stable carbon isotope analyses address the question of the importance of food sources to consumers.

This study was an attempt to couple estimates of food origins and fates using $\delta^{13}C$ analyses with indications of food value and consumer response. Consumer response was estimated by monitoring growth of transplanted mussels. We surmised that this approach will provide the basis for identification of important consumer-autotroph linkages in wetland habitats. The results discussed here are part of a larger study designed to trace the sources and fates of organic carbon and nitrogen in the Padilla Bay food web. We addressed the following questions: 1) what are the origins, quantities, and qualities of consumer food resources in different Padilla Bay habitats?; 2) what are consumer growth rates in the different habitats?; and 3) what is the relationship between growth rates and biological and physical habitat characteristics?

Methods

Padilla Bay is a shallow (2-3 m) estuary extending 81 km^2. It receives fresh water input from the Skagit River and several small sloughs (Fig. 1). Four estuarine habitats were sampled along a decreasing salinity gradient: 1) a neritic site at the western extreme of the Bay (NR); 2) an extensive eelgrass bed near the mouth (EG); 3) a mudflat site in mid-estuary near a main tributary channel (MF); and 4) a tidally-influenced slough at the head of the estuary (SL) (Fig. 1). Tidal flats at lower elevations in the central and northern parts of the bay are characterized by dense beds of eelgrass (*Zostera marina* and *Z. japonica*) and patches of macroalgae (*Ulva* and *Enteromorpha* species) covering over 2500 hectares. Fringing salt marsh habitats characterized by *Carex lyngbyei, Salicornia virginica* and *Distichlis spicata* occur along the sloughs and at higher tidal elevations.

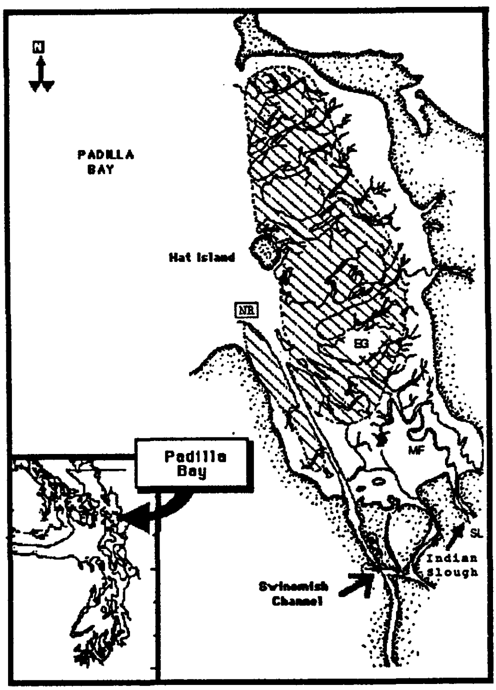

Figure 1. Map of neritic (NR), eelgrass (EG), mudflast (MF) and slough (SL) study habitats in Padilla Bay, Washington.

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The seston in each habitat was characterized by sampling: 1) total suspended particulate matter (SPM) and calculating the organic and inorganic fractions; 2) particulate organic carbon and nitrogen (POC and PON) (<64-73 µm); 3) chlorophyll *a* (chl *a*); and 4) total algal species < 63 µm (Strickland and Parsons, 1972; Simenstad and Wissmar, 1985). Autotrophic sources of carbon sampled were vascular marsh plants, estuarine and marine macrophytes, phytoplankton and epiphytic and epibenthic algae.

Stable carbon isotope analyses were performed on the autotrophs and POC sampled to estimate sources of autotrophic carbon in the POC-seston. $\delta^{13}C$ is calculated as a parts per thousand difference from a marine limestone standard (Fry and Sherr, 1984). $\delta^{13}C$ values are presented as enriched (heavier, with more ^{13}C, less negative values) or deplete (lighter, with less ^{13}C, more negative values).

The bay or blue mussel, *Mytilus edulis*, was used as a representative consumer organism because of its adaptability to a variety of environmental conditions (Bayne, 1976). Between 20 to 60 mussels were placed in cages at each of the four study sites in April, July, and December 1986 in order to monitor growth rates and $\delta^{13}C$ composition during different seasons. Consumer $\delta^{13}C$ values have been shown to be close to those of their food sources (Fry and Sherr, 1984). Because *Mytilus* is a generalist suspension feeder (Bayne 1976), its carbon isotope signature should reflect whatever carbon source is most readily available, rather than a selected food. Mussel growth rates were determined by measuring increases in shell length of at least five mussels from each cage (Incze et al, 1980).

Temperature and salinity were obtained using a salinometer. Cage elevations were determined by surveying of cage locations in reference to local tidal charts.

Results

Average mussel growth rates decreased from the neritic and eelgrass habitats towards the slough habitat during all seasons (Fig. 2).

The stable carbon isotope data indicate that mussels derived their carbon (food) from sestonic POC (Fig. 3). Within each habitat, mussel $\delta^{13}C$ values were enriched relative to POC-13 values by an average of +2.1‰.

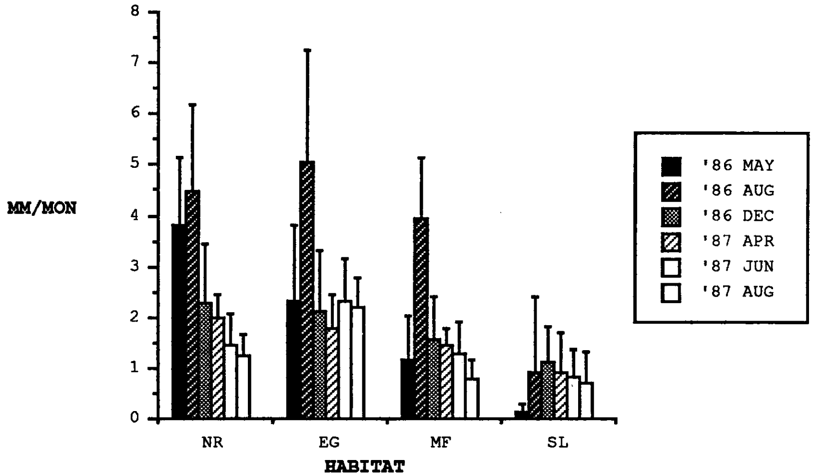

Figure 2. Monthly mussel growth rates at the neritic (NR), eelgrass (EG), mudflat (MF) and slough (SL) habitats in Padilla Bay, Washington, 1986-1987. Growth rates are based on shell length increases of at least 5 mussels per site per date; error bars = 1 standard deviation.

The POC is comprised of carbon from a number of different autotrophic sources. Evidence for the origins of sestonic POC in each habitat comes from $\delta^{13}C$ analyses, POC:PON (C:N) ratios and habitat associations of algae in the seston (Table 1). In general, mussels from the neritic (NR) site derived most of their carbon from marine phytoplankton, evidenced by $\delta^{13}C$ and C:N values characteristic of marine phytoplankton (-18 to -24°/₀₀; C:N = 6-7). The majority of algal species in seston from the NR site originated from phytoplankton. The eelgrass (EG) site POC was the most $\delta^{13}C$ enriched of any other habitat, suggesting carbon inputs from relatively enriched *Zostera*, *Ulva*, *Enteromorpha* and/or epiphytic and epibenthic algae in addition to marine phytoplankton. The C:N values ranged from close to those of phytoplankton (6-7) to those more typical of detrital material (>10). The eelgrass sestonic algae had a greater percentage of benthos-associated species than the NR site seston.

The mudflat habitat (MF) mussels appeared to have a mixture of carbon sources in their sestonic food, similar to the EG habitat, but excluding *Zostera* and epiphytes.

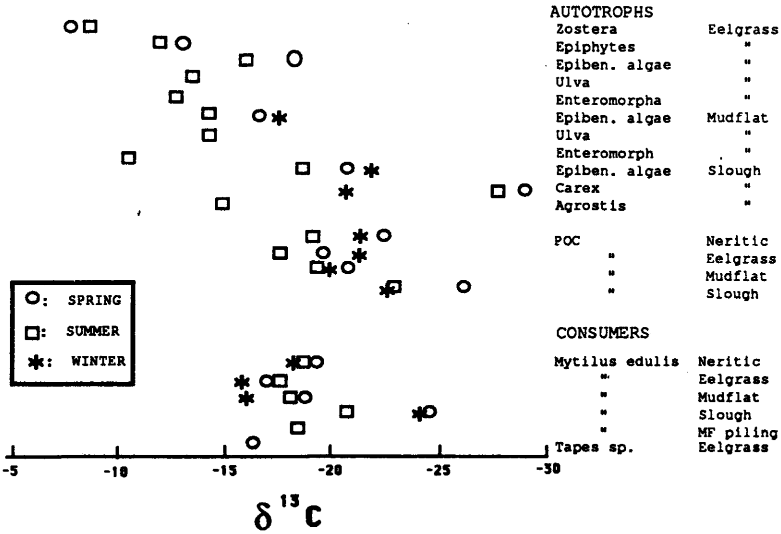

Figure 3. Stable carbon isotope composition of autotrophs, POC and consumers in neritic, eelgrass, mudflat and slough habitats in Padilla Bay, Washington, 1986-1987.

Resuspension of epibenthic flora and detrital material was suggested by higher C:N ratios and a greater percentage of benthic algal species in the seston. Mussels in the slough (SL) habitat consumed POC which was the most highly $\delta^{13}C$ deplete, indicating the presence of marsh macrophyte and/or terrestrial carbon sources in addition to marine phytoplankton in the seston. Again, the elevated C:N ratios and high percentage of benthic algal species in the

Table 1. Summary of seston characteristics in Padilla Bay, WA, 1986-1987. del ^{13}C = POC $\delta^{13}C$ values; C:N = particulate organic carbon: particulate organic nitrogen; habitat associations of algal species were determined from the literature.

			Algal habitat associations (%)	
HABITAT	del ^{13}C	C:N	PLANKTONIC	BENTHIC
NERITIC	-19 to -23	7 to 11	60	40
EELGRASS	-18 to -22	6 to 14	35	65
MUDFLAT	-19 to -21	6 to 14	20	80
SLOUGH	-24 to -26	6 to 29	20	80

seston indicated that resuspension processes may have been important in determining the contributions of carbon to the POC-seston pool in the slough habitat.

Food quantities increased landward. The concentrations of POC (430-1600 $\mu g\ l^{-1}$), PON (60-170 $\mu g\ l^{-1}$)and chl *a* (1.8-3.7 $\mu g\ l^{-1}$) generally increased from the NR towards the SL habitat. On the contrary, food quality indicators (the percentage of organic matter, chl *a* and POC in the seston) decreased from the neritic (NR) towards the mudflat (MF) habitat (Table 2). Seston quality was most variable in the slough (SL) habitat; the percentage of organic matter in the seston was the highest of any of the four habitats, and the percentage of chl *a* in the seston ranged from the greatest in the summer to the lowest in the spring bay-wide.

Physical habitat characteristics in the Bay indicated increasing stress to mussels along the habitat gradient. Temperatures and salinities exceeded physiologically stressful levels in the mudflat and slough habitats (>20°C and <15°/₀₀). The concentration of inorganic matter in the seston, which can dilute available food and/or over-load the filtering apparatus of mussels, increased from the neritic towards the mudflat and slough habitats (8-55 $mg\ l^{-1}$). Submergence time of the cages also decreased along the same gradient, resulting in decreased feeding time and increased physiological stresses associated with aerial exposure (neritic and eelgrass = 23 hours day^{-1}; mudflat = 19; and slough = 17 hrs day^{-1}).

Table 2. Seston quality indicators from Padilla Bay, WA. 1986-1987. AFDW = percent organic matter in the seston; POC:SPM = percent particulate organic carbon in the seston; and chl *a*:SPM = percent chlorophyll *a* in the seston.

HABITAT	AFDW (%)	POC:SPM (%)	CHL a:SPM (%)
NERITIC	17.1-30.0	5.0-8.9	0.16-0.41
EELGRASS	16.1-22.9	4.5-9.5	0.21
MUDFLAT	12.6-20.1	2.8-3.6	0.10-0.14
SLOUGH	18.9-24.0	4.1-7.9	0.05-0.22

Discussion

In general, the $\delta^{13}C$ composition and growth rates of mussels were poorly correlated with the isotopic composition and quantities of primary producers in their habitats. The origins of sestonic food in Padilla Bay varied across the four study habitats and during different seasons. Neritic phytoplankton were an important component to the seston in all Padilla Bay habitats. However, mussels in the eelgrass, mudflat and slough sites consumed seston which appeared to be supplemented with carbon sources from autotrophs within their habitats, especially in spring and summer. Yet, despite high *Zostera* and *Carex* productivities in the EG and SL habitats, carbon derived from those macrophytes was not central to mussel diets, probably due to its refractory nature. These results are contrary to studies in southeastern U.S. estuaries, in which food webs were hypothesized to be based primarily on carbon from seagrasses, marsh macrophytes and their detritus (Teal, 1962; Fry and Parker, 1979).

$\delta^{13}C$ evidence for consumption of a food source does not indicate its importance. Habitats with the highest food concentrations did not have highest mussel growth rates. Food quality indicators also did not adequately explain the growth rates observed in the Bay. It is apparent that we also have to consider the effects of physical habitat characteristics on mussel growth rates. For example, decreased food intake as a result of aerial exposure, in addition to stresses associated with low salinity, high temperatures and/or high concentrations of particulate inorganic matter, may override the effects of high seston qualities and quantities measured in the slough. When mussel growth rates were standardized to the submergence regimes at the neritic and eelgrass sites, calculated growth rates in the mudflat and slough habitats increased slightly. This relationship suggests both direct and indirect effects of physical habitat characteristics on food availability and mussel physiology, which influence suspension feeder growth rates.

Furthermore, it is important to recognize that mussel growth rates are integrated responses to habitat quality. Our methods of sampling food quantity, quality and physical characteristics represented instantaneous, "snapshot" views of habitat characteristics the mussels experience. Habitat characteristics inferred from periodic and spatially patchy sampling in a highly variable environment such as Padilla Bay must be interpreted with caution. Both seston characteristics and mussel responses demonstrated high temporal and spatial variability during the study.

In order to be able to use mussel growth rates as indicators of habitat quality and estuarine functioning, we need to get a better idea of how the effects of habitat characteristics on mussel growth change seasonally and spatially. For example, integrative measures of water flow and food quality and quantity would allow us to estimate the actual rates of food delivery to the mussels over a given period of time.

Recommendations

Greater than half of the wetland habitats in Puget Sound have been destroyed by human activity (PSWQA Management Plan 1987). Continued loss and degradation of these habitats will result in loss of important functions such as those described in this paper. The generation of organic matter as food for estuarine consumers is crucial to maintenance of food web health and diversity. Padilla Bay mussel diets were comprised of organic carbon from terrestrial, salt marsh, estuarine, and neritic habitats, the relative contributions from which varied seasonally and spatially. Thus, the timing and location of perturbations in the estuary will affect the magnitude of the resultant impact, and cumulative effects of such impacts. For example, if alterations of estuarine morphology or bathymetry create a change in hydrodynamic resuspension processes in the Bay, the availability of detrital and epibenthic algal carbon sources that are important to mussel diets when phytoplankton biomass is low could be reduced. In order to more clearly elucidate the origins and composition of seston in estuarine habitats, we suggest a multiple isotope approach. Analyses of stable isotopes of nitrogen and sulfur can be used to separate nitrogen-fixing vs. nitrate using autotrophs and benthic vs. planktonic producers (Peterson et al, 1985). Also, clarification of the origins of carbon in mussel foods could be facilitated by monitoring $\delta^{13}C$ of consumers with different feeding modes (e.g., deposit and suspension feeders in the same habitat).

The approach described here could also be useful in habitats beyond the littoral zone. The relative contribution of nearshore productivity to food webs in deeper parts of Puget Sound is generally unknown, but could be estimated using multiple isotope analyses of the biota. A greater understanding of the effects of changes in wetland functioning on other habitats in Puget Sound may give arguments for preservation and restoration of wetland habitats needed ammunition.

Acknowledgments

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Native Freshwater Wetlands:
Rare and Threatened Ecosystems?

Linda M. Kunze[1]

Introduction

The purpose of the Natural Heritage Program is to collect and maintain data on rare, endangered and sensitive plant species, and native ecosystems, including wetlands.

These data are used in several ways, including setting priorities for protection of the different "elements" (species or ecosystems) identified as important, and the acquisition and management of sites containing these elements as Natural Area Preserves.

At its beginning, the Program lacked information on native wetland composition, rarity, threats, and locations of high quality examples. After compiling existing information from the literature, the Program began an inventory effort with the intent of answering the following questions:

1) What different kinds of native wetlands exist in Washington, and what are their component plant communities?

2) How many high quality examples of each kind of wetland still exist?

3) What are the threats to the continued existence of these systems?

4) What is the priority for protection of each kind of wetland?

5) How can examples of these wetlands be protected?

A primary inventory and analysis effort for freshwater wetlands in the Puget Trough region was conducted between 1983 and 1987. The inventory was funded by grants from The Nature Conservancy, and federal Coastal Zone Management funds administered by the Washington Department of Ecology (Contract Numbers C0086077 and C0087090).

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Methods

The inventory was restricted to the lowlands (below 610 meters) of the Puget Trough region (Whatcom, Skagit, Snohomish, King, Pierce, Thurston, Mason, Jefferson, Kitsap, Island and San Juan counties).

Aerial photographs (scale of 1:12,000) were reviewed to identify possible field survey sites. The search image was intact, impounded, wetland systems (or in some cases, large fragments) having upland buffers, and no apparent human alteration (dikes, ditches, fill, dredging, mining or structures).

224 low elevation, freshwater wetland sites were identified for potential field survey. Of these, 170 were surveyed in the field.

Field surveys were conducted from June to October during 1983, 1984, 1985 and 1986. Surveys concentrated on wetland vegetation, collecting relevé data on plant communities, and plant species cover values. Data were also collected on physical site characteristics (hydrology, soils, physical setting, and past and present uses or disturbance).

All data and information were entered into the Natural Heritage Database.[2]

Results

The 170 surveyed sites were placed into 3 categories:

1) First Tier Sites: the most pristine examples of native wetland systems. They had no evidence of human-caused topographic or hydrologic alteration. Exotic (non-native) plant species occurred infrequently, if at all. There was relatively little apparent human-caused disturbance of the native vegetation. Adequate buffers existed to protect the site from adjacent land uses. The site had no known major water quality problems.

2) Second Tier Sites: those sites which were disturbed but have good potential for restoration. There was no (or isolated) human alteration of the wetland topography. There was no human-caused alteration of the wetland's hydrology, or the wetland appeared to have recovered from the alteration. There were low cover and frequency of exotic plant species. There was relatively little human-related disturbance of the native vegetation, or excellent recovery from past disturbance. If the wetland system was

[2] The Natural Heritage Database is part of the Washington Natural Heritage Program, Division of Land and Water Conservation, Department of Natural Resources, Olympia WA, 98504. Some data within the system are considered publicity sensitive and may be withheld.

degraded, it still contained a viable and high quality example of a wetland community. The site had no known major water quality problems.

3) Those sites too altered to be considered further.

A preliminary classification of low elevation, freshwater, impounded, wetland plant communities was developed based on the relevé data collected.[3] 28 plant communities were identified: 8 sphagnum bog communities and 20 non-sphagnum communities.

Table 1 summarizes the results of the inventory effort.

TABLE 1. Results of freshwater wetland survey for the Puget Trough lowlands.

	Sphagnum*	Non-Sphagnum	Mosaic	Total
Wetlands surveyed	34	85	51	170
First Tier Sites	3	5	8	16
Second Tier Sites	9	5	22	36

*Sphagnum = sphagnum bog wetlands. Non-sphagnum = non-sphagnum wetlands. Mosaic = sites which are a mosaic of sphagnum and non-sphagnum communities.

Several threats to the continued health and function of wetlands were identified through the study. They include: filling, diking, dredging, conversion to other uses, pollution, excessive human use, introduction and spread of exotic plant and animal species, peat mining, destruction of associated habitat, alteration of nutrient levels, alteration of water chemistry or temperature, alteration of water levels (total as well as normal minimum and maximum levels), logging, roads, biocides, powerline and gasline rights-of-way, siltation, livestock use and pets. Most of these threats are associated with industry and human habitation, but many are related to resource management and recreation.

[3] Copies are available from the author.

Discussion

The 170 sites surveyed represent less than 1/10th of the existing wetlands in the Puget Trough lowlands. Of the sites surveyed, only 16 appear to be nearly pristine (First Tier). Recovery may be possible for an additional 36 sites (Second Tier). This suggests that less than 1% (0.9%) of the freshwater wetlands which still exist in the Puget Trough lowlands are nearly pristine. An additional 2% (2.1%) may be recoverable.

There has been a change in the types and permanence of human-caused alterations to wetlands in the past 150 years. While early alterations were mechanical or hydrological (logging, filling, draining, diking, etc.). more recent changes also include chemical (nutrient, biocide, toxicant) and biological (exotic species invasion) alterations. These recent alterations are pervasive and may prove more difficult to mitigate than earlier disturbances.

Non-sphagnum wetlands appear to be more threatened than sphagnum bogs. Sphagnum bogs may have natural buffers to some kinds of disturbance. Floating bogs seem less affected by changes in water level, being able to rise and fall with the water surface. They also seem less conducive to the invasion of most exotic plant species. There is also less human use of bogs than non-sphagnum wetlands. Agricultural, recreational, housing and industrial uses are concentrated around non-sphagnum wetlands and lakes. The rate of alteration is higher for non-sphagnum wetlands than sphagnum wetlands at this time, but economic pressures (i.e., peat extraction) may change this in the future.

The results of the wetland inventory have applications to the study and protection of Puget Sound. Puget Sound is part of a larger landscape and is intimately connected hydrologically with that landscape. The goal of having a healthy, functional Puget Sound is remote if the streams, lakes, and wetlands within the basin are not healthy and functioning. This study shows the need to expand our vision to include the whole Puget Trough landscape.

It is evident that pristine native wetlands are rare in the Puget Trough region. Many wetlands have already been degraded and destroyed and the activities that caused their loss continue today. Unless these destructive activities cease, the remaining native wetlands are in danger of being lost.

Recommendations

As many of the First and Second Tier wetlands as possible should be protected in perpetuity. Native, relatively undisturbed wetlands, which have other values (e.g., wildlife values) not addressed in this study, should also be identified and protected.

An intensive and extensive research effort is needed on wetland functions, values and processes, which will assist in landscape planning and management. There is an urgent need for research related to preservation of freshwater wetland systems. Topics which need to be studied include the effects of: 1) high nutrient levels; 2) unusually high and low water levels; 3) increased isolation of individual sites; 4) reduced species populations and their distribution; 5) exotic species of plants and animals; 6) biocides and other toxicants; and 7) the amount and kinds of human uses which wetlands can absorb.

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Marine Plants on State-Owned Aquatic Lands: Their Status and Management

Thomas F. Mumford, Jr.[1]

Abstract

The marine plants of Washington includes over 600 species of seaweed and eelgrass. Eelgrass beds occupy over 165,000 acres of bedlands, while floating kelp beds stretch along 414 miles or 25% of Puget Sound and the Straits of Juan de Fuca. It is acknowledged that these provide critical habitat for many important species of fish and shellfish. The majority of these resources occur in state-owned lands and are therefore managed by the Department of Natural Resources. The status of the quantity and quality of many of these marine plant resources is uncertain. Existing inventories lack sufficient resolution and are out-of-date. Information about extent or functions of subtidal eelgrass beds is almost non-existent. The department's new Aquatic Plant Resource Management Plan (APRMP) focuses on five areas: preservation, habitat protection, habitat restoration, harvest management, and culture. Inventory of these resources is of immediate concern in all these areas, in terms of both quantity (extent, amount) and quality(functions). This paper presents a plan for an initial inventory effort of eelgrass and kelp beds, and protocol development using multispectral scanning remote sensing, aerial photography, and ground truthing. These efforts will be in six areas of the state.

Introduction

Extent of marine plant resources

Eelgrass. Eelgrass beds are considered to be an important marine fish and wildlife habitat (Phillips, 1984). Eelgrass beds front between 50-75% of the shore of Puget Sound and the two outer coast estuaries of Washington- Grays Harbor and Willapa Bay (Puget Sound Atlas, 1987). The beds cover at least 165,000 acres or 11% of the total area of these

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regions. Approximately two thirds of these eelgrass beds are on state-owned bedlands (below ELLW) and are very poorly mapped. Only 40,200 acres are mapped in the Coastal Zone Atlas (DoE, 1979) or 25% of the approximate total.

Kelp. Floating kelp beds, consisting of the two Phaeophycean species *Nereocystis luetkeana* (bull kelp)(see Mumford, 1987 for a complete bibliography) and *Macrocystis integrifolia* (giant kelp)(see Mumford, in prep. for a complete bibliography) are considered to be critical habitat for most commercially important fish. These beds are almost exclusively on state-owned bedlands. They front over 414 miles of Puget Sound, approximately a fourth of the total shoreline (Puget Sound Atlas, 1987). Most of the San Juan Islands are fronted by bull kelp beds. The only complete survey to assess standing crop in Washington, Rigg (1912, 1915) estimated that there was 354,500 wet metric tons of kelp in Washington from Cape Flattery to Olympia.

Other marine plants. There are over 600 other species of marine plants found in Washington (Scagel, et al., 1986). Nearly all shallow subtidal areas support marine plants (Thom, 1978).

Importance of resource

The importance of these plants in the ecosystem fall into four major areas.

Productivity: Marine plants provide material to the food web in four ways- 1) directly while the material is still attached, 2) indirectly by providing detritus that fall to the bottom and is consumed, 3) by producing dissolved organic matter (DOM) that is food for many microorganisms, and 4) providing an important substrate for epiphytic macroalgae and diatoms (Duggins, in press).

> "...the parameter which may ultimately distinguish their role here.... is benthic primary productivity. The ultimate form of a substantial quantity of the organic matter produced by kelps is detritus and dissolved organic matter..... In effect, kelp productivity may be a major input supporting nearshore secondary productivity, Furthermore, the effectiveness of kelp beds as habitat, nursery areas, and protective cover from pelagic predators depends in large part upon the amount of kelp biomass produced." (Duggins, 1980)

Habitat. *Nereocystis luetkeana, M. integrifolia* and eelgrass provide a significant habitat for a number of organisms. The beds provide a place of refuge, and a substrate for reproduction (Foster, 1982; Phillips, 1984).

The canopy formed floating kelp during the summer and fall shades the plants below, thereby influencing the amounts and kinds of plants that co-exist in the kelp beds (Foster, 1982).

A study on *Nereocystis luetkeana* (Leaman, 1976) performed in Barkeley Sound, Vancouver I., B.C. found:

> "The plants create a habitat wherein diversity and abundance of fish species increases over non-kelp areas. The majority of resident kelp bed fish are not directly related to the plants. Indirect relationships may involve feeding, shelter and reproduction. Experimental canopy harvest of the outer segments of the kelp bed decreased the abundance of neritic fish without affecting the benthic fish, whereas harvest of the middle and inner segments of the kelp canopy decreased benthic fish abundance and diversity but increased these features of the neritic fish populations. Resident kelp bed fish will be affected to the least degree if kelp is harvested by patches from the middle segments of the kelp beds rather than from the more accessible seaward segments.
>
> None of the resident kelp bed fish are of direct commercial importance, however, large numbers of associated and transient fish, notably juvenile herring and salmon, are found around the periphery of the beds. The effects kelp harvesting may have on these commercial species should be determined in order to establish a sound kelp harvesting policy."

Hydrodynamics: Hydrodynamic effects can be broken into those with physical and biological ramifications. Kelp beds absorb wave energy and dampen wave action shoreward of the bed. Wave action influences beach slope and stability, and beach material makeup and therefore removal of the kelp and the resultant wave dampening may change the beach makeup and the types or numbers of organisms that use the beach material (Foster and Scheil, 1985).

Kelp plants act as active transporters of rock material (Emery and Tschudy, 1941). Young sporophytes begin growth on any rock surface in size from sand grains up to boulders. When the plant reaches the size at which the hydrodynamic drag of the plant can move the rock substrate, the plant/rock may be moved into deeper water, onto the shore, or along the shore. Significant amounts and sizes and rocks up to one foot in diameter can be moved in this manner. Reduction in the number of plants or plant size will reduce this material transport.

Duggins (in press) hypothesizes a hydrodynamic model for the kelp bed itself, in which the surface area of the kelp bed act to reduce longshore current flow and water turbulence. This reduction influences particulate fluxes and sedimentation rates within and near the bed. The biological ramifications of this reduction bear on feeding rates of suspension feeders, and larval and planktonic dispersal and recruitment. Most animals in kelp beds are suspension feeders and most have passive planktonic larval reproduction. The effects on feeding and reproduction may be high. Duggins and Eckman are presently studying these hydrodynamic effects in a *Laminaria* bed, a bottom-oriented kelp bed.

Eelgrass beds provide an important function of sediment trapping and substrate stabilization (Phillips, 1984).

Exploitive. Seaweeds are the source of human food, fodder, fertilizer, valuable extracted chemicals and may be used as a source of biomass for energy production (Chapman and Chapman, 1980). In Washington, the primary use of seaweeds has been for human food. Harvesting of kelp for food by Asian-Americans is becoming widespread. Bull kelp is a particularly sought-after species. A recent application for an eventual harvest of 15,000 wet metric tons per year of bull kelp was rejected by the Department of Natural Resources on the grounds that insufficient information is known about the environmental effects of harvest. *Macrocystis* is presently being harvested in one test project of quantities of 2-5 tons to be used the substrate for herring-roe-on-kelp or "kazonoku-konbu." It is assumed that harvest pressure will increase substantially in the next five years

Impacts on resource

A variety of pollutant can impact marine plants. Eutrophication, while perhaps initially increasing plant growth, increases phytoplankton standing crop, thereby reducing light penetration, and hence reducing the depth of the compensation point. Higher nutrient levels may cause a higher epiphyte load in eelgrass beds and reduce eelgrass production. Silt causes a number of problems- reduction of light in the turbid water column, coating plants reducing gas and nutrient exchange and light penetration, and smothering of microscopic spores, germlings or life-history phases. A number of organic and heavy metals are toxic. These impacts are poorly documented in Puget Sound. Dredging and trawling are great threats to eelgrass (Phillips, 1984)

Historical information

Information regarding the extent, density, and standing crop of marine plants in Washington is sketchy, incomplete, and in some cases inaccurate. In general, information about intertidal regions is more complete than that for the subtidal. Rigg's maps (1912, 1915) show floating kelp bed extent and rough density estimates. These surveys were made from small boats. The Coastal Zone Atlas (DoE, 1978-79), and the derived maps in the Puget Sound Atlas (1987) map kelp (no distinction between floating vs submerged beds or species) and eelgrass beds (no distinction of species). These maps are made from aerial photo interpretation and coverage stops at some point in the shallow subtidal depending upon tidal height when the photograph was taken and water transparency. Thom (1985) has made a preliminary study of the the changes in these kelp beds. A great deal of unanalyzed data exists in the form of WDF herring row grapples (M.L. Mills, pers. comm.), diving surveys (R. Phillips, pers. comm.) and extensive aerial photography. Unfortunately, most historical aerial photos are in black and white or color and not necessarily taken at low tide. Surveillance photos taken by the U.S. Army Corps of Engineers in color infrared are useful but were not necessarily taken at low tides. A recent effort by Webber et al. (in prep) has used Landsat satellite imagery to map the eelgrass beds of Padilla Bay and appears to have good usefulness in areas of large eelgrass meadows. As discussed below, one of the tasks of the proposal is to gather this information and assess its usefulness, and to map small areas if appropriate.

In general, marine plant habitats in Washington are extensive, poorly mapped, and considered to be critical habitat for many marine species as well as being of considerable commercial importance in their own right.

DNR's Aquatic Plant Resource Management Plan (APRMP)

Ownership

Upon statehood, the state of Washington assumed ownership from the federal government of bedlands and shorelands of tidal waters (intertidal) and beds and shorelands of freshwater areas. This includes approximately 2 million acres of bedlands, and 2,960 miles of shoreline. Since statehood, about 60% of the marine shorelines had been sold to private holdings, but approximately 1,400 miles of shorelands and all submerged bedlands remain in state ownership.

Aquatic Plant Resources

As proprietor of these state-owned aquatic lands, the Department of Natural Resources manages a significant portion of Washington's aquatic plant resources. These include all subtidal aquatic plant communities (kelp, eelgrass, seaweeds), over forty percent of intertidal plant communities[2] (seaweeds, eelgrass), and freshwater plants living on the beds and shorelands of navigable rivers and lakes. These plants are important both environmentally and economically. They are habitat for many commercially and recreational important species, form the base of food webs, and may be directly exploited for food, chemicals, and fertilizer as well as having other functions.

Management

The Department is charged with managing state-owned aquatic lands for a balance of public benefits (RCW 79.90.450). These benefits include fostering water-dependent uses, ensuring environmental protection, encouraging direct public use and access promoting production on a continuing basis of renewable resources, allowing suitable state aquatic lands to be used for mineral and material production; and generating income from use of aquatic lands in a manner consistent with the above.

Further direction for management of these resources comes through RCW 79.90 in which the department is directed to "consider the natural values of state-owned aquatic lands as wildlife habitat, natural area preserves, representative ecosystem or spawning areas prior to issuing any initial lease or authorizing any change in use."

The department is also directed to "foster.....the commercial and recreational use of the aquatic environment... and may develop and improve production and harvesting of seaweeds and sea life attached to or growing on aquatic lands or contained in aquaculture contains..." (RCW 79.68.080)

Implementation

Management implementation will come through direct management of state-owned aquatic lands and through coordination and cooperation

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2 Most of the state-owned intertidal areas are along the Straits of Juan de Fuca and the outer coast, areas supporting the vast majority of intertidal seaweeds. Puget Sound, where the majority of the shorelands have been sold, contains few intertidal seaweeds in comparison.

with other proprietary and regulatory programs. There are numerous local, state, and federal agency programs, as well as privately funded programs which may have overlapping directions and hence will affect use of aquatic plants. These include the Shoreline Management Act, Departments of Fisheries and Wildlife Hydraulics Project Approval, DNR Natural Heritage Program, Puget Sound Water Quality Authority and its Management Plan, Corps of Engineers Sections 10 and 404 programs, public upland landowners, such as DNR and Wildlife, and private groups such as the Nature Conservancy. All these programs have an important role to play and the department will cooperate with these entities to avoid duplication of efforts and ensure efficiencies.

<u>Opportunities</u>

Hence, the presence of these plants occurring on state-owned aquatic lands presents at least five management opportunities-

<u>Preservation.</u> Identify and affect protection for unique aquatic plant species, habitats, and communities and an adequate number of representative areas within defined geographic regions on state-owned aquatic lands.

<u>Habitat protection</u>. Strive to maintain or enhance quality and quantity of aquatic plant habitat.

<u>Habitat restoration</u>. Strive to take physical actions in order to restore plant habitats to their original function(s).

<u>Harvest management</u>. Identify specific species and areas for aquatic plant harvest while maintaining habitat functions including productivity, consistent with the balance of public benefits.

<u>Culturing</u>. Develop and improve production of seaweeds to produce food, and to reduce harvesting pressure on natural beds and to replace aquatic plants lost through development or pollution.

It is appropriate then that the department undertake development of a comprehensive program for management of aquatic plants on state-owned aquatic lands.

<u>Objectives</u>

Aquatic plant management is being approached from both from long-term and a short-term perspectives. The proposed long-term plan outlined in this section details issues, objectives, planning and implementation activities necessary for a comprehensive program. However, it is recognized the complete program will require a number of years to develop and considerably more funding than is currently available. The short term perspective is useful in that it assists in understanding relationships between the different components and in designing basic studies which will be useful in several applications.

The short term objective is to coordinate together resources from a number of interagency programs to meet the most urgent aquatic plant management needs. These programs include the Puget Sound Water Quality Management Plan Wetlands planning (W-1, W-2, W-6), Coastal Zone Management Wetlands funding, EPA's Puget Sound Estuarine Program, and others, as available. The full range of possible related programs is shown on Table 1. The proposed short term planning program is detailed below.

Scope

This aquatic plant management plan will guide management by the department of aquatic plants and the specific lands to which they are attached. These are state-owned tidelands, shorelands, harbor areas and beds of navigable waters[3]. Aquatic plant communities of special interest include subtidal and intertidal kelp and eelgrass and freshwater macrophytes. the plan will relate aquatic plant management to other DNR aquatic land policies and programs mentioned above. It also relates to a number of fisheries and marine mammals habitat concerns in other state and federal agencies.

Wetlands

While all aquatic plant communities are considered wetlands, the definition of wetlands also include non-vegetated aquatic lands. Therefore, this plan is not a "wetlands" management plan *per se*. Management of non-vegetated state-owned aquatic lands (near-shore and open-water) is guided by other DNR aquatic lands policies and plans such as water-dependency criteria, environmental protection, geoduck harvest program, and open- water dredged material disposal program.

Priorities

Due to the importance and predominance of state ownership of marine plant communities, marine plants will be addressed earlier than

[3] Beds of navigable waters include submerged lands lying waterward of extreme low tide in navigable tidal waters and waters beyond the line of navigability in navigable lakes, streams, rivers and streams.

freshwater plants. Marine plant communities will include marine and estuarine seaweed, eelgrass and kelp beds, and areas that may be covered only slightly or seasonally with ephemeral seaweeds or eelgrass. This will generally include marine bedlands down to a depth of the lower limit of vegetation or about sixty feet.

Watershed Approach

While the aquatic plants of concern occur on state-owned aquatic lands, consideration will be given to activities on other ownerships. Aquatic plant habitats are at the "bottom of the hill" and receive waters from development, logging, agriculture, municipal sewage discharges, etc. Because of water flow and pollutant transfer, these sites cannot be isolated in the same manner as upland sites. Hence, there will be a strong component of cooperation with various levels of government and within DNR on a watershed or ecosystem basis.

Objectives

The following are four objectives of aquatic plant management:

1. Preserving unique aquatic plant species and communities;
2. Maintaining , enhancing , or restoring aquatic plant habitat;
3. Ensuring that aquatic plant harvest maintains habitat function; and
4. Developing and improving production of seaweeds for production of food and fiber, to reduce harvesting pressure on natural beds, and to replace aquatic plants lost through development or pollution.

Inventory

A common task in these four objectives is to inventory aquatic plant resources. Each objective's inventory task has a slightly different set of criteria (see Table 1), but it is critical that any inventory work be carefully designed to gain as much information useful to each portion in order to avoid duplication of effort. The Table also shows that other state and federal agencies have a high level of interest in aquatic plant inventory work, especially the PSWQA's Monitoring and Wetlands programs, and EPA's Puget Sound Estuarine Program. Local governments, through their Shoreline Master Plans and issuance of shoreline permits have a high need for this information.

Table 1. Inventory tasks and agency program interests.

Inventory Tasks	DNR APMP	PSWQA Mon	PSWQA W-1, 2,3	PSWQA W-6	DoE CZM	NCRI CZM	EPA Adv ID	PSEP	OCS
<u>Preservation</u> Unique or representative communities or species	•		•	•		•	•		
<u>Habitat Protection</u> Based on vegetation classification scheme; Map habitat and function; Monitor long-term changes	•	•	•	•	•	•	•	•	•
<u>Harvest Management</u> Map resources-quantity, standing crop, location, and extent, year-to-year variation, seasonality, Long-term for variation and effect of harvest	•			•		•			

DNR APMP- Department of Natural Resources Aquatic Plant Resource Management Plan
PSWQA- Puget Sound Water Quality Authority ; Monitoring; Wetlands - Criteria Development (W-1); Wetlands - Identification of Wetlands to be preserved (W-2); Wetlands - Wetland preservation (W-3); Wetlands - Program for state-owned lands (W-6)
DoE/CZM- Department of Ecology/ Coastal Zone Management
NCRI/CZM- National Coastal Research Institute/Coastal Zone Management
EPA Adv ID- Environmental Protection Agency- Advanced Identification of Wetlands Program
PSEP- EPA's Office of Puget Sound Puget Sound Estuarine Program
OCS- Mineral Managment Service- Outer continental Shelf

<u>Proposed Protocol Development and Characterization Project</u>

The proposed project will develop protocol for inventory techniques, perform a trend analysis for habitat distribution, and make a partial baseline inventory to be used for future trend analyses. The output will be a protocol for inventory, a trend analysis for selected wetland types and detailed data on six specific areas.

Tasks

1. Develop protocols for intertidal/upland marine and estuarine habitat inventory
    a. Compare costs vs. resolution for image acquisition for
        - Multiple Spectral Scanning
        - Aerial Photography
            - Color IR
            - Color
    b. Compare methods and costs for classification
        - Photo interpretation
        - Photo digitization
        - Signatures for digitized images (MSS or photos)
        - Hybrid interpretation/digitization
    c. Develop methods and costs for ground truthing
    d. Input into GIS

2. Develop protocols for subtidal habitat inventory using sidescan sonar

3. Characterize habitat quantity (trend analysis) for eelgrass, floating kelp, and salt marshes

4. Perform habitat inventory for test sites and map in GIS

Future efforts will include inventorying the functions of these habitats, determining the annual variation in these habitats and their functions, and documenting long-term changes of habitat and functions.

Discussion

The marine plant resources of Washington, long overlooked, are now recognized as providing the major part of the foundation of the Puget Sound ecosystem. These resources are largely in state ownership.

We are still in the exploratory phase, trying to discover where and how much of these resources are present and whether they are maintaining themselves. Next will come detailed studies elucidating the functions of these habitats. Then studies to allow us to restore, rehabilitate or create these habitats. Then wisest decisions can be made concerning their protection and use.

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ROCKY HABITAT MITIGATION USING ARTIFICIAL REEFS

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Introduction

Mitigation for man caused degradation of habitats is often required by resource agencies in an attempt to obtain the objective of no net-loss of in-kind habitat (USFWS 1981). Mitigation projects which fail to achieve this objective are often the result of using habitat modification techniques which are unproven. These failures are usually a manifestation of inadequate site selection and project evaluation studies, which are often "based not on quantitative measurements but on a walk over the site" (Steinhart 1987). This cursory treatment of many mitigation projects occurs because mitigation is often "seen .. as part of the bargaining process rather than part of the biological challenge of development" (Steinhart 1987).

Some mitigation projects which fail to achieve the no net-loss objective are often the result of practices which change the community structure in the mitigation habitat through the mass introduction of a desired species. Either the introduced species does not survive because the species is inappropriately placed in environmental conditions detrimental to its survival (Steinhart 1987), or it is eventually out-competed by a dominant resident species (Carter et al. 1985, Boesch 1987). Other unsuccessful mitigation projects often result from the introduction of a new habitat in locations which are physically inappropriate, or locations in which the dominant biota are not compatible with the introduced habitat (Bohnsack and Sutherland 1985).

The placement of rock (artificial reefs) has been extensively documented in the scientific literature as a successful technique for marine fisheries enhancement (see Bohnsack and Sutherland 1985). The successful use of artificial reefs is dependent only on the colonization by a natural succession of organisms which normally occur in natural rocky habitats. However, the application of artificial reefs as mitigation for damaged or lost rocky habitats has

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not been extensively studied or documented. In recognition that artificial reefs "...can enhance the habitat and diversity of fishery resources..." the National Artificial Reef Plan called for more research to determine if artificial reefs can be used as mitigation tools (Stone 1985).

The purpose of this paper is to demonstrate that artificial reefs can be used to achieve the no net-loss objective for man-caused losses of rocky habitats in Puget Sound, Washington. Data for this paper was obtained from the Port of Seattle's efforts to mitigate for a shoreline development (fill) project in Elliott Bay, which impacted approximately 2.83 hectares of rocky habitat, by constructing an artificial reef.

Methods and Materials

Future biological production on artificial reefs in Puget Sound is predicted by using a biota index (Natural Reef Index (NRI) comparison system developed for this region (Hueckel and Buckley 1988). Using this system, the strength of a prospective artificial reef site's biological production potential is based on the degree of similarity between the organisms common to productive natural rocky habitats in Puget Sound, which comprise the NRI, and the number of NRI organisms identified at that site.

The NRI organisms were used to gauge the biological productivity of potential mitigation sites against that of the development site. A site for the mitigation reef was considered to be acceptable if it was limited in rocky habitat and the number of NRI organisms identified at that site was equal to, or greater than, the number of NRI organisms identified at the development site. SCUBA was used to conduct random-search surveys, lasting 30 minutes, for NRI organisms at the development site and potential mitigation reef sites.

SCUBA surveys using the strip transect method (see Brock 1954) were conducted on various habitats to document the occurrence of recreationally important fish species. The habitats surveyed for this study included the development site, a rocky bottom approximately 1050 m west of the development site, the mitigation reef, and a firm sand bottom approximately 200 m north of the mitigation reef (Fig. 1). The development site consisted of .3 m -1.2 m dia rip-rap and pier pilings which extended from MHHW to 12.2 m below MLLW. The bottom substrate below 12.2 m consisted of soft sand and mud. The rocky bottom consists of .15 m-1.2 m dia rocks covering firm sand, and served as a reference for the development site. The mitigation reef was constructed during May 1987 of 181,400 metric tons of 0.3 m 1.2 m dia quarry rock situated in 14 piles covering a firm sand bottom. The firm sand bottom served as a control for the mitigation reef site.

Epifauna was sampled from hard substrates at the development site (prior to filling) and the mitigation reef using an airlift sampling device described by Benson (1988). Infauna was sampled from the sand bottom control site by penetrating 30.5 cm X 2.9 cm diver

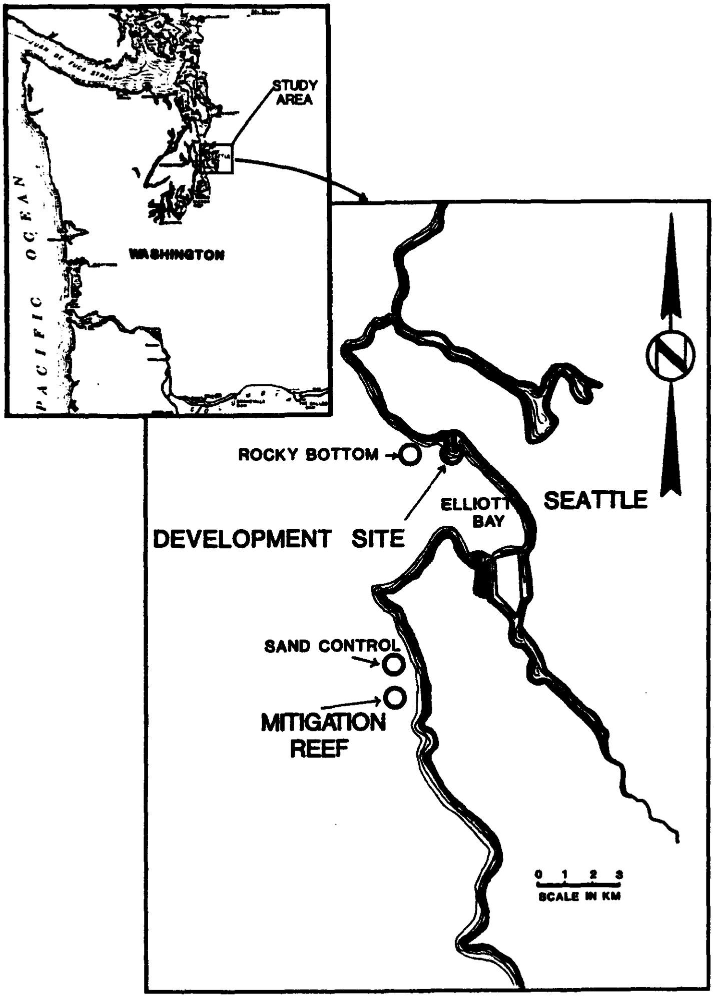

Fig. 1. Location of the mitigation reef, development site, rocky bottom, and sand control surveyed for this study in Puget Sound, Washington.

held core tubes 15 cm into the substrate. Infauna was sampled around and under concrete slabs at artificial reefs constructed during 1980 on cobble (BI) and during 1982 on sandy mud (CB)(see Hueckel and Buckley 1988), measuring from 1.2 m X 1.5 m to 1.2 m X 2.4 m, using the modified airlift sampling device at BI and core tubes at CB. Lift bags were used to move the concrete slabs.

Results and Discussion

Using the prediction capability of the NRI, four biologically acceptable sites for the mitigation reef were located. Three of these sites adjacent to the development site were eliminated from consideration due to potential conflicts with existing commerce boat traffic and/or commercial net fisheries. The closest biologically acceptable site was located eight km south of the development site (Fig. 1).

A total of 19 NRI organisms were identified at the chosen mitigation reef site which were associated with an outfall pipeline and other artifacts (tires, bottles, etc) compared to 9 NRI organisms identified at the development site (prior to filling). Similar numbers of NRI organisms observed at the mitigation site were identified at three different rocky habitat limited sites off Blake Island (BI), Olympia (IL), and Whidbey Island (PP) in Puget Sound (see Hueckel and Buckley 1988). Surveys of artificial reefs subsequently constructed at these three sites identified up to twice the number of fish species important to local fisheries (Table 1), and significantly greater diversities of sessile and epibenthic biota assemblages, as the development site prior to filling (Table 2). From this information, it was predicted the mitigation reef would develop similar biota assemblages and diversities as these three artificial reefs.

Table 1. Densities (per 100 m^2) of rocky reef fish species observed during SCUBA surveys on three artificial reefs, during their first two years of submergence, and the development site prior to filling, in Puget Sound, Washington.

	ARTIFICIAL REEFS IL N=7	BI N=15	PP N=5	DEVELOPMENT SITE N=3
Shiner perch (*Cymatogaster aggregata*)	2.9	2.7	1469.6	8.0
Striped seaperch (*Embiotoca lateralis*)	100.4	9.3	37.7	18.5
Pile perch (*Rhacochilus vacca*)	70.8	3.6	47.0	1.6
Brown rockfish (*Sebastes auriculatus*)	0.7	0.3	0.1	0.5
Copper rockfish (*Sebastes caurinus*)	0.8	2.8	0.8	0.2
Yellowtail rockfish (*Sebastes flavidus*)			0.7	
Quillback rockfish (*Sebastes maliger*)	0.3	0.4	1.9	
Copper/Quillback juvenile rockfish	0.1	<0.1	5.2	0.3
Black rockfish (*Sebastes melanops*)	<0.1		<0.1	
Yelloweye rockfish (*Sebastes rubberimus*)			<0.1	
Kelp greenling (*Hexagrammus decagrammos*)				<0.1
Lingcod (*Ophiodon elongatus*)	1.1	0.1	3.4	
Cabezon (*Scorpaenichthys marmoratus*)	0.1	<0.1	<0.1	
(TOTAL)	(177.2)	(19.2)	(1566.4)	(29.1)
TOTAL NUMBER OF SPECIES	9	8	11	6

Table 2. Invertebrate species collected from the mitigation reef, IL*, BI*, and PP* artificial reefs, and the development site prior to filling in Puget Sound, Washington during July. (*During the first two-four years of submergence).

INVERTEBRATES	MITIGATION REEF No. per m²	DEVELOPMENT SITE No. per m²	ARTIFICIAL REEFS IL	No. per m² BI	PP
PORIFERA					
Scypha sp.			280		
PLATYHELMINTHES					50
NEMERTEA					
Nemertean worms:					
Nemertean sp.					100
ANNELIDA					
Polychaete worms:					
Ampharetidae			20		
Cirratulidae					
Ophellidae	100	1675			
Nereidae		125			50
Phyllodocidae	20		20	20	150
Polynoidae			1000	20	
Sabellariidae			600		
Sabellidae					250
Syllidae	20		160		50
MOLLUSCA					
Chitons:					
Tonicella lineata				20	
Snails and limpets:					
Crepidula nummoris				40	
Crepipatella lingulata			480		
Lacuna variegata				20	
Nassarius mendicus			80		
Notoacmea scutum				20	
Ocenebra interfossa				20	
Odostomia sp.		1250	600	20	
Seaslugs:					
Nudibranch sp.			60	40	
Bivalves:					
Clinocardium nuttallii	20				
Chlamys hastata				40	
Chlamys rubida					50
Hiatella gallicana			60		
Macoma sp.		50			
Mytilus edulis			20		300
Musculus sp.		25			
Pododesmus cepio					100
Psephidia lordi					100
ECHINODERMATA					
Eupentacta sp.			20		
ARTHROPODA					
Pycnogonida				20	
Copepods:					
Calanoida			20		
Harpacticoida			40		
Tanaidacea	20		40		
Barnacles:					
Balanus glandula	2220	200			3650
Mysids:					
Mysid sp.		50	120		100
Amphipods:					
Caprellidea			100	180	
Gammaridea	40		480	200	
Shrimp:					
Crangonidae		75			
Eualus herdmani				60	
Eualus sp.	20				
Lebbeus sp.			20	20	
Spirontocaris sp.				20	50
Crabs:					
Cancer oregonensis			120		650
Hyas lyratus			60	20	50
Pagurus sp.				40	
Pugettia gracilis		25			
Megalops	20				
CHORDATA					
Sea squirts:					
Ascidia sp.			20		
Boltenia villosa			20		
Chelyosoma columbianum			520		50
Corella willmeriana			120		
SHANNON'S DIVERSITY INDICES	.230	.542	1.124	1.066	.632

While the mitigation reef site was relatively free of commerce conflicts, the impact to the sand bottom habitat from the reef still had to be considered to insure impacts would not outweigh the benefits of future reef production. Few scientific studies have examined the effects of artificial reefs on biota inhabiting the surrounding open bottom (Bohnsack and Sutherland 1985). Most artificial reefs are rapidly colonized by adult fish, which presumably originated on nearby natural reefs (Bohnsack and Sutherland 1985, Mathews 1985), but the long-term affects on the natural reefs is still unknown. Fish, other epifauna, and infauna in the areas surrounding artificial reefs appear to be unaffected (Walton 1982, Davis et al. 1982).

The open bottom habitat which surrounds artificial reefs is known to supply food items for some reef fishes (Hueckel and Stayton 1982). Energy is transferred from this habitat to the reef when open bottom habitat fish are consumed by other reef fishes. Additional energy transfer occurs through reef dwelling detritivours consuming feces deposited by open bottom foraging reef fishes (Bray 1985, Hueckel and Buckley 1987). The interface between the reef and surrounding open bottom has also been shown to be important to some key reef fishes (Wilson and Kren 1986). Based on this research, we consider the open bottom around artificial reefs to be of equal importance to the reef community as the reef itself.

The affects of concrete slabs placed over sand, and sand and cobble bottoms were examined at the CB and BI artificial reefs. The diversity of infauna decreased significantly from around, to under, the three slabs at each location (Table 3, Table 4). The open

Table 3. Density (per m²) and diversity indices of infaunal organisms identified from core samples taken from and and under six large concrete blocks following five years of submergence at the CB artificial reef in Puget Sound, Washington.

	Number of Organisms (per m²)				Shannon's Diversity Indices			
Location	BLOCK				BLOCK			
From Block	1	2	3	(AVG)	1	2	3	(AVG)
1 meter	4541	6052	9078	(6557)	.579	.458	.436	(.491)
Adjacent	15129	1513	4539	(7060)	.301	.602	.880	(.594)
Inside Edge	1513	6052	4539	(4035)	.477	.602	.000	(.360)
Center	0	1513	0	(504)	.000	.150	.000	(.050)
Inside Edge	3026	1513	0	(1513)	.000	.000	.301	(.100)
Adjacent	12104	6052	3026	(7061)	.477	.000	.527	(.335)
1 meter	10591	7565	7565	(8574)	.678	.452	.477	(.536)

Table 4. Density (per m^2) and diversity indices of infaunal organisms identified from core samples taken from and and under six large concrete blocks following six years of submergence at the BI artificial reef in Puget Sound, Washington.

Location From Block	Number of Organisms (per m^2) BLOCK 1	2	3	(AVG)	Shannon's Diversity Indices BLOCK 1	2	3	(AVG)
1 meter	1347	2383	1399	(1710)	1.059	1.101	.973	(1.044)
Adjacent	3420	2953	1761	(2711)	1.112	.887	1.040	(1.013)
Inside Edge	518	1709	1450	(1226)	.797	.849	.974	(.873)
Center	363	311	1036	(570)	.641	.540	.593	(.591)
Inside Edge	259	259	673	(397)	.413	.413	.662	(.496)
Adjacent	2901	1813	932	(1882)	1.025	1.126	.829	(.993)
1 meter	2435	1191	1295	(1640)	1.074	1.017	.820	(.970)

sand bottom at the mitigation reef site supports a diverse assemblage of infaunal organisms (Table 5) which have been shown to be important prey items for some reef fishes (Hueckel and Buckley 1987). The commercially important geoduck clam (Panope generosa) were also present in the reef area.

The mitigation reef was constructed using a 1:2 ratio of reef:open bottom, with each reef structure imprinting approximately 615 m^2. This spacing provided natural open benthic foraging areas between structures while maintaining continuity of the reef fish community and the trophic level relationships normally occurring between the reef structures and surrounding habitats. A 1:2 reef: open bottom ratio did not alter the colonization of fishery important fish species to other Puget Sound artificial reef structures (Buckley and Hueckel 1985). The ratio of reef:open bottom, which yielded approximately 15.2 m between the reef structures, also minimized the overcovering of geoducks in the reef area.

Mitigation should be viable for as long as the habitat subject to the mitigation is impacted, which is often permanent. There is often no long-term commitment to mitigation as the interest for the mitigation project declines once the project requirements are met (Steinhart 1987). Therefore, artificial reefs for mitigation should be constructed with non-deteriorating materials which are suitable for the development of a living natural reef community. Artificial reefs which have been constructed from materials which deteriorate in water have developed viable natural reef communities (Bohnsack and Sutherland 1985), but the limited life spans of the base materials are detrimental to the reef communities (Turner et al. 1969).

Table 5. Invertebrate species collected in 15 core samples taken between 12.2 m to 18.3 m at the mitigation reef site during March 1987 prior to reef construction.

	No. per m^2
ANNELIDA	
Capitellidae	100
Chaetopteridae	300
Glyceridae	200
Goniadidae	100
Hesionidae	100
Maldanidae	300
Magelonidae	100
Nereidae	100
Nephtyidae	100
Oligochaeta	100
Phyllodocidae	100
Spionidae	400
Terebellidae	100
Trichobranchidae	100
NEMERTINA	100
MOLLUSCA	
Snails:	
Bittium eschrichtii	100
Clams:	
Axinopsida sericata	200
*Panope generosa	1.7
Psephidia lordi	100
Veneridae	100
ARTHROPODA	
Amphipods:	
Gammaridea	400
SIPUNCULA	
Golfingia sp.	100

* Panope generosa (Geoduck clams) were quantified using three 100 m X 2 m strip transects through the mitigation reef site.

Continued replenishment of these types of materials may not provide for adequate mitigation, as the reef community would be in a constant state of early successional development, and would thus fail to replicate the habitat subject to mitigation.

The mitigation reef was constructed from quarry rock because this material meets the positive factors required for long-term mitigation of rocky habitats and it is available in large quantities. The stacking of this material during construction and the use of different sized rocks (.3 m-1.2 m dia) creates the diverse crevice habitat necessary for the attraction and survival of many reef-associated and free-living organisms (Hueckel and Buckley 1987).

The mitigation reef has met the objective of developing a similar fish species assemblage as the development site prior to its filling, and greater fish densities, during the reef's first eight months of submergence (Table 6). Fish species diversity and densities on the mitigation reef has surpassed that observed on the rocky bottom adjacent to the development site. Some displacement of resident fish appeared to have occurred as evidenced by the greater diversity and number of flounder species observed on the adjacent sand bottom compared to those observed on the sand bottom between the mitigation reef structures (Table 6).

Table 6. Densities of fish species (per 100 m²) on the mitigation reef, sand control, and rocky bottom from July 1987 through January 1988.

	Mitigation Reef (n=7)	Rocky Bottom (n=7)	Sand Control (n=7)
Shiner perch	209.2	4.0	
Striped seaperch	87.7	5.8	
Pile perch	38.0	0.3	
SURFPERCH (TOTAL)	(334.9)	(10.1)	
Brown rockfish	0.3		
Copper rockfish	2.6	0.1	
Quillback rockfish	14.5	10.1	
Juvenile rockfish	1.0		
ROCKFISH (TOTAL)	(8.4)	(0.2)	
Lingcod	0.1		
Cabezon	0.5		
Rock sole (_Lepidopsetta bilineata_)	0.2	0.2	1.4
Speckled sanddab (_Citharichthys stigmaeus_)			0.3
C-O sole (_Pleuronichthys coenosus_)			0.3
English sole (_Parophrys vetulus_)			0.6
Flounder sp. (Pleuronectidae)	0.2		2.5
FLOUNDER (TOTAL)	(0.4)	(0.2)	(5.1)

The number of rocky reef fish species which colonized the mitigation reef is similar to those which colonized the IL, PP, and BI artificial reefs (see Hueckel and Buckley 1988, Fig. 2). The mitigation reef is also undergoing similar successional development as other productive artificial reefs in Puget Sound. The domination of the reef surfaces by barnacles (Table 2), and the subsequent

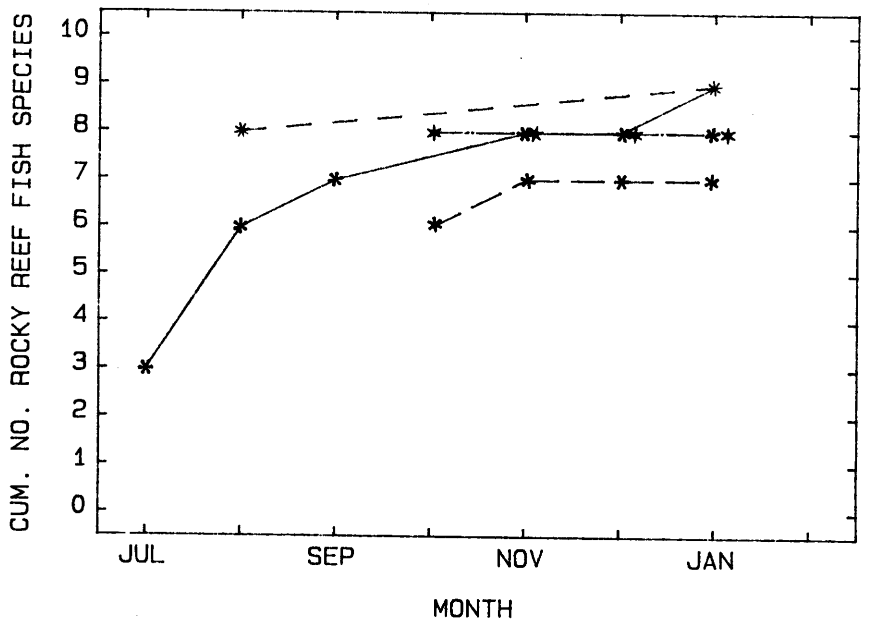

Fig. 2. The number of rocky reef fish species observed on the mitigation reef (*), and the PP (*), BI (*), and IL (*) artificial reefs in Puget Sound during their first year of submersion.

grazing by starfish (Evasterias troscheli and Pycnopodia helianthoides) is a precursor to the development of turf algae and increased densities and diversities of fish prey items and fish species (Hueckel and Buckley 1987). It can be expected that the mitigation reef will be colonized by additional fish species as this successional process continues.

Acknowledgements

We wish to thank the Port of Seattle for providing the funds for this research and to construct the mitigation reef.

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Juvenile Salmon Foraging In A Restored Wetland[1]

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Abstract

A 9.6 acre wetland system was constructed in 1985-1986 in the Puyallup River estuary to mitigate the filling of a proximal, similarly-sized wetland. The Lincoln Avenue wetland system is a habitat mosaic of tidal channels, mudflats, a sedge marsh, a cattail marsh, a swamp, riparian hardwoods, and a grassland. Monitoring since March 1986 has shown that the wetland system is utilized in the spring by outmigrating juvenile salmon. Chironomid insects (midge larvae, pupae, and adults) were the dominant prey in chum salmon (*Oncorhynchus keta*) and chinook salmon (*O. tshawytscha*) stomachs. There was essentially no overlap between epibenthic prey potentially available and the prey consumed. Continued monitoring of the restored wetland will not only increase our understanding of the functional value of such wetlands to juvenile salmon but should also have a significant bearing on the design and monitoring of future mitigation of this type. Comparable data from a "natural" wetland in southern Puget Sound would be invaluable in assessing the success or failure of the restoration project, but is not available in the forseeable future.

Introduction

During the last century, 98.6% of the wetland habitat once present in the Puyallup River estuary has been destroyed through dredging, diking, and filling (Bortleson et al. 1980). Historical losses and alteration of wetland habitat may have severely impacted the Puyallup River salmon runs, but the outmigration of juvenile salmon (*Oncorhynchus* spp.) through the Puyallup River estuary has only been documented recently.

Pacific Northwest wetlands are hypothesized to be important habitats for juvenile salmon temporary residence, seawater acclimation, refuge from predation, foraging, and staging (Dorcey et al. 1978; Simenstad et al. 1982). The value of wetland habitats as foraging areas, however,

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is poorly understood and has been examined only in marsh habitats of the Fraser River (Dunford 1975; Northcote et al. 1976; Levy et al. 1979; Levy and Northcote 1982) and the Skagit River (Congleton et al. 1984).

The principal objective of this ongoing study is to assess the functional value of the restored Lincoln Avenue wetland habitat as a foraging area for outmigrating juvenile salmon. The Lincoln Avenue wetland system is expected to be a valuable foraging area, based on the assumption that habitats where juvenile salmon reside and grow are relatively more important than habitats used merely for migratory purposes through urbanized estuaries. In addition, the nascent Lincoln Avenue system is the only wetland habitat now available to outmigrating juvenile salmon in the Puyallup River estuary.

Study Area

In 1985-1986, the Port of Tacoma constructed, by dredging and grading, a 3.9 ha (9.6 acre), 2.2 million dollar mitigation wetland in the Puyallup River estuary located in central Puget Sound, Washington. The river dike was breached on February 20, 1986 following excavation of approximately 55,000 m^3 of solid waste landfill and contouring of the new wetland topography (Thom et al. 1987). In accordance with resource agency criteria, 50% of the areal habitat was designed to support juvenile salmon, 20% waterfowl, 10% raptors, and 10% small mammals. Transplanting of 49,000 culms of the sedge, *Carex lyngbyei*, onto flats has been successful and recruitment of other plant taxa appears to be enhanced by the presence of the transplants. In its present state of development, the restored Lincoln Avenue wetland is a tidally influenced estuarine wetland system comprised of tidal channels, mudflats, the transplanted sedge marsh, a cattail (*Typha* spp.) marsh, a swamp, trees, and a grassland. The tidal channels, mudflats, and central basin, which cover approximately 2.2 ha, represent the principal fish habitat.

Methods and Materials

As an initial test of the functional value of the wetland as a foraging area for juvenile Pacific salmon, fish were collected from tidal channels at ebb slack tide on eight occasions between March 5 and July 15, 1986 and on twelve occasions between February 17 and June 16, 1987 using a 9.2-m floating beach seine with a 6-mm mesh bag and solid core lead line. Captured fish were sorted and subsamples of at least ten individuals of each different species were preserved in 70% isopropyl alcohol. In the laboratory, all preserved fish were identified, enumerated, measured (nearest 1 mm fork length [salmon] or total is poorly understood and has been examined only in marsh habitats of the Fraser River (Dunford 1975; Northcote et al. 1976; Levy et al. 1979; Levy and Northcote 1982) and the Skagit River (Congleton et al. 1984).

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As an assessment of juvenile salmon prey resources, epibenthic organisms which occupy the sediment-water column interface were suctioned monthly (March-June) from tidal flats, channels and the central basin using a battery powered epibenthic suction pump (Thom et al. 1987). Each suction sample was filtered directly through a 150 um mesh sieve and preserved in 5% buffered formalin. In the laboratory prey organisms were sorted and identified to the lowest possible taxonomic/life history level.

Results

Catch summary

Eleven species of six families of fish were collected from the wetland during 1986-1987 beach seine sampling (Table 1). The family Salmonidae was represented by five species, four of which--pink (*O. gorbuscha*) chum, coho (*O. kisutch*), and chinook salmon--are common estuarine residents as juveniles during their migration to the Pacific ocean. Mean fish densities varied markedly within and between years (Fig. 1). Chum salmon (31 to 58 mm) and chinook salmon (30 to 93 mm) accounted for between 20 to 50% of the total density. In 1986, the two peaks of chinook salmon density on March 28 (1.79/100 m^2) and June 13 (0.36/100 m^2) corresponded to the two peaks of total fish density. Chum salmon density peaked on April 25 (0.10/100 m^2) and June 27 (0.11/100 m^2). In 1987, total fish density peaked on March 17 (6/100 m^2) and May 30 (8/100 m^2); chinook salmon density peaked in late March (0.6/100 m^2) and late May (3.3/100 m^2); and chum salmon density peaked on April 2 (0.4/100 m^2) and May 30 (0.7/100 m^2). Mean total fish densities nearly doubled between 1986 and 1987 and in both years peak densities were observed shortly after hatchery releases of juvenile salmon.

Prey available/prey consumed

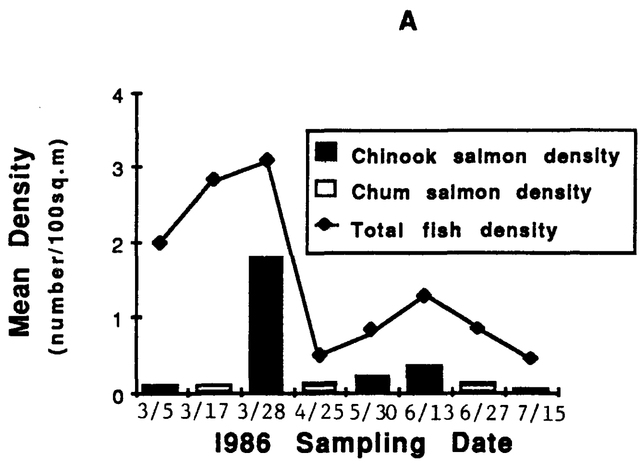
Epibenthic organisms throughout the wetland were considered to be potential available prey to foraging juvenile chum and chinook salmon. The principal taxa among the epibenthos were Nematoda, Oligochaeta,

Table 1. Fishes collected in 1986-1987 beach seine samples at Lincoln Avenue wetland, Puyallup River estuary.

Family/Species/Common name	
1986	1987
Family Salmonidae	Family Salmonidae
Prosopium williamsoni (mountain whitefish)	*Prosopium williamsoni* (mountain whitefish)
Oncorhynchus gorbuscha (pink salmon)*	*Oncorhynchus keta* (chum salmon)
O. keta (chum salmon) (chum salmon)	*O. tshawytscha* (chinook salmon)
O. kisutch (coho salmon)*	Family Cyprinidae
	Rhinichthys cataractae (longnose dace)+
O. tshawytscha (chinook salmon)	*Richardsonius balteatus* (redside shiner)
Family Cyprinidae	Family Catostomidae
Richardsonius balteatus (redside shiner)	*Catostomus macrocheilus* (largescale sucker)
Family Catostomidae	Family Gasterosteidae
Catostomus macrocheilus (largescale sucker)	*Gasterosteus aculeatus* (three-spine stickleback)
Family Gasterosteidae	Family Cottidae
Gasterosteus aculeatus (three-spine stickleback)	*Cottus asper* (prickly sculpin)
Family Cottidae	
Cottus asper (prickly sculpin)	*Leptocottus armatus* (staghorn sculpin)
Leptocottus armatus (Pacific staghorn sculpin)	Family Pleuronectidae
Family Pleuronectidae	*Platichthys stellatus* (starry flounder)
Platichthys stellatus (starry flounder)	

*only collected in 1986 +only collected in 1987

Harpacticoida, and Tardigrada. Midges (Chironomidae), stonefly nymphs (Plecoptera), two species of gammarid amphipods (*Corophium* spp.), flies (Diptera), and *Daphnia* spp. were among the dominant prey found in both chum and chinook stomachs. The most important prey category overall in terms of percent total IRI was Chironomidae (midge larvae, pupae and adults). A variety of other prey taxa (Arachnida, *Chaoborus*, Coleoptera, Collembolla, *Eogammarus confervicolus*, Harpacticoida, Homoptera, Hymenoptera, unidentified Insecta, *Neomysis mercedis*, Odonata, and



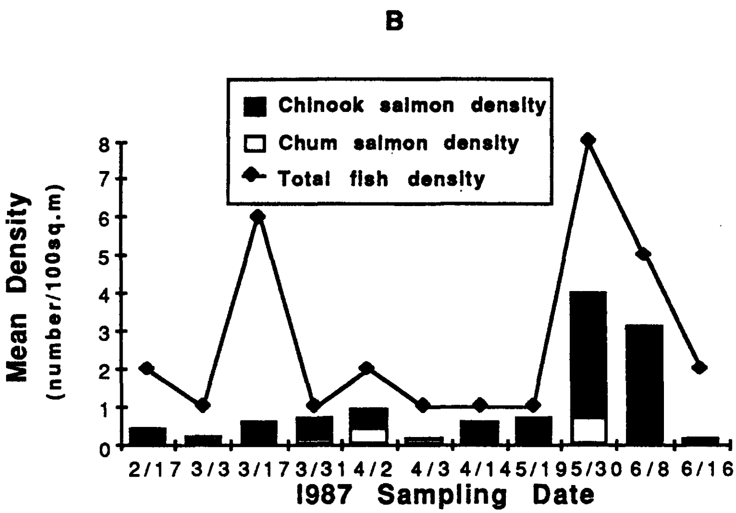


Fig. 1. Chinook, chum and total fish density comparisons for Lincoln Avenue wetland, Puyallup River estuary: (A) 1986; (B) 1987.

Osteichthys) were consumed infrequently or in small numbers. No nematodes, oligochaetes or tardigrades were found in stomachs. Comparisons of prey available and prey consumed during the outmigration period (March-May) indicated minimum overlap (Fig. 2).

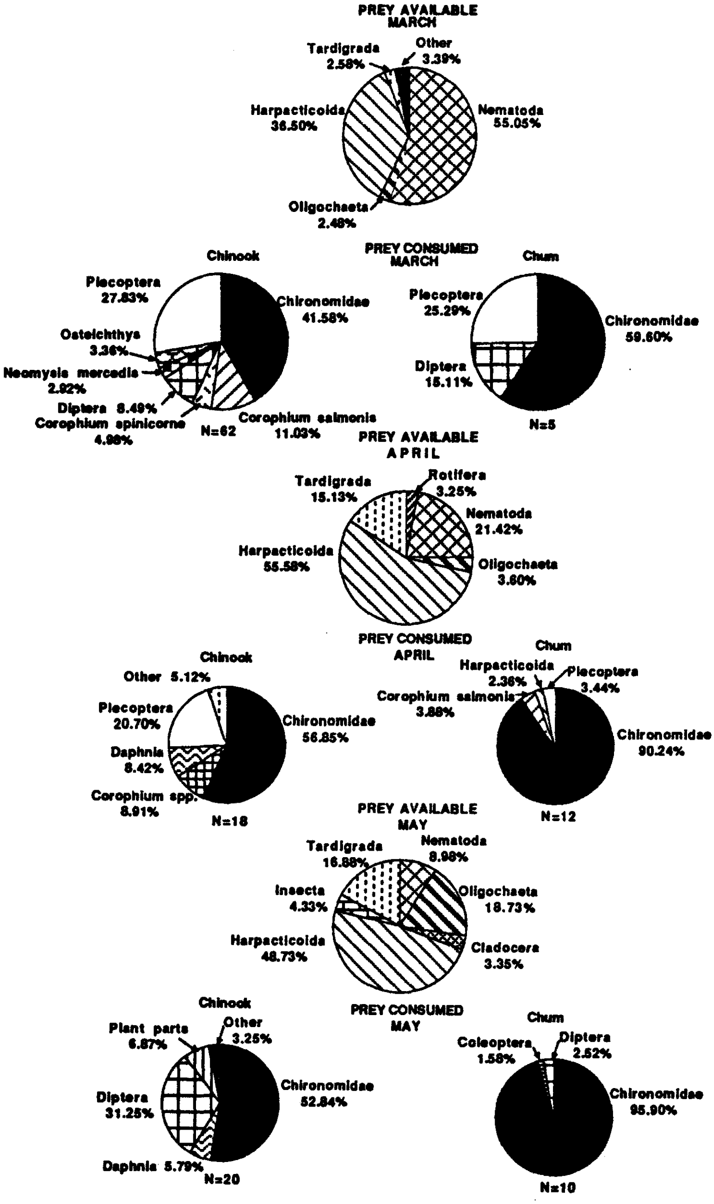


Figure 2. Prey availability (% composition of epibenthos) and rank importance (total % IRI) of items in the stomach contents of juvenile chum and chinook salmon collected in Lincoln Avenue wetland, Puyallup River estuary, March-May 1987. Sample sizes (n) are indicated below each pie.

Discussion

In the two years since the Lincoln Avenue wetland was constructed, eleven species of oligohaline and euryhaline fish have appeared in the new habitat. Juvenile chum and chinook salmon showed the highest densities of the eleven species utilizing the wetland habitat. The range of observed salmon densities (0.1-3.3/100 m^2) was low in comparison to densities of up to 77/100 m^2 in the Fraser River (Levy and Northcote 1982) or up to 178/100 m^2 in the Skagit River (Congleton et al. 1981). Changes in the bathymetry and morphology of the tidal channels due to sedimentation, the salinity regime, the composition and standing stock of prey resources, and the timing of hatchery releases may in part explain why juvenile salmon utilization of the wetland is still in a state of flux.

Our pilot experiments to document residence time in 1987, although preliminary, suggested that outmigrating juvenile chum and chinook salmon may reside in the wetland system for up to a week. If residence time is, indeed, several days to a week, the occurrence and standing stock of prey required to sustain normal growth rates (i.e., "carrying capacity") will be a critical measure of the wetland's success. Direct comparsion of epibenthic prey available in the wetland with prey found in chum and chinook salmon stomachs, however, indicated essentially no overlap. Although a number of the epibenthic taxa are only marginally available as prey for foraging salmon (e.g., organisms which are in the surface sediments or are difficult for the salmon to see in the turbid waters), many of the other taxa could constitute prey available for consumption. Based on the epibenthic samples from 1987, we estimate that between 500 and 2,000 organisms/m^2 represented the potential prey resource in the wetland between March and June. Yet, few of these organisms were fed upon frequently or in abundance by juvenile chinook and chum salmon (Fig. 2). Among a number of possible explanations for this discrepancy, two are most likely: (1) epibenthos sampling did not accurately assess the availability of the prey selected by the fish, which were predominantly emergent or drift insects and tubicolous gammarid amphipods; and, (2) the selected prey originated outside the wetland and were either consumed in the river before the fry entered the wetland or advected into the wetland as drift.

The foraging function of a restored wetland is one in a suite of possible performance criteria for evaluating wetland mitigation, but is especially difficult to assess for mitigation projects which may not achieve an equilibrium invertebrate community for many years. Simple descriptive documentation of the incidence of fish in a restored wetland does not test the benefit(s) derived from their utilization of the habitat. In the comparatively short period we typically have to monitor the development of mitigation wetlands, such as Lincoln Avenue wetland, manipulative experiments are needed. Thus, in order to verify the origins and quantities of food organisms available for consumption in the Lincoln Avenue wetland, we will conduct several experiments during the 1988 outmigration period in which we: (1) expand our sampling of prey to include emergent and drift insects and document the advection of drift prey from the river into the wetland; (2) obtain more precise estimates of juvenile chum and chinook salmon wetland residence times; and, (3) generate *in situ* consumption rate estimates for juvenile salmon residing in the wetland over diel foraging periods.

The adaptive value of delayed seaward migration and wetland foraging may ultimately be related to the availability of suitable types, sizes, and quantities of prey. But whether growth and survival rates of outmigrating juvenile salmon are enhanced by residence and foraging in wetland habitats requires clear demonstration. There is good support in the literature for increased survival with larger size. If juvenile salmon growth in wetland habitats is more rapid than growth in riverine or disturbed estuarine habitats, wetland foraging should correlate with increased survival. The hypothesized adaptive value of wetland residence and foraging for outmigrating juvenile Pacific salmon is intuitively appealing, but the need for greater understanding of the relationship between habitat production and dependency of juvenile salmon on wetlands is glaringly evident.

Ultimately, even rigorous evaluations of wetland functions, such as their value as foraging habitats, will depend upon comparable data from "natural" habitats in order to scale the results to a reference, or baseline, situation. At this time we are severely constrained in our interpretation of the status of the Lincoln Avenue wetland system because there are virtually no reference data available from emergent plant wetlands in the region which describe fish consumption rates and prey carrying capacity. The Lincoln Avenue wetland restoration project, the largest estuarine mitigation project in the state of Washington, is being closely followed by the Port of Tacoma, the Puyallup Indian Tribe, and state and federal resource agencies as a means of compensating for fish habitat losses. However, until we document foraging of juvenile salmon in undisturbed, natural estuarine marsh habitats, it is virtually impossible to evaluate mitigation projects of this nature.

Summary

Although the Lincoln Avenue wetland system presently appears to be in a state of flux, catch results from 1986-1987 demonstrate that the system is being increasingly utilized by juvenile pink, chum, coho, and chinook salmon, as well as six other non-salmon species. Overlap between prey available and prey consumed by juvenile chinook and chum salmon was minimal, highlighting the need for continued and revised monitoring. The critical question which needs to be addressed is whether outmigrating juvenile Pacific salmon which forage and temporarily reside within wetland habitats, especially wetland mitigation sites such as the Lincoln Avenue wetland system, exhibit measuraby better growth and survival rates than fish which bypass these wetlands. Sampling of the Lincoln Avenue wetland in 1988 will focus on the residence time, daily ration and prey sources of outmigrating juvenile salmon. Field collections and experimental manipulations should help to further ellucidate the role and status of the restored wetland habitat as a foraging area.

Recommendations

As expansion of urbanization in Puget Sound continues to place pressures on the few remaining wetlands in Washington's river valleys, the need for greater understanding evaluation of wetland functions is increasingly urgent. Wetland habitat is at a premium, but the viability of creating and restoring wetlands as a means of compensating for fish habitat losses remains questionable. The time to preserve our remaining wetlands is now before these invaluable habitats fall victim to backhoes and bulldozers. Wetland mitigation tomorrow is a poor alternative to wetland preservation today.

Acknowledgments

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An Assessment of Wetland Mitigation Practices Pursuant to Section 404 Permitting Activities in Washington State*

INTRODUCTION

Of the many issues associated with the Clean Water Act Section 404 program, the one debated most often is the policy of replacing natural wetlands with created or "artificial" wetlands. The concept, which has come to be known simply as "mitigation", originated as a method to allow development to occur without suffering a net loss of wetland habitat. (1)

Habitat creation is a concept which has long been used in the management of wildlife preserves. Eventually the idea was applied within the regulatory arena to offset or mitigate for resources lost to development. Not surprisingly, it has been readily accepted within the Section 404 permitting process. But, after the approval and construction of numerous wetland mitigation projects, various researchers began to question their success. The cries for caution are best summarized by Zedler's (1986) statement that "on a national level, the technology of wetland creation/restoration is experimental and unpredictable."

Today, realizing that habitat creation may not work, regulators often cite this lack of technology for creating wetlands as the reason for many project failures. This statement may be well-founded. However, a closer look reveals that other factors are contributing to the poor success rates. Inadequate mitigation negotiation, documentation, planning, monitoring, and enforcement may doom many projects before they ever reach a stage at which we can blame technology. The focus of this study was to evaluate the effectiveness of the mitigation negotiation and planning process in achieving the intended goal of offsetting the loss of wetland ecosystem.

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*Authors- Kathleen Kunz, Michael Rylko, and Elaine Somers of the Environmental Protection Agency, Region 10, Seattle.

(1) For the purposes of this study, the term mitigation is defined as compensation for wetland losses in the form of wetland creation, restoration, or enhancement.

METHODS

Each mitigation project was evaluated for content of the mitigated agreement, compliance with the agreement, and a qualitative assessment of the habitat types, functions and values of the original wetlands compared to those planned for replacement. Habitat types were defined according to the USFWS classification system (Cowardin, et.al., 1979).

Data were obtained from EPA and Corps of Engineers (COE) project files, from interviews with persons involved with the various projects, and from qualitative field assessments. Only projects located in Washington State and permi ted between 1980 and 1986 were considered (no mitigation projects were discovered prior to 1980). To cooperate with a national study by EPA's Corvallis Environmental Research Laboratory (CERL), all data gathered were entered into the Wetlands Values Database, a computerized system designed and provided by CERL. The database included information on both the original wetlands permitted for development, and the mitigation sites.

RESULTS

Thirty-five mitigation projects were identified as having been approved via the 404 process between 1980 and July of 1986. Table 1 illustrates the temporal distribution of these projects over the seven year period; Figure 1 depicts the geographic distribution of the projects. After reviewing the data, several trends and conclusions surfaced:

1. Few wetland losses under Section 404 were mitigated.

2. The number of mitigations required increased steadily since 1980.

Table 1. Temporal distribution of required mitigation projects

	Year	Number of Projects
	1980	1
	1981	1
	1982	4
	1983	5
	1984	9
	1985	6
	1986	9
TOTAL		35

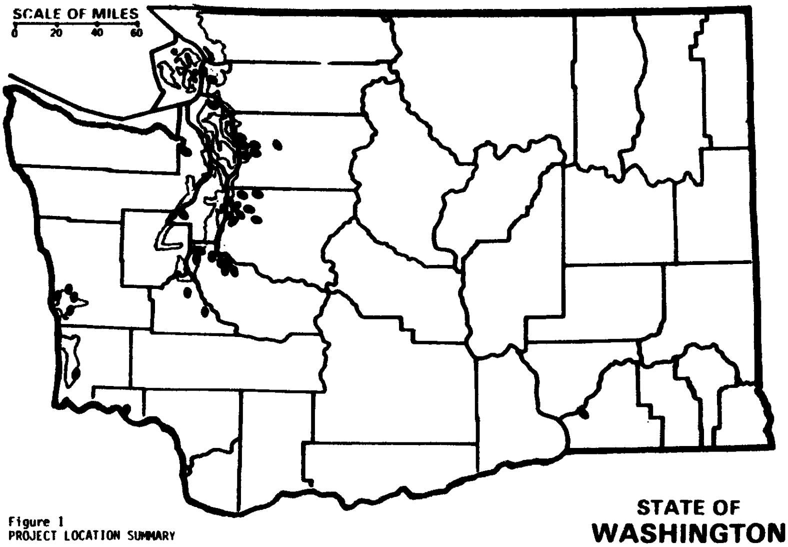

Figure 1
PROJECT LOCATION SUMMARY

3. Mitigation projects occurred most often in the more densely populated areas of the state. Western Washington projects were more often mitigated than those in Eastern Washington.

4. With mitigation there was still a substantial net loss of wetland acreage.

5. With mitigation there was a net loss of wetland diversity.

6. Not all wetland functions and values were considered or replaced.

7. Wetland losses occurred in time as well as in space. Temporal losses were not taken into account.

8. Mitigation designs were not effectively incorporated into the final 404 permits.

9. There was no routine procedure for tracking the functional success of all mitigation projects.

Few Wetland Losses Under Section 404 are Mitigated

Less than 1% of all Section 404 permits required mitigation (2). In an effort to understand why this was true, we compared the size of development projects involving mitigation. The average size of permitted development projects involving mitigation was 4.3 acres, while the average size of all §404 projects with or without mitigation was approximately .5 acre (3).

(2) Data compiled in EPA Region X Wetland Tracking Database.
(3) Ibid

Apparently there is a tendency to seek mitigation for the filling of larger wetland parcels rather than for all wetland losses. This may be due to (a) the large number of permits issued each year. Pursuing mitigation for projects under one acre may not be considered an effective use of agency staff time. And/or (b) agencies may not yet recognize the significance of smaller wetland losses. Published data on the cumulative effects of small §404 projects is sparse and has not yet drawn close attention or wide recognition.

With Mitigation There is Still a Substantial Net Loss of Wetland Acreage

Few applicants (7 of 35) proposed to compensate lost wetland acreage on a 1:1 basis; fewer (6 of 35) proposed compensating wetland acreage on a greater than 1:1 basis. Between 1980 and 1986, mitigation negotiations resulted in the exchange of 152 acres of natural wetlands for 100 acres of created/restored wetlands--a replacement rate of only 67% (Figure 2).

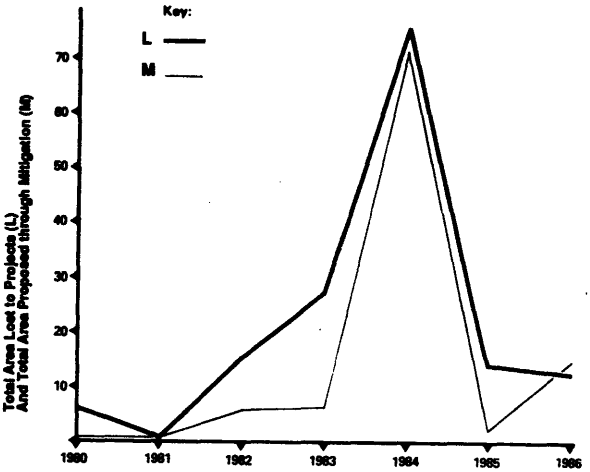

Figure 2
Mitigation Area vs. Project Impact Area

Further losses occurred when mitigation projects failed to develop as planned or were never constructed. In this study, 5 mitigation projects were not constructed or restored as negotiated. Because formal procedures for designing and implementing mitigation projects were not in place, these mitigation agreements "fell through holes" in the process and are not likely to be initiated even though development occurred.

A margin for error is also needed. In order to better offset losses, some amount of wetlands in excess of that lost must be planned for replacement. This is supported by researchers in California who also have reported that mitigation wetlands typically need to be larger than the original wetlands to achieve intended goals (Race, 1985; Eliot, 1985; Baker, 1984).

Mitigation Resulted in a Net Loss of Wetland Diversity

A wetland area may contain one or more habitat types. For example, one system may contain open water, creek bed, emergent vegetation, and a forested tract. For many of the projects surveyed (16 of 35), replacement of this diversity was not fully mitigated.

In this study, 73 wetland habitat types were permitted to be filled; 49 habitat types were proposed for mitigation--a net replacement rate of 68% (Figure 3). This loss was not equally distributed over the major wetland types (Table 2). Estuarine systems lost the least diversity even though they absorbed the largest number of projects. The prevailing emphasis on preserving anadromous or other commercial fishery habitat by the public and resource agencies may be the reason for this.

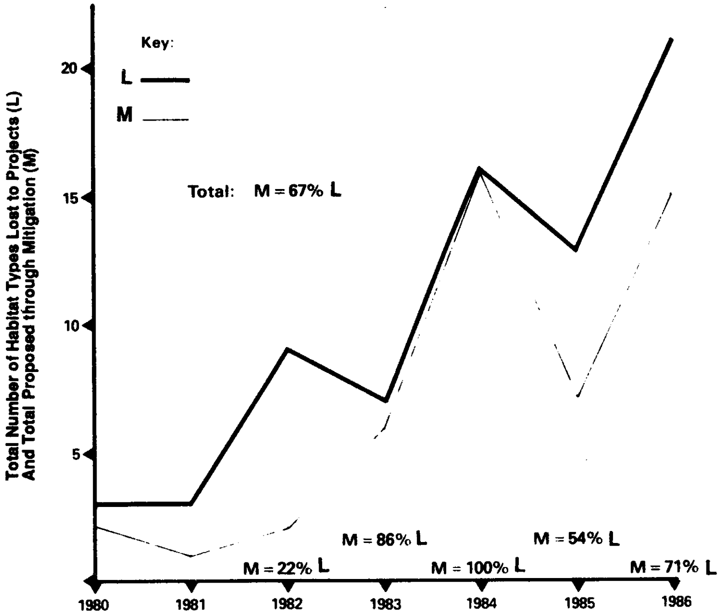

Figure 3

Number of Habitat Types Lost and Proposed

Table 2. Listings of habitat types impacted, created, restored, and percent of habitat replaced.

	Estuarine Habitat Type		
TYPE	# OF TYPES IMPACTED	# OF TYPES CREATED	% HABITAT REPLACED
EIUS*	15	13	87
EIRS	1	4	400
EIRE	1	1	100
EIEW	4	6	150
EIAB	4	4	100
ESAB	2	1	50
ESUB	4	1	25
ESRE	1	1	100
TOTALS	32	32	97%
	Riverine Habitat Types		
RTUS	1	1	100
RTEW	3	1	33
RTRB	1	0	0
RTUB	1	0	0
RUAB	4	2	50
RUUS	1	0	0
RLUS	1	0	0
TOTAL	12	4	33%
	Lacustrine Habitat Types		
LLUB	2	1	50
LLUS	4	2	50
LLAB	2	0	0
LLEW	0	2	200
LLRS	0	1	100
TOTALS	8	6	75%
	Palustrine Habitat Types		
PASS	5	1	20
PAEW	8	7	87.5
PAFW	5	0	0
PAAB	1	0	0
PAUS	1	0	0
TOTALS	20	8	40%
OVERALL TOTALS	72	49	68%

*Types after Cowardin, et.al. (1979) listed in Appendix A

In contrast, no forested wetlands were replaced. Forested wetlands are complex systems which take many years to mature. It is likely that the ecological understanding of these habitat types was not sufficient to create them and, therefore, replacement was not required.

It is not necessarily a mitigation goal to produce more wetland types than are lost to development. A small, diverse wetland may be less ecologically important than a larger system with less diversity. However, the loss of diversity may indicate that current mitigation practices fall short in replacing wetland functions and values.

Not All Wetland Functions and Values Are Considered and/or Replaced

Specific replacement goals are stated in most mitigation project files. For example, intertidal wetlands may be created to improve fishery habitat or have a dual goal of enhancing fishery and shorebird habitats. We compared a number of functional values that were associated with the developed wetlands with the number of intended values of the mitigation projects (Figure 4).

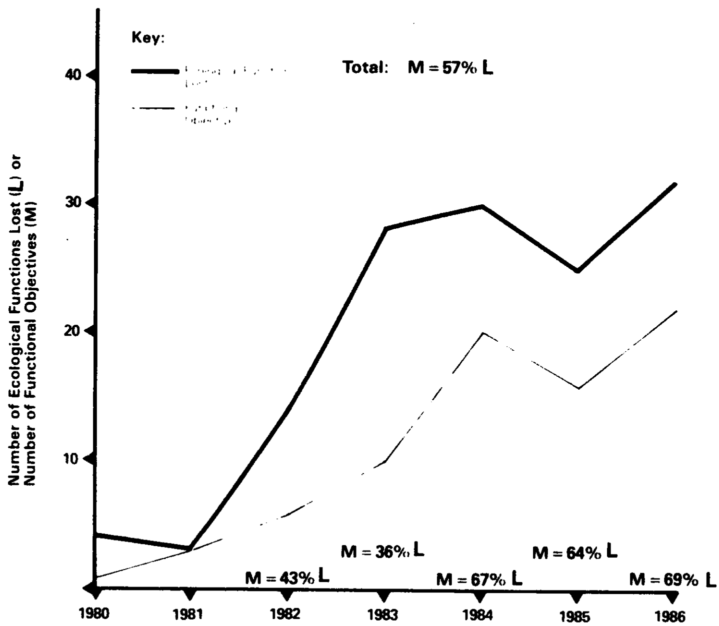

Figure 4
Wetland Functions Lost to Development vs. Functional Objectives of Mitigations

The following is a list of the ecological wetland functions that were considered: (4)

° fisheries habitat
° wildlife habitat
° ecological food chain support
° endangered species habitat
° nutrient retention
° sediment trapping
° flood storage and desynchronization
° uniqueness/rareness

The ecological functions lost were assessed using a qualitative method created for this study. Only functions rated as having a "high" potential value with respect to the original wetlands, were included in this comparison. Similarly, only those functional objectives of the mitigation projects that were documented somewhere within the project files were included.

The assessment revealed that 128 functions provided by the original wetlands were lost to development while the stated objectives of the mitigation projects only sought to replace 78 functions. The results (Figure 5) support Baker's (1984) observation from San Francisco Bay studies; the objective of almost all mitigation plans is to secure fish and wildlife habitat rather than to replace the full spectrum of wetland values. Sediment trapping, nutrient retention, and shoreline stabilization are usually not considered for replacement.

This type of comparison is helpful for evaluating the mitigation process. However, one should realize that although a given function may not have been specified for replacement, it may develop via natural processes. To an unknown degree, these inadvertent "gains" may be offset by the fact that not all planned objectives are fulfilled. To better anticipate these trade-offs, more research and better planning are required.

Wetland Losses Occur in Time As Well As In Space

Substantial time lags exist between project construction and mitigation completion (Figure 6). Between 1981 and 1984 the time lags ranged from 45 to 165 weeks. These lag times are underestimates. Many of the projects permitted in 1985 and 1986 have only recently been initiated and, therefore, have time lags which are still accruing. Data represent an average lag time as of 4/87, reflecting a minimum possible value. Projects with development impacts not yet initiated have not been included in this evaluation. Also, the mitigation completion dates were obtained from the contractors and represent official construction completion dates--not the beginning of ecological function, which is assumed to evolve sometime later.

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(4) As we lacked the tools to adequately evaluate groundwater recharge and discharge, these functions are not addressed even though they may be in operation.

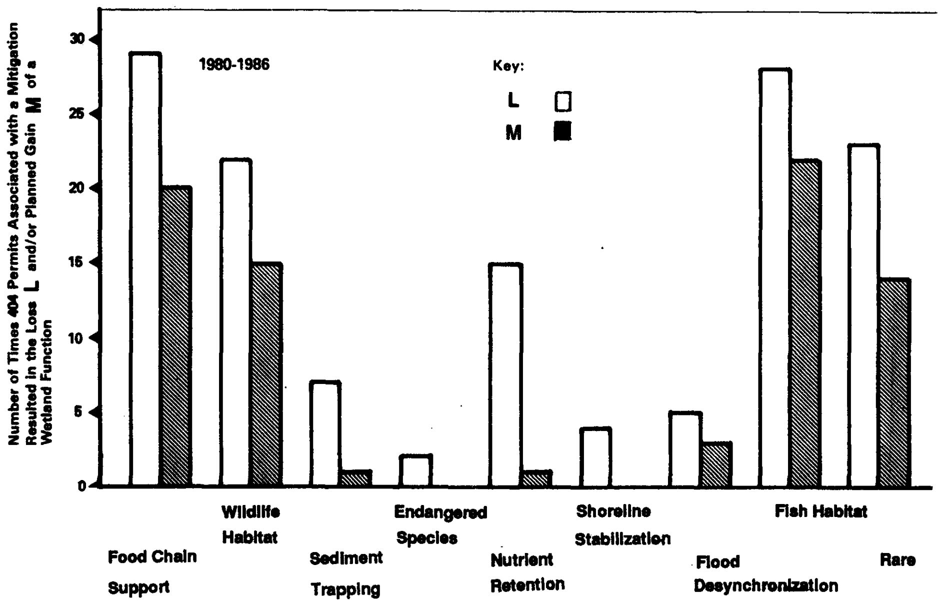

Figure 5
Total Number of Functions Lost vs. Mitigation Functional Objectives

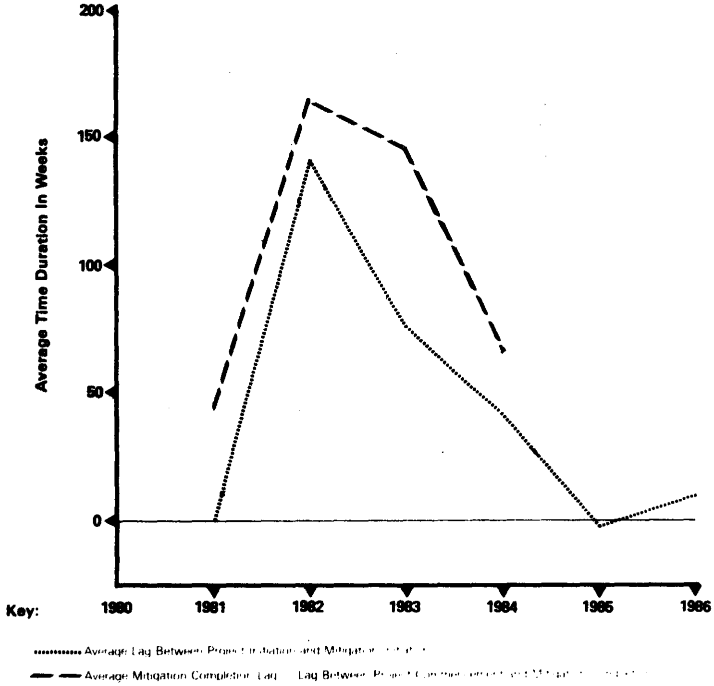

Figure 6
Time Lags Within the 404 Mitigation Process

Of the 26 completed projects (9 have not been completed), in only 2 cases was the mitigation project complete prior to the destruction of the original wetland. These extensive time lags represent losses of at least 1 to 3 growing seasons per project. In none of the projects reviewed was the loss of resource functioning time considered as a value requiring compensation.

It appears that much of the lag time between project impact and mitigation completion is due to delays in initiating the mitigation project. The time lags will diminish as mitigation projects are initiated earlier. Unless mitigation is completed prior to destruction of the original wetland, this functional loss of habitat will continue to occur and should be acknowledged in the negotiations.

Mitigation Designs Were Not Effectively Incorporated Into the Final 404 Permits

Documentation of mitigation plans and their various components within the §404 permits was inconsistent and incomplete. Figure 7 illustrates how these permit specifications varied over the seven year period. In only 22 (63%) of the 35 issued permits requiring mitigation was the concept of mitigation even mentioned. Resource agencies required monitoring studies in 18 cases (51%), but in only 11 permits (31%) were the monitoring requirements actually documented.

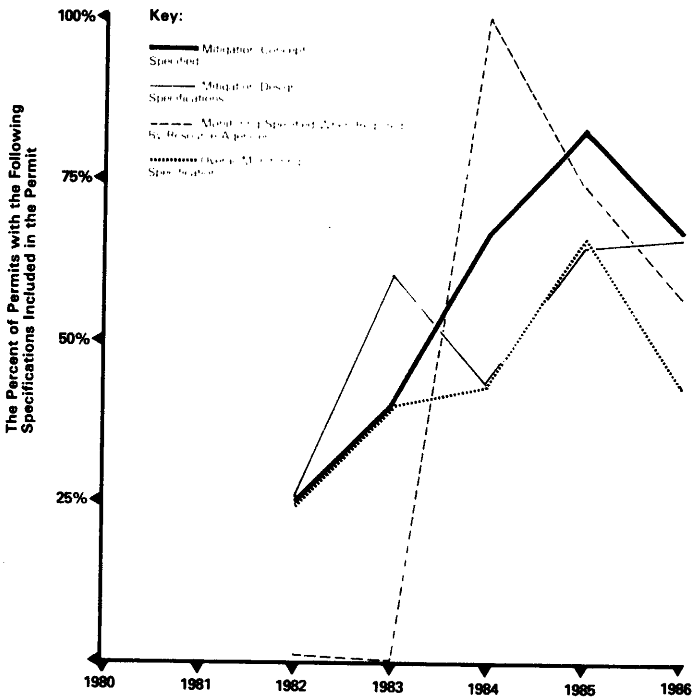

Figure 7
Permit Specifications

Nineteen of the 35 permits (54%) included some type of design criteria. No criteria were mentioned prior to 1982. The following is a list of the various design criteria included in the permits reviewed:

- ° acreage of the mitigation
- ° location
- ° methods of construction
- ° objectives (mitigation goals)
- ° completion deadlines
- ° authorization deadlines (to make application for mitigation within a certain timeframe)
- ° contingency planning (if certain performance standards are not satisfied)
- ° maintenance requirements
- ° performance bonds

If any of the above design criteria were incorporated in the written permit, permit blue print, or blue print notes, it was included in this tally (Figure 8). No permits included all of the above criteria and 16 permits (46%) did not specify any. The design criteria most frequently incorporated into the permit were area, location, and construction methods. Though these are potentially the easiest to enforce, they do not determine or guarantee ecological success.

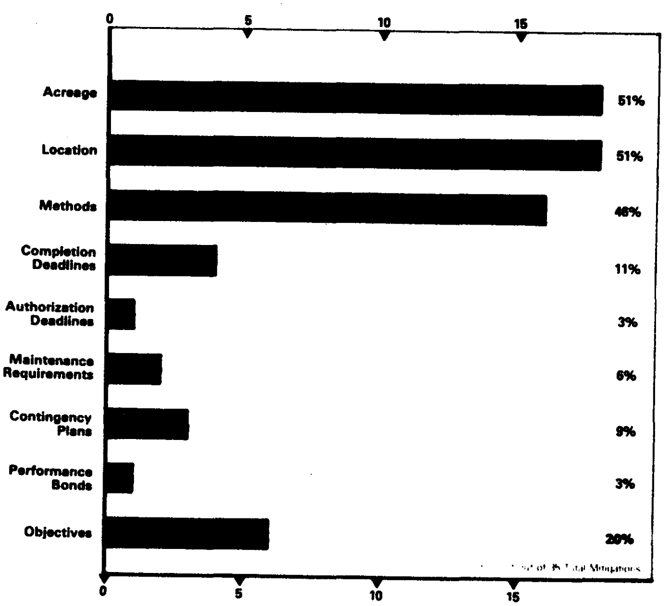

Figure 8
Total Number of Times Mitigation Design Criteria Were Specified in the 404 Permit

Routine Follow-up For Mitigation Projects is Lacking

Since wetland restoration and creation is still an experimental science, there are expectations that even the best intentioned and designed projects may not function as planned. Clearly, routine follow-up in the form of compliance tracking, project monitoring for success/failure, and contingency planning are needed.

There was no routine procedure for tracking mitigation compliance. This may have contributed to the fact that 5 mitigation projects were never constructed or restored as negotiated. Monitoring was required in 18 of the 35 mitigation projects. Contingency planning was only required in 3 of 35 projects. Monitoring studies applied within the mitigation process should trigger the use of contingency plans as needed, and yet these two project components were rarely required together.

To date, there is no consistent, standardized process for negotiating, planning, implementing, or evaluating wetland mitigation projects. No single agency, federal or state, maintained comprehensive records of wetland mitigation projects. The information contained in federal agency Section 404 project files was dispersed and incomplete. Monitoring results were sparse, construction/restoration completion dates were inconsistent, and the degree of functional success was rarely documented.

The fragmented regulatory design of the Section 404 program invites this chaos. The multiple agency approach may offer some check and balance advantages, but it inhibits effective project management with respect to wetland mitigation. Either (1) an assemblage of regulatory agencies must monitor, analyze results, and respond to inadequacies; or (2) this responsibility must be assigned. Zedler (1986) cites the importance and advantage of an adaptive management-type approach which would allow the developer some freedom to determine the most effective methods in meeting mitigation goals while still fulfilling specific requirements. In view of the awkward alternative, this recommendation may be a good one. But in order for regulatory agencies to learn from the process, they must remain informed through efficient project follow-up--a prerequisite to adaptive management.

RECOMMENDATIONS

Our understanding of wetland ecosystems is far behind that needed to consistently replace lost wetlands. The rates of mitigation failure and wetland losses are higher than we might expect. This may be partially attributed to the failure of regulatory and resource management agencies to adequately negotiate, plan, and track mitigation projects.

Filling of wetlands will continue until the public places a higher value on the resource. Because of this fact, regulators will continue to require mitigation to offset the losses. We do not recommend that all mitigation work be abandoned; as Race (1985) states "the technology is an important tool for balancing the demand for coastal development with the need for the conservation of wetland habitats." However, we do recommend significant improvement in the mitigation process to ensure a higher degree of success for created/restored wetlands and to reduce net losses. Mitigation negotiations have improved over time, but serious inconsistencies still exist.

Several authors have recommended procedures for the development of mitigation plans which, in theory, might improve our chances of resource replacement both at the negotiated level and in reality (Race, 1985; Race and Christie, 1985; Baker, 1985; Zedler, 1986; Harvey and Josselyn, 1986; and Cooper, 1987). We have compiled their recommendations and our own in the following list of criteria for mitigation projects:

ELEMENTS NEEDED IN MITIGATION PLANS

1. Ecological Assessment of Wetland(s) To Be Lost Through Development. It is critical to understand the chemical, physical, and biological interactions of a system in order to replace it.

2. Statement of Goals. The mitigation goals should include a discussion of the functions and values lost, and those planned for replacement.

3. Methods. The questions of what, where, when, and how should be answered, i.e., acreage of mitigation; wetland habitat type(s) to be constructed/restored; location; dates for beginning and completion of the project; methods of construction; and maintenance requirements. Ensure fair compensation in both time and space, by requiring more acreage for replacement than that lost to development.

4. Standards of Success. A qualitative, and to the extent possible, a quantitative description of what will be considered a successful, functioning wetland must be included.

5. Monitoring Strategy. Design a monitoring system to determine whether or not the mitigation goals and standards of success are met.

6. Contingency Plan. If the mitigation should fail or only partially succeed, a plan outlining possible restorative measures is necessary. A performance bond should be included to ensure the applicant's compliance with the terms of the mitigated agreement.

To ensure that all the above criteria are included, a single, structured format for the development of mitigation projects should be developed. Lack of information and conflicting agency goals for wetlands regulation have resulted in the haphazard and ineffective process existing today. All information and all requirements and monitoring results pertaining to mitigation projects should be located in at least one comprehensive file system. Similarly, a complete mitigation plan should be attached and alluded to within the 404 permit from which the agreement originated.

Mitigation plans should be discussed with all agencies and groups involved with the permitting process. Agencies' goals for wetland habitats sometimes differ. It is important to consider these during negotiations.

And finally, it is important for policy and decision makers to realize that mitigation is not "the only answer" to development/preservation conflicts. We must fully understand the limits of the technology of wetland creation. Some habitats (e.g., bog systems) cannot be replaced within our short time references. This understanding must be reflected in the implementation of public policy.

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Wetland Types After Cowardin, Et.Al. (1979)

Major Wetland Types	Sub-types	Qualifiers
E = Estuarine	I = intertidal	US = unconsolidated shore
	S = subtidal	RS = rocky shore RE = reef AB = aquatic bed UB = unconsolidated bottom
R = Riverine	T = tidal	US = unconsolidated shore
	U = upper perennial	EW = emergent wetland RB = rock bottom
	L = lower perennial	UB = unconsolidated bottom AB = aquatic bed
L = Lacustrine	L = littoral	UB = unconsolidated bottom US = unconsolidated shore AB = aquatic bed EW = emergent wetland RS = rocky shore
PA = Palustrine		SS = scrub shrub EW = emergent wetland FW = forested wetland AB = aquatic bed US = unconsolidated shore

Summary Statement

Charles A. Simenstad*

This session, addressing *Habitat Modification: What is the Status of Puget Sound Habitats and Can We Create Habitats to Mitigate Loss?,* was designed to go beyond the minutiae of each researcher's studies and encompass the larger scale question of the significance of habitat loss to the Puget Sound ecosystem. And it was for this reason that the session included several papers from state and federal resource scientists, because they are often the sole interpreters of the scientific evidence justifying habitat protection and management.

The first paper, by George Blomberg from the Port of Seattle, described the historic changes in the shorelands of our immediate back yard, the Duwamish River estuary, since before the turn of the century. As one of our most industrialized corners of Puget Sound, we think that it is important to understand the processes by which we have altered our shoreline; not that we can return to the pristine setting of the 1800s, but so that we can more effectively design and site critical restoration and enhancement projects in the context of the estuary's original structure. In this sense, it is important to examine not only habitat loss, but also changes in the estuary which equally affect its ability to function, e.g., water discharge rates and patterns.

In a paper with a similarly historical perspective, Ian Hutchinson from Simon Fraser University integrated anthropogenic changes in intertidal wetlands with the naturally dynamic of these habitats in the diverse Puget Trough estuaries. Only through the knowledge of these processes can we effectively allocate our limited resources toward wetland habitat preservation and restoration. In this case, allogenic changes over the Quaternary, and through future climatic change, superimpose additional considerations on wetland management, especially when applying priorities to the management and rehabilitation of brackish versus saline marshes.

Given these descriptions of the extant and distribution of wetland habitat loss, do we know enough about wetland function to evaluate the significance of these and continued losses to Puget Sound? Mary Ruckelshaus addressed her studies of but one wetland function, that of food web support linking the production and import of organic matter in estuaries to the growth and production of consumer organisms. An important aspect of this paper was the experimental nature, which implied that such functions can be compared both among habitats within an estuary as well as between habitats among different estuaries.

Current and proposed procedures to inventory and manage our surviving wetlands were addressed in two papers by two wetland ecologists in the Washington Department of Natural Resources. Linda Kunze described the purposes, approaches to wetland inventory, and management strategies of the Natural Heritage Program. One of the most important points to keep in mind in evaluating this paper is that their survey of 244 wetlands encompassed less that 10% of those estimated to exist in the Puget Trough lowlands and, of these, only 16 or less than 1% were considered "nearly pristine"; is there a mechanism to inventory and secure protection for these rare, disappearing wetland? Tom Mumford addressed the approach, objectives, and governmental mandate behind his Department's program to manage marine plants on state-owned aquatic lands; that is, those owned in common by the populace of this state. The phenomenal pressures to exploit aquatic plants and their habitats demand that we better understand their distribution and function.

The final papers in the session turned specifically toward the topic of wetland mitigation, i.e., our ability to restore or construct wetland habitats and the scientific bases behind this ecotechnology. The implementation of mitigation was presented as two different, and some might suggest extreme, approaches to compensation for wetland loss. Greg Hueckel[1] of the Washington Department of Fisheries prepared an eloquent argument for artificial reefs as functional mitigation for rocky habitat loss. The obvious success of the Department's artificial reefs as a valid strategy for replacement of rocky habitat loss is not, however, valid justification for substituting artificial reefs as mitigation for loss of non-rocky habitats. Both in terms of habitat and strategy, the paper presented by Dave Shreffler, of the Wetland Ecosystem Team at Fisheries Research Institute, University of Washington, described a different approach to wetland restoration of estuarine habitat in Commencement Bay. He described their approach to measuring just one wetland function, that of the habitat's utility as a foraging area for juvenile salmon. If there is one critical conclusion from this study, it is that considerable research effort and long-

[1]Greg Hueckel was unable to make his presentation at the symposium, but it is included here as an important contribution to the selected papers in this session.

term experimentation is required to functionally test the efficacy of wetland mitigation projects. Although these presentations addressed wetland mitigation project functions, the reader should remember that these are not representative of the normal evaluation criteria required in assessing wetland mitigation success.

As anchorperson to this session, Kathy Kunz of the U.S. Environmental Protection Agency, Region X office, described the somewhat startling results of their recent evaluation of just what has been required to gain approval for wetland mitigation projects in the region. In essence, it was a sordid description of a general lack of documentation, negotiation, evaluation, or post-construction monitoring specifications. Certainly, much of this can be attributed to the infancy of the technology; but, how is the science to evolve and improve if the mandate to implement the (CWA, Section 404) law and police mitigation projects is not exercised?

In assessing the implications of these papers, and the session as a whole, please consider the early state of our knowledge about the way that wetland habitats function, and the detailed, long-term studies required to gain such a "functional" understanding before the technology of wetland rehabilitation and creation can be effectively applied.

TOXIC CHEMICALS AND THEIR EFFECTS

DO WE HAVE GOOD INDICATORS OF PROBLEMS?

Edward R. Long,
National Oceanic and Atmospheric Administration
Session Chair

Introductory Statement

Edward R. Long*

Chemical analyses of environmental samples provided needed information on the nature and degree of contamination of selected areas by potentially toxic compounds and elements. However, they do not provide information on the consequences of the contamination to biological organisms. Indications of "problems" are often dependent upon the demonstration of some biological effect to answer the rhetorical "So what?" question. Accordingly, all the papers in this session involve the development of biological measures of environmental quality, as opposed to chemical measures.

All represent attempts to develop a better understanding of how the measures respond to contaminants in the environment. All attempt to refine existing tolls to increase their sensitivity and utility. All also attempt to relate biological measures either made in the laboratory or hypothesized by ecologists to measures observed in the field. All have been, or could, be applied in Puget Sound. Similar efforts to evaluate and refine biological measures of effects are underway elsewhere in the United States, but much of the pioneering work has been and is being conducted in Puget Sound by local scientists. Many methods initially used here are now being employed elsewhere by others. The Puget Sound region has become a leading center of expertise in sophisticated measures of environmental quality.

The five papers presented in this session involve testing of three environmental media: Water, sediments, and fish. The first two papers describe methods of testing the toxicity of water; the first involving a method of evaluating the toxicity of single compounds, and the second describing a test of complex mixtures in the field or laboratory. The third paper describes an attempt to isolate the effects of organic enrichment versus those of toxic chemicals in sediments upon the communities of bottom-dwelling organisms. The fourth and fifth papers describe research in which measures of the health of bottom-dwelling flatfish are described and related to either contamination of the fish or the sediments.

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Herring Embryos as Indicators of Marine Pollution

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Introduction

Developing embryos are one of the most sensitive stages to the adverse effects of contaminants. Investigators here and in other regions have successfully employed sea urchin and oyster embryos in bioassays of pure chemicals and complex mixtures. In addition, several species of fish embryos have been used to study the carcinogenic and teratogenic potential of numerous compounds. These systems have proven to be useful laboratory models, but they have not been used for in situ monitoring. In this paper we will describe the utility of Pacific herring embryos for both in situ and in vitro testing of environmental contaminants.

Pacific herring (Clupea harengus pallasi) are widely distributed in Puget Sound. They serve as forage for many predatory species, including salmon, and are sold commercially for bait. In addition, new aquaculture industries utilizing herring roe are emerging in the Pacific Northwest. In Puget Sound, herring spawn from early February through late May. If one considers the entire West Coast, however, spawning adults can be obtained from November until June. The eggs of Pacific herring measure 1.2-1.5 mm after exposure to seawater and hatch in 10-14 days, dependant upon temperature. The eggs are sticky and adhere, in Nature, to eelgrass, kelp, other seaweeds and rocks in nearshore areas. The adhesiveness of the eggs makes them especially attractive for experimental use, because they can be spawned directly onto artificial substrates such as microscope slides. The eggs can be fertilized while attached to the slides and the slides can then be placed in the field for in situ testing, or manipulated in the laboratory for in vitro testing. Because the eggs are hardy, one can transport the slides to field sites or examine the slides microscopically without damaging the developing embryos. Because the eggs can be placed on the slides in regular arrays, one can recognize individuals and can maintain accurate measurements of fertilization rate, embryo mortality, etc. Because the chorion is transparent and because the eggs are relatively large, one can visually monitor such events as embryonic development, eye formation, heart beat, and movement.

We have successfully used herring embryos both in the laboratory (Kocan et al., 1988; Westernhagen et al., 1988) and in the field (Kocan, 1988) to test complex environmental mixtures. In this paper we will describe the results of a recent pilot study that was conducted in response to observed herring mortalities.

The Washington Department of Fisheries (WDF) has conducted and reported the results of herring spawning surveys since 1972. These reports document high mortality in herring egg sets at certain locations in Puget Sound including Tulalip Bay, an area near Dockton in Quartermaster Harbor, the south shore of Port Madison and the east shore of Port Gamble Bay (Pentilla et al., 1985). In the latter two sites, mortalities ranging from 20 to 100% have been observed, most often affecting eggs during the first five days of incubation. The WDF reports indicate that this heavy, unexplained mortality is not attributable to predation, suffocation, dessication or thermal stress, all known to cause mortality in herring spawn. They also state that other spawning areas within Puget Sound rarely show such extensive mortality and never with the regularity that occurs at these two sites.

Although the observed mortalities are severe at specific sites, the severity varies from season to season. Eggs deposited outside the affected sites but within the same spawning area appear to develop normally.

In order to evaluate the possible cause(s) of the observed mortalities, a study was undertaken in Port Gamble Bay to evaluate the biological activity of water and sediment from three sites within the bay and to compare these with similar samples collected outside the affected spawning area. Sample sites were selected on the basis of WDF spawn mortality estimates. The study was performed using herring embryos fertilized in the laboratory under controlled conditions. The method used was a modification of the procedure developed by Rosenthal and Alderdice (1976) to evaluate herring roe requirements for maximum survival and normal development.

Materials and Methods

Sample Collection: Water samples were collected from three sites within Port Gamble Bay. These stations, designated by WDF as Sites 25, 30, and 33 (Pentilla et al., 1985), were selected because they often exhibit heavier than normal embryo mortality. At the time the present study was conducted, mortality rates were 30% (Site 33), 80% (Site 30), and 90-100% (Site 25) (Yake and Norton, 1987). The samples were collected in glass containers and transported to the University of Washington School of Fisheries where they were stored at 9 C until used as incubation medium for newly fertilized herring eggs. A field control water sample was also collected from a site (Bywater Bay) well outside Port Gamble Bay for comparison with those samples taken within the affected area.

Sediment samples (0.5 kg) were collected with a Van Veen grab from the same four sites. Two hundred grams of each sample was extracted by vigorous shaking with 1 liter of synthetic seawater (27 ppt). The extract was then sterilized by passage through a 0.45 um filter and stored at 9 C until used as incubation medium for laboratory spawned herring embryos. Sites 25 and 30 were pooled during collection and were subsequently treated as a single sample.

Herring Embryos: Sexually mature adult herring were obtained from a commercial bait dealer and returned live to the School of Fisheries. Eggs from a single ripe female were squeezed onto glass slides (20-25 per slide) and fertilized with sperm from two males. Twelve hours after fertilization the slides were labeled and examined to determine the number of fertile eggs per slide. Five slides were then allocated to each of the experimental groups. A complete description of the technique can be found in Kocan et al. (1988).

Eggs were examined every other day to determine mortality rates and hatching success. Live hatched larvae were collected daily, anesthetized in methane tricaine sulfonate (MS 222), preserved in 10% neutral buffered formalin and examined microscopically for the presence of physical defects--primarily defects of spine, eye and skull development (Rosenthal and Alderdice, 1976).

Field Exposures: From March 27-March 31 (Exposure I) and from March 31-April 3 (Exposure II) a minimum of 100 newly fertilized eggs (5 slides; 20-25 eggs/slide) were suspended 2-3 feet from the bottom at each site. This was accomplished by placing five slides into glass slide carriers such as those used for histologic processing, covering the carriers with fine mesh screen to exclude predators, and attaching these to a mooring line that was anchored to the bottom and marked with a buoy for retrieval. Eggs maintained in synthetic seawater at the School of Fisheries served as a laboratory control that could be compared with the field control.

Following the exposure period, the slides were retrieved and returned to the laboratory for evaluation of embryo survival. This was done by examining each slide under 60 X magnification and calculating embryo mortality (%). Eggs from Exposure II were maintained for an additional period of time in the laboratory and then evaluated for hatching success (% live hatch as calculated from the number of live embryos present at the time the slides were retrieved from the field) and developmental defects (% abnormal larvae as calculated from the total live hatch).

Laboratory Exposures to Water and Sediment Extract: Five slides, each containing 20-25 newly fertilized eggs, were placed into 5 ml of water or sediment extract for incubation (100+ eggs/exposure). The eggs were observed microscopically prior to hatching to determine the mortality rate and the hatched larvae were counted for calculations of hatching success (% live hatch). Following hatching, the larvae were also examined for the presence of physical defects (% abnormal as calculated from the total live hatch).

Results

Field Exposures: In Exposure I, significantly elevated embryo mortality was recorded from Sites 25, 30 and 33 immediately after retrieval from the field. In Exposure II, significantly elevated mortality at the time of retrieval was recorded only from Site 33; however, embryos from all three sites ultimately experienced reduced hatching success and elevated deformity rates relative to controls. There was no statistical difference between the laboratory and field controls, even though there was a slight decrease in live hatch and an increase in the number of abnormal larvae in the field controls (Figs. 1 and 2).

Laboratory Exposures to Water: Water from all three Port Gamble sites and from the control site produced no difference in hatching success (93-98%) for embryos exposed continuously from 24 hours post fertilization until hatching. Water from Sites 30 and 33 did, however, produce a significant increase in physical defects (63% and 42%, respectively) compared to Site 25 and the field control (15% and 10%, respectively). Figure 3 summarizes the results of the laboratory exposures to water.

Laboratory Exposures to Sediment Extract: Seawater extracts of sediment from Sites 25/30, Site 33 and the field control site produced no difference in embryo survival (95-99%) following continuous exposure from 24 hours post fertilization until hatching. Sediment extracts from Sites 25/30 and Site 33 did, however, produce a significant increase in deformity rates (26% and 22%, respectively) as compared to the field control (3%). Figure 4 summarizes the results of the laboratory exposures to sediment extract.

Discussion

The data obtained from field and laboratory exposures suggested that some type of water soluble toxic substance is present in Port Gamble Bay which can either cause embryo mortality or induce teratogenic defects.

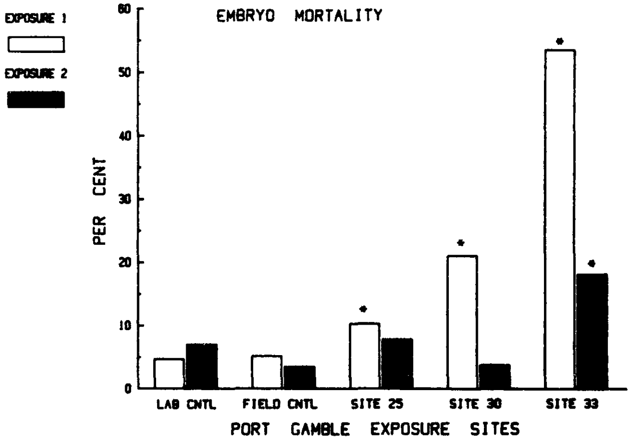

Figure 1. Laboratory observations were made on field-exposed embryos 24 hours following their return to the laboratory. Exposure I (96 hr) resulted in significantly elevated mortality rates from Sites 25, 30, and 33, with Site 33 being the most severely affected (54% mortalities). Exposure II (72 hr) resulted in significant embryo mortalities only from Site 33 (27% mortalities). Field controls exposed at Bywater Bay and laboratory controls exposed to synthetic seawater produced no significant difference in embryo survival during the same exposure period. (*=P< 0.01; Chi Square)

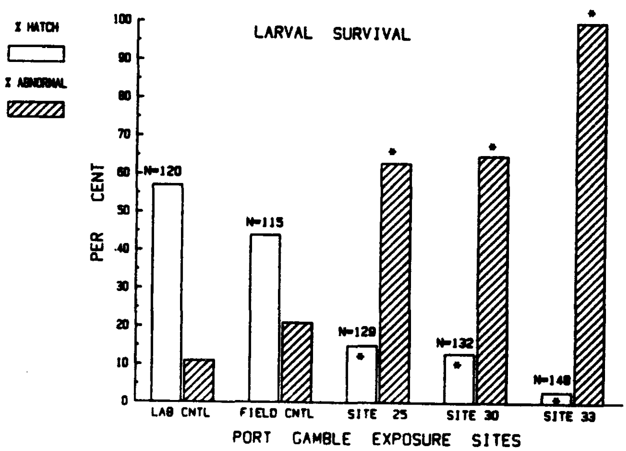

Figure 2. Embryos from field Exposure II (72 hr) completed development in synthetic seawater in the laboratory. Hatching success (%, based on total number of viable eggs at the end of the field exposure) was significantly reduced at Sites 25, 30 and 33, with Site 33 being the most severely affected. The frequency with which abnormal live larvae were produced was also significantly increased from these same three sites. (*=P< 0.01; Chi Square).

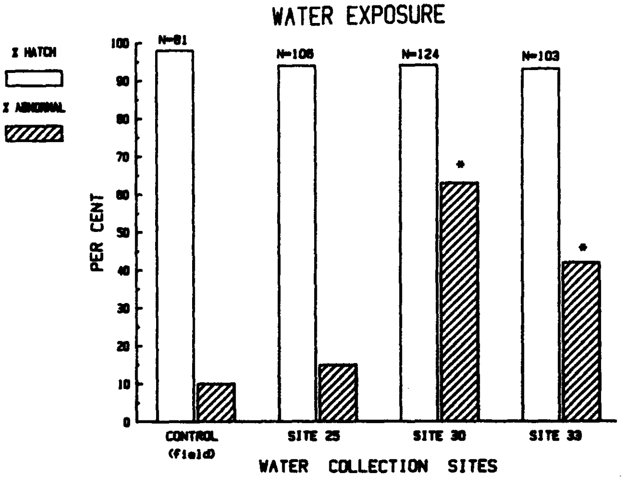

Figure 3. No difference in hatching success was observed in herring embryos exposed to water collected from three Port Gamble sites when compared to Bywater Bay. There was, however, a significant increase in the number of physical defects in live larvae that hatched from eggs incubated in water from Sites 30 and 33. (*=P<0.01; Chi Square)

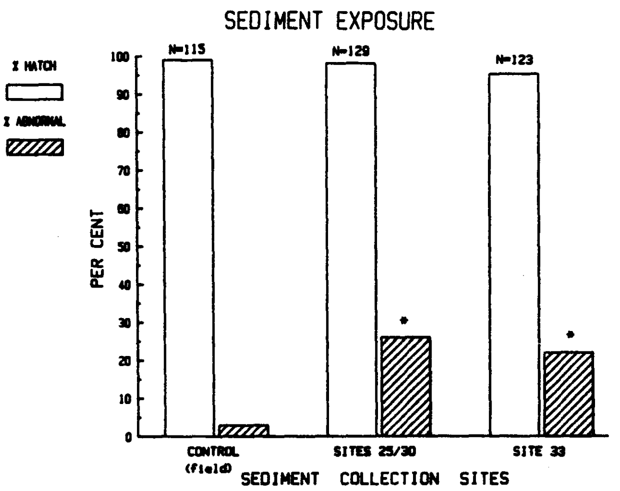

Figure 4. No differences in hatching success were observed following exposure of herring embryos to seawater extract of sediment collected from Port Gamble or Bywater Bay. Exposure of embryos to sediment extract from Sites 25/30 and Site 33 did, however, result in a significant increase in the number of physically defective larvae compared to that produced by the field control extracts. (*=P<0.01; Chi Square)

Since mortality and larval defects are nonspecific responses to numerous physical and chemical insults, it was not possible to implicate specific substances in Port Gamble Bay without conducting more complete biological and chemical analyses of the water and sediment. The study did show, however, that herring embryos are sensitive to environmental toxins and that they can be used as test organisms in both the laboratory and the field.

The suitability of herring embryos for use in toxicity studies depends upon their being carefully spawned under controlled laboratory conditions. By spawning the eggs onto glass slides as described here, it is possible to control the number, density and spacing of eggs in relation to each other, thus allowing for direct comparison of different exposure groups. Previous studies have shown that the spatial relationships of herring eggs to each other influences their rate of development and survival to hatching. Toxicity studies in our laboratory with herring eggs have shown that data cannot be compared from one exposure group to the next unless all eggs are exposed to the same physical conditions. These include water to egg ratio, depth of water over eggs, number of eggs in contact with each other, and air circulation at the water surface. We found the optimum conditions to be two rows of eggs at the water surface and 20-25 eggs/5 ml seawater. This combination allowed for the maximum number of eggs to be exposed in the minimum volume of water without affecting normal development.

Being able to conduct an on-site toxicity evaluation with the species being affected is an obvious benefit when it comes to interpretation of the results. No extrapolation from species to species is necessary and only those components of the environment known to naturally contact the organims need be examined. Disadvantages to using a species such as herring include the fact that their availability is seasonal and that capturing ripe adults for spawning is not always possible. It is also not possible to evaluate a specific site on a continuous basis unless samples are collected and stored until herring are available, a process which might introduce problems associated with changes in the chemical composition of the sample arising during from storage.

The short duration of the present study precluded any prediction of whether the toxic phenomenon is short-lived (e.g. seasonal) or continuous. Since field observations indicate that herring embryos have been affected over a period of several years, it can be assumed that toxicity did not result from an isolated occurrence such as a spill. If the toxicity is continuous or occurs intermittently during the year, we would not be able to detect it after the spawning season unless some other indicator species were used. To do this it would be necessary to run parallel exposures of herring embryos and an alternate species, compare their responses, then use the alternate during periods when herring were not available. This, of course, would require selection of a species which is available throughout the year.

The cause of the observed toxicity could be continuous run-off from a terrestrial source, leachates from some previous contamination source, a product of the day-to-day operation of the saw mill at the mouth of the bay, or metabolites of natural algal blooms. This last possibility is supported by a recent report (Aneer, 1987) which relates Baltic herring embryo mortality to exposure to exudate from the filamentous brown alga Pilayella littoralis. Baltic herring mortalities occurred only during the period of heavy spawn deposition and did not appear to be related to anoxia as was previously suspected. A similar situation may be responsible for the observed mortality seen in Port Gamble and other sites within Puget Sound.

Summary

Herring embryos appear to be a sensitive indicator for detecting the presence of some toxic substances in the marine environment. They can be used in the laboratory as well as the field provided that they are spawned in such a way as to ensure uniformity in egg density, spatial arrangement and egg to water ratio. The major drawback to their use is limited availability, a problem which can be alleviated either by storage of samples throughout the year or by use of an alternate species provided parallel comparative studies are performed in advance of their use.

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A New Approach for Modeling Response to Toxicants[1]

James Jay Anderson

Introduction

A fundamental problem in toxicology is how to quantify and predict the effects of chemicals on organisms in their natural environment. At present, toxicity data derived in laboratory experiments are commonly used in simple dose-response models to predict the environmental effect of toxicants. The models contain large uncertainties because they include little or no information on the conditions of the environment. In this paper, I outline an improved dose-response model that combines standard population survival data with the standard toxicity data.

The Present Approach

The basis of all toxicity analysis is the standard 96-hr LC_{50} test, which determines the concentration of a toxicant that kills 50% of a population with a 96 hr exposure (Stephan, 1976). Over the past decade, much effort has been put into refining the methodology by focusing on the need for statistical integrity, the virtues of different experimental chambers, the choice of appropriate test species, and how to use the resulting information to set acceptable environmental levels of a toxicant. Toxicity testing and regulation has become an industry of its own involving: government regulators; commercial, university, and government laboratories; and a plethora of consultants. The success of this industry was virtually assured from the beginning with the establishment of various congressional laws and regulations, the design of simple and relatively inexpensive testing methodology, and the acceptance of a simple regulatory decision procedure based on a single number, the LC_{50}. Many harmful chemicals have been identified and regulated using the LC_{50} method. A major problem arises, though, when LC_{50}s are used to set acceptable environmental levels of toxicants. Because the method is so simple, its ability to predict many environmental responses is highly questionable.

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In the current methodology, mortality of a population under chronic low-dose toxicant exposure is predicted using information from short-term, high-concentration experiments (Bickis and Krewski, 1985). Extrapolations to low doses are controversial, so regulators seek conservative models to assess risk. The simplest, and perhaps the most common, model is known as the quotient method, in which a toxicant is judged to be of risk if the ratio of the expected environmental concentration to the LC_{50} exceeds some value. This value is usually set arbitrarily to be on the order of 0.01 (Suter, 1986). The main reason for using the quotient method is based on the claim that, in many cases, there is not enough information to use other methods. However, there is, in fact, a large amount of additional information to bring to bare on the problem.

A New Approach

The new approach I present here combines, through a mathematical model, laboratory-derived toxicity information with field observations of population survival. A comparison of survival in field and laboratory conditions clearly illustrates that the force of mortality operate at vastly different rates in the two cases. While laboratory experiments are designed to kill 50% of a population in four days, in the natural environment a 50% mortality may take days to years depending on conditions and species (Fig.1). If the rate of mortality and survival in the two cases can be expressed through a single model, we have the potential of extrapolating between the two cases to predict the less drastic effects of low toxicant exposures on population survival.

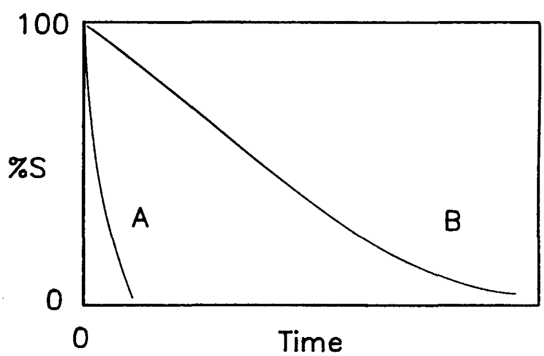

Figure 1. Survivorship curves for a population exposed to high toxicant concentration in laboratory conditions (A), and for a population in its natural environment (B).

To describe the rate of mortality and the survivorship curve we apply a model based on an abstract measure of an organism's "health". The basic idea is that, over time, the health of an organism increases and decreases in response to the beneficial and deleterious events that it experiences, with mortality occuring when its health goes to zero. The problem of expressing the rate of mortality thus is reformulated in terms of the organism's health. The dynamics of health is modeled through a stochastic differential equation known as the Weiner process (Goel and Richter-Dyn, 1974). In the equation, the rate of change of health is described by a deterministic loss rate, which is dependent on the toxicant concentration, and a stochastic loss rate, which fluctuates over time, randomly taking on positive and negative values.

Figure 2 depicts two examples of random paths of organism health over time. A path begins with some initial value and ends when the health reaches zero. Each path is different, but the average properties of an ensemble of paths is determined by two parameters: Y, the initial health divided by the intensity of the fluctuations in the rate of change in health and R, the average rate of change in health divided by the intensity of the fluctuations in the rate of change in health.

The survivorship curve of a population, representing the percent of a cohort alive as a function of the cohort age, is also described in terms of the initial health and the health loss rate. Two basic types of survivorship curves

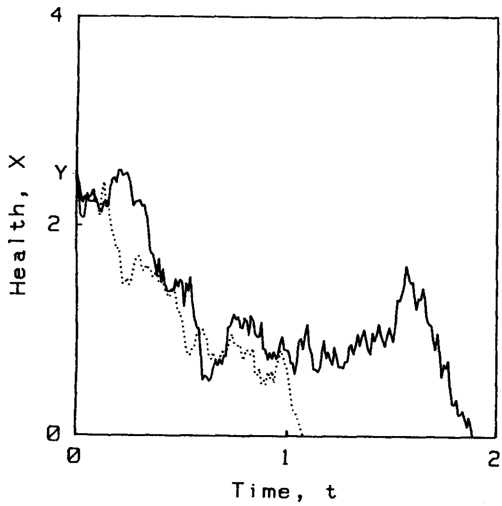

Figure 2. Random paths of organism health over time.

are generated with the model: If Y is large, most organisms die at an old age, while if R is large, most organisms die at a young age (Fig. 3). The initial health, Y, is a boundary condition, expressing the health before an organism is exposed to the toxicant. The loss rate, R, is dependent on the environment, and in particular, the toxicant concentration in the environment. The model assumes that different factors contribute to the rate of health loss. A base rate is ascribed to organisms in their natural environment. The effect of a toxicant is represented by an additional rate that is a function of concentration. This rate, which is added to the base rate, is described by a formula that uses two parameters to quantify the toxicity of the chemical.

To apply the model, we use the data to estimate the initial health, the base health loss rate, and the two toxicity parameters relating the toxicant induced health loss rate to toxicant concentration. The initial health and base health loss rate can be obtained by a regression to natural survivorship data. The two toxicity parameters can be estimated by a regression of dose-response data. In this manner, information from the laboratory and field is extracted from data and rigorously combined in the model to predict the effects of a toxicant on organisms in their natural environment. In general, by using nonlinear regression techniques, the model provides good fits to laboratory and field data. Figure 4 illustrates the fit

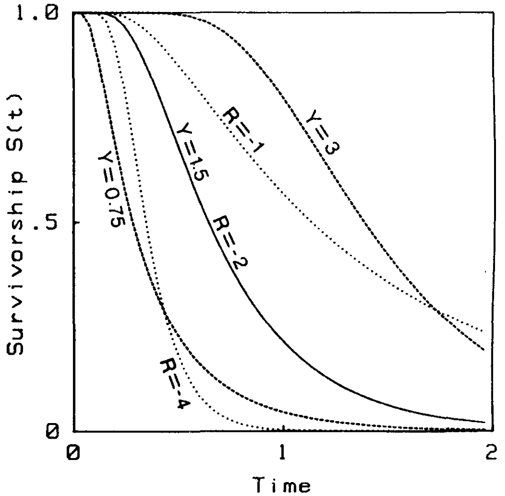

Figure 3. Survivorship curves as function of model parameters: Y = initial health, and R = health loss rate.

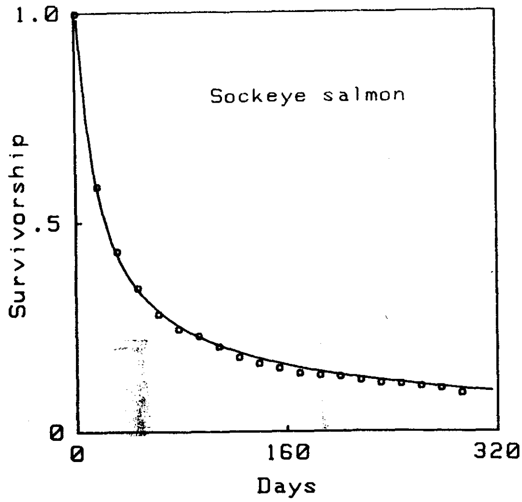

Figure 4. Regression of model to survivorship data of juvenile sockeye salmon (data from Foerster, 1938).

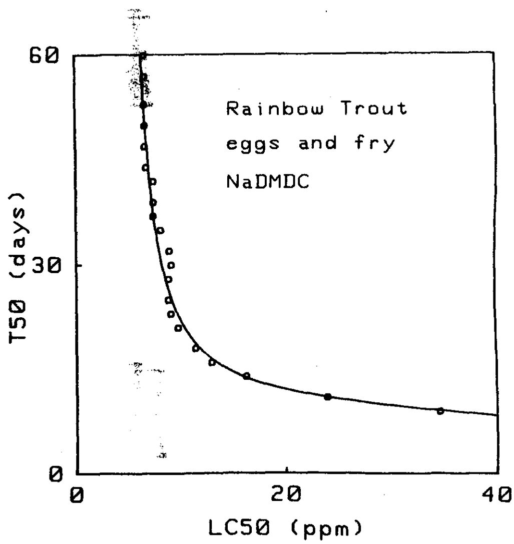

Figure 5. Regression of model to dose-response information of time to 50% mortality vs. concentration for rainbow trout exposed to a dithiocarbamate (data from Van Leeuwen, Espeldoorn and Mol, 1986).

of the model to survivorship data for juvenile sockeye salmon (*Oncorhynchus nerka*) in Cultus, British Columbia (Foerster, 1938). Figure 5 illustrates the fit of the model to data from a dose-response experiment of egg and fry rainbow trout (*Salmo gaidneri*) exposed to a dithiocarbamate (Leeuwen, Espeldoorn and Mol, 1986). The regression in Fig. 5 is time to 50% mortality vs. toxicant concentration.

Summary

In summary, through the health model we can describe low and high concentration dose-response curves over short- and long-term intervals in terms of four model parameters: the initial health and a base health loss rate, which are estimated from survivorship data; and two toxicity factors, which are estimated from dose-response data. Present toxicity assessment practices generally estimate toxicity from laboratory data only. The approach outlined in this paper combines field and laboratory data and should provide more realistic predictions of environmental toxicity.

Acknowledgments

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Measuring the Effects of Organic and Toxicant Inputs on Benthic Communities

Donald P. Weston*

Attempts to measure the impact of man's activities on the marine environment have relied to a large degree on the monitoring of the seafloor and the benthic invertebrates which inhabit these sediments. There are three principle advantages to using the benthos in pollution monitoring. First, many of the pollutants of greatest concern are associated with sediment particles, and thus ultimately are deposited on the seafloor. Secondly, many benthic invertebrates are sedentary and therefore must tolerate the chemical and physical environment, rather than move in and out of an area as conditions allow. Finally, the composition of benthic communities represents the integral effect of pollutants over a period of months to years, and is an indication of both acute and chronic impacts of pollutants.

While benthic community monitoring has proven to be valuable in pollution assessment, its full potential has not yet been realized. Near catastrophic community changes can readily be detected by any number of measurements. However, if tools were available to detect more subtle community changes our monitoring capabilities would be much enhanced and remedial actions could be initiated prior to catastrophic disturbance.

Of all the various forms of pollution, the impact of organic enrichment is the best understood. Conceptual models are available which predict the trends in community stucture and function which are generally observed along a gradient of organic input (Pearson and Rosenberg, 1976; 1978; Gray, 1979). One of these models is illustrated in Figure 1. The model represents a composite of observations made in the vicinity of many sources of organic input including pulp mills, a seaweed processing plant, and sewage discharges. The model predicts a wide variety

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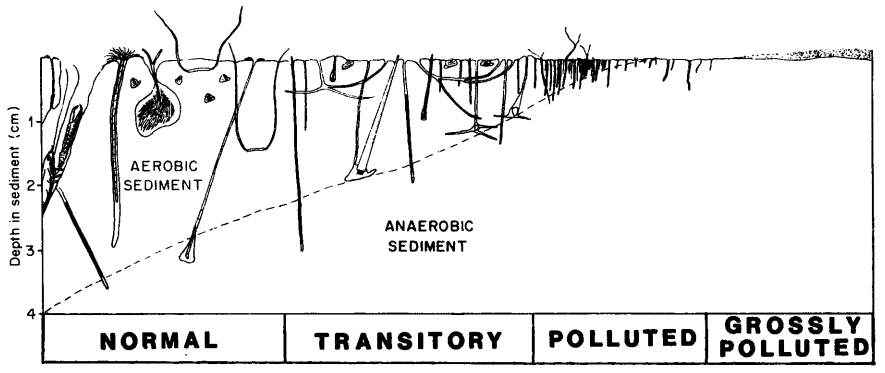

Fig. 1. Diagrammatic representation of faunal changes under increasing organic loading. From Pearson and Rosenberg, 1976.

of structural and functional changes in benthic communities subject to excessive organic input. These changes, and their utility in existing pollution monitoring programs, include:

- o <u>Decrease in the number of species</u> - routinely used in pollution monitoring;
- o <u>Increase in abundance of a few opportunistic species</u> - routinely used in pollution monitoring;
- o <u>Decrease in total biomass</u> - occasionally used in pollution monitoring;
- o <u>Decrease in average individual size</u> - rarely, if ever, used in pollution monitoring;
- o <u>Decrease in the variety of feeding strategies employed</u> - used in only a few studies, and in a simplified fashion by the infaunal trophic index (Word, 1978);
- o <u>Decrease in the depth of the sediment column occupied by the benthos</u> - not generally used, except in a qualitative manner by the REMOTS camera system (Rhoads and Germano, 1982);
- o <u>Decrease in the proportion of motile organisms</u> - rarely, if ever, used in pollution monitoring.

Current pollution monitoring programs utilize only a small fraction of the information available in benthic community structure and function, and the capacity to detect disturbance may be enhanced by measuring more of the changes predicted by existing models. My research has three primary goals:

1) to quantify some of the qualitative changes which the models predict are likely to occur in benthic communities subject to high organic input;

2) to use the model predictions in developing alternative and more sensitive indicators of pollution stress in benthic communities;

3) to develop conceptual models for pollution by toxicants comparable to models currently available for organic enrichment.

This research program remains in progress, and definitive results are still several months away. A much more detailed presentation of data is anticipated upon completion of the study. This paper is intended to serve as an introduction to some of the concepts being evaluated, and provide a glimpse of some of the encouraging preliminary results obtained to date.

Materials and Methods

Two study areas were chosen for intensive field sampling. The first area was Clam Bay at the south end of Bainbridge Island. This bay is the site of several commercial salmon farms in which the fish are raised within floating net-cages. The rain of feed and feces to the seafloor has created sites with a high rate of organic input, but with few, if any, associated toxicants. The farm of interest is a relatively large facility consiting of 160 pens, each 6 x 12 m, producing 617 metric tons of coho salmon annually. The second study area was Eagle Harbor on the east side of Bainbridge Island. Discharge of creosote over many years from the Wyckoff wood-treatment facility has resulted in sediment contamination with a variety of polynuclear aromatic hydrocarbons (PAHs). Eagle Harbor served as a site of toxicant pollution for purposes of contrast with the organic enrichment of Clam Bay.

In both study areas five sampling sites were established along a distance gradient from the pollution source (Figure 2). Sediment sampling was conducted in June of 1987 by use of a 0.06 m^2 spade box corer. Each core was subdivided into up to six vertical strata as permitted by total depth of penetration (0-1 cm, 1-2, 2-5, 5-10, 10-20 and >20) and each stratum processed separately. Three cores were taken at each site for analysis of benthic invertebrates. All strata above 5 cm were sieved on stacked 0.5 and 1.0-mm screens; strata below 5 cm were sieved on only the 1.0-mm screen (except for one core at each station for which both screens were used for all strata). The material retained on the screens was pre-erved in 5% buffered formalin stained with Rose Bengal, and later transferred to 70% ethanol. All organsms were identified to the lowest practical taxonomic level, generally species. Subsamples were taken from all cores for physical and chemical analysis. These analyses included grain size distribution, priority pollutant trace

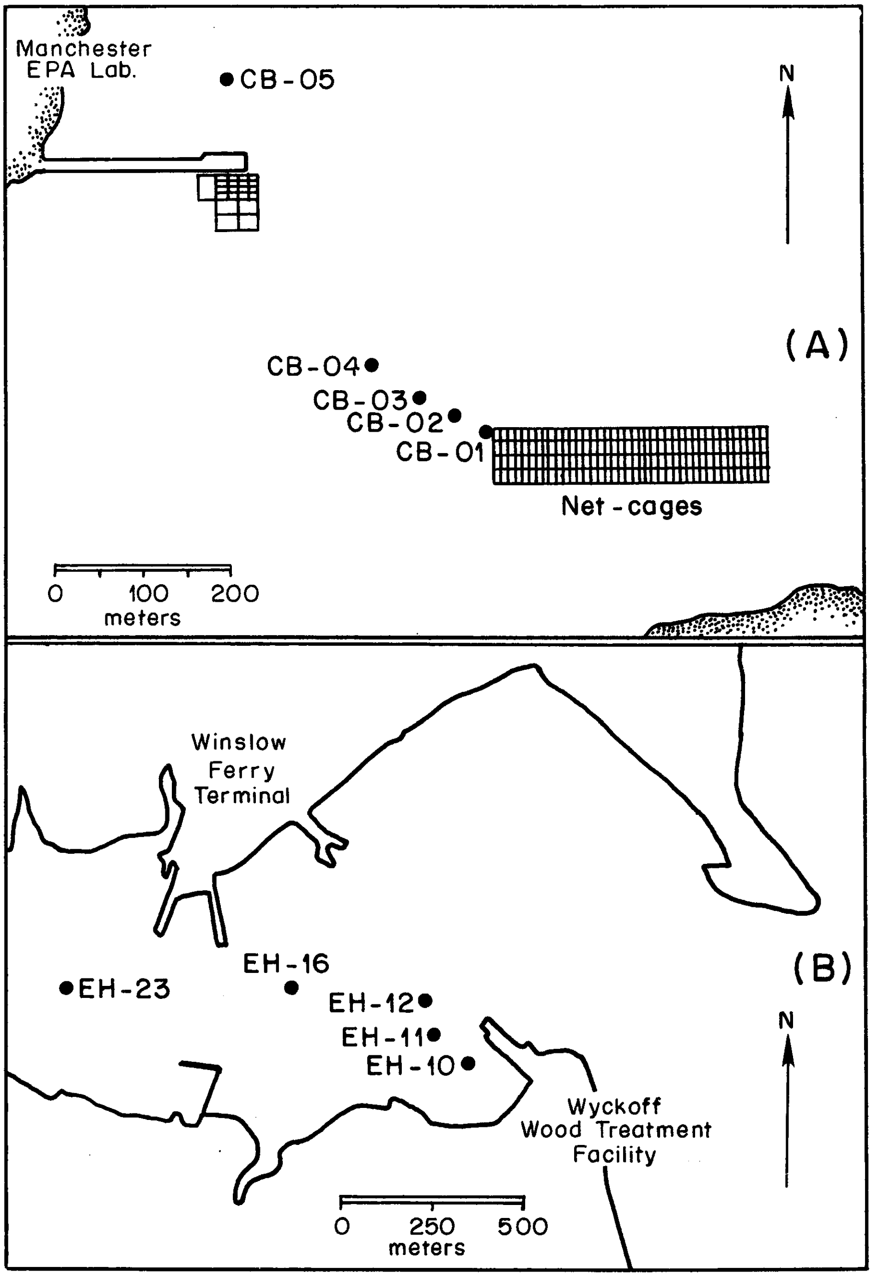

Fig. 2. Location of sampling sites: (A) Clam Bay; (B) Eagle Harbor.

metals, total organic carbon (TOC), total nitrogen (TN), and dissolved sulfides. A fourth core was collected at all Eagle Harbor stations and analyzed for PAHs. Chemical analyses were conducted separately on each vertical stratum as for the biological samples.

The biological results reported here are based entirely on the polychaetous annelids, for identifications have only been completed for this group. This group alone comprised about 75% of the total macrofaunal individuals collected. PAH data for Eagle Harbor are not yet available at the time of this writing.

Results and Discussion

Site characteristics

The Clam Bay sampling sites were located in 13-20 m water depth (MLLW) and the sediments were composed of 92-97% sand. There were no consistent trends in trace metal concentrations among the stations. Sediment chemistry analyses indicated organic enrichment to a distance of 45 m from the net-cages (Station CB-02). Concentrations of TOC, TN and dissolved sulfides were 3-6 times greater in sediments at the perimeter of the net-cages than in sediments from CB-03, 04 and 05. Mats of the bacterium, *Beggiatoa*, a visible indicator of organic enrichment, were apparent to a distance of at least 100 m from the net-cages (Station CB-03).

The Eagle Harbor sampling sites were located in 8-11 m water depth (MLLW) and the sediments were composed of 65-92% silt and clay. Sediments at EH-10 had a greater concentration of many trace metals (Ag, As, Cd, Cu, Ni, Pb, Zn) than did the other Eagle Harbor sites, although the increase in concentration was generally small (about 1.5-fold). Concentrations of TOC and TN were 1.5-2 times greater at EH-10, 11, and 12 than at EH-16 and 23. The stations were deliberately chosen to reflect a dramatic gradient in the concentration of PAHs. Although samples from this study are still being processed, there was visible evidence of gross creosote contamination at EH-10 and 11, and moderate contamination at EH-12. Previous sampling (Black and Veatch, 1986) indicated a 50-fold increase in PAH concentration from EH-23 to EH-10.

Evaluation of mesh sizes

The relative advantages of the 0.5-mm (or smaller) and 1.0-mm mesh sizes have been the subject of much discussion in the Puget Sound area. Preliminary results of this study indicate that the 1.0-mm screen size was effective at retaining only about 50% of the total polychaete individuals (8,100 of the 16,500 individuals). This proportion

TABLE 1
Percentage of polychaete individuals retained on a 1.0-mm mesh sieve.

Depth stratum (cm)	Clam Bay[1] Stations 01-04	Eagle Harbor Stations 10-23
0-1	27	15
1-2	19	34
2-5	62	63
5-10[2]	81	75
10-20[2]	61	57
>20[2]		43

[1] Excluding CB-05 where the core penetrated only to 10 cm.

[2] Individuals retained on the 0.5 screen estimated from one replicate. All other data are mean of three replicates at each station.

varied with depth in the sediment column, for the coarser mesh size was much more efficient in sampling the polychaetes at depths greater than 2 cm than it was in the 0-2 cm stratum (Table 1). These samples were obtained in June when there was likely to be a high proportion of new macrofaunal recruits in the community, and the 1.0-mm screen may retain a greater proportion of the total individuals during other times of the year. For certain species, however, (e.g., Exogone spp., Sphaerosyllis spp., Euchone incolor) the coarser mesh size retained only a negligible proportion of mature individuals, and is likely to be ineffective regardless of the time of sampling. It is clear that the 1.0-mm screen does not adequately sample the total macrofauna, but the question of whether detection of pollutant impacts is impaired by use of a 1.0-mm rather than a 0.5-mm screen must await completion of the species identifications from this study and further analysis of the data.

Trends in species richness, abundance and biomass

Spatial trends in species richness and infaunal abundance have long been a standard tool in pollution monitoring. Trends in biomass do not routinely receive attention, but have been examined on many occasions (e.g., the work of Brown, et al., 1987 under Scottish salmon net-cages). Species richness, abundance and biomass exhibited predictable trends near the Clam Bay net-cages, but showed rather confusing trends near the Eagle Harbor wood-treatment operation (Figure 3). The number of polychaete species showed a clear increase with distance from the net-cages, consistent with the predictions of Pearson and Rosenberg's (1978) model. The abundance data, however, does not support the model prediction of a mode representing the high abundances of a few opportunistic polychaetes. The opportunistic species were present, for Capitella capitata alone comprised 99% of the individuals at CB-01. As the density of C. capitata decreased with distance from the net-cages, the density of several other species increased resulting in an essentially unchanged total polychaete density between CB-01 and 04.

In Eagle Harbor only EH-10 showed any suggestion of a decrease in number of species or biomass. The traditional measurements of species richness, abundance, and biomass appeared to be relatively insensitive to changes in benthic community structure along a gradient of creosote contamination.

Vertical distribution of infauna

Most benthic samples in Puget Sound have been collected with a Van Veen grab, and the entire contents of each grab is composited as a single sample. Sampling in this manner results in the loss of all information on the vertical distribution of the infauna in the sediment. The sampling protocol of this study has two principal advantages over traditional techniques. First, the spade box corer penetrates to about twice the depth of the Van Veen grab. The corer collected many types of deep-burrowing infauna (e.g., sipunculans, echiurans, geoducks and several other bivalve species) that were under-represented or completely absent in Van Veen grab sampling done concurrently for a separate project. Secondly, vertical sectioning of the core sample retained information on the vertical distribution profile for each individual species and for the infaunal community as a whole. The sampling design allows the examination of changes in these vertical distributions along gradients of organic enrichment and toxicant concentrations.

It is apparent even from the analyses done to date that many species were restricted to discrete vertical intervals in the sediment column (Table 2). Sphaerosyllis brandhorsti primarily occupied the upper 0-2 cm stratum of the sediment, Tharyx multifilis occupied mid-depths of 2-10 cm, and Notomastus tenuis was distributed throughout the sediment column but with slightly higher densities in

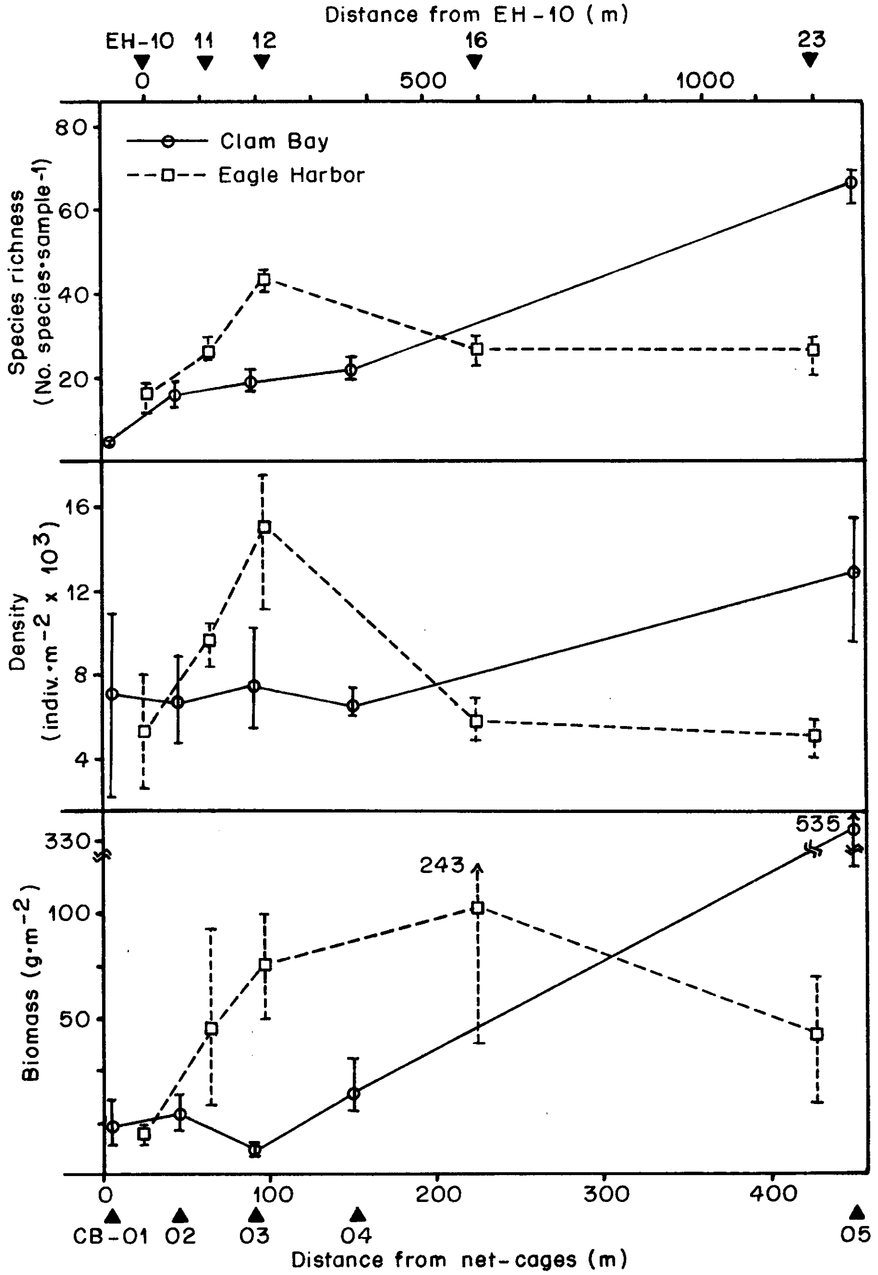

Fig. 3. Trends in polychaete species richness, abundance and biomass in Clam Bay and Eagle Harbor. Mean and range of the three replicate cores indicated for each station.

TABLE 2
Vertical distribution of three selected species

	Number of individuals • 1000 cm^{-3}		
Depth Stratum (cm)	Sphaerosyllis brandhorsti	Tharyx multifilis	Notomastus tenuis
0-1	50	5	0.2
1-2	11	18	0.07
2-5	2	39	0.2
5-10	0.2	12	0.7
10-20	0	0.3	0.3
>20	0	0.2	0.02

the 5-20 cm stratum. Such data have important implications if attempting to compare data taken with different sampling gear which penetrate to different depths in the sediment, such as a grab and box corer. Both devices adequately sample S. brandhorsti, but both T. multifilis and N. tenuis would be under-represented in the grab sample.

Existing models of the impact of organic enrichment on the benthos predict that as the organic loading decreases, organisms should occupy a progressively deeper portion of the sediment column. The ramifications of this pattern in pollution monitoring efforts have never been fully exploited, and the initial results from this study are intriguing. In subsequent figures of vertical abundance and biomass distribution (Figure 4 and 5) it is important to recognize that, as indicated by the dashed lines at each cm boundary, the figures portray the percentage of the total abundance (or biomass) collected in each centimeter stratum. The 5-10 cm stratum at CB-05 for example, contained 20% of the total polychaetes at that station, therefore it is arbitrarily assumed that each of the five 1-cm strata within this interval contained 4% of the total abundance. This method of portraying the data is necessary since the sampling strata varied in thickness, and it would mislead the reader to show the data in any way other than a per cm basis.

The vertical distribution of polychaetes at the Clam Bay sites (Figure 5) indicated that most individuals were restricted to the upper 5 cm at all stations, yet a large

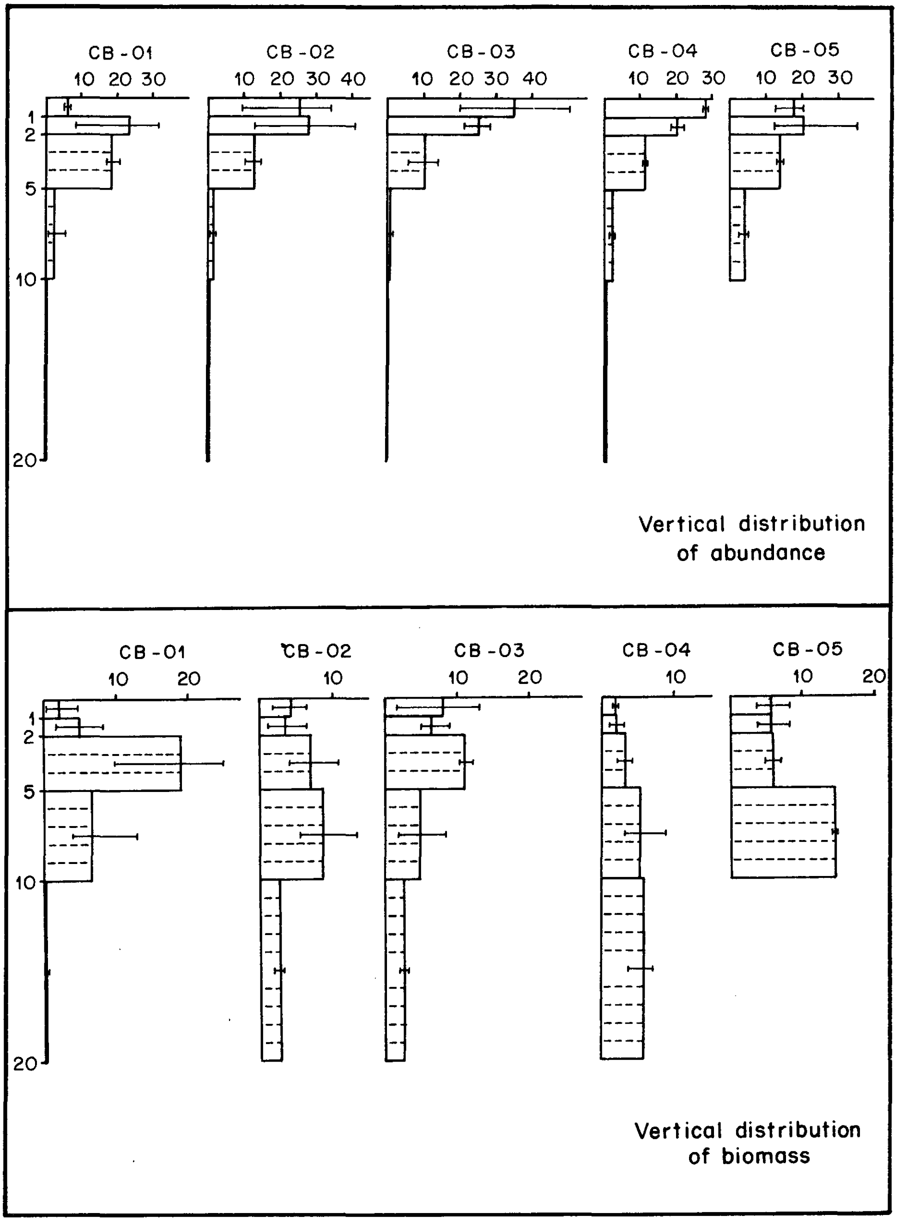

Fig. 4. Vertical distribution of polychaetes at the Clam Bay study sites as indicated by the percentage of total abundance or biomass within each centimeter stratum. Mean and range of the three replicate cores indicated for each depth stratum. Lengths of vertical axes indicate average depth of penetration (cm) of the replicates.

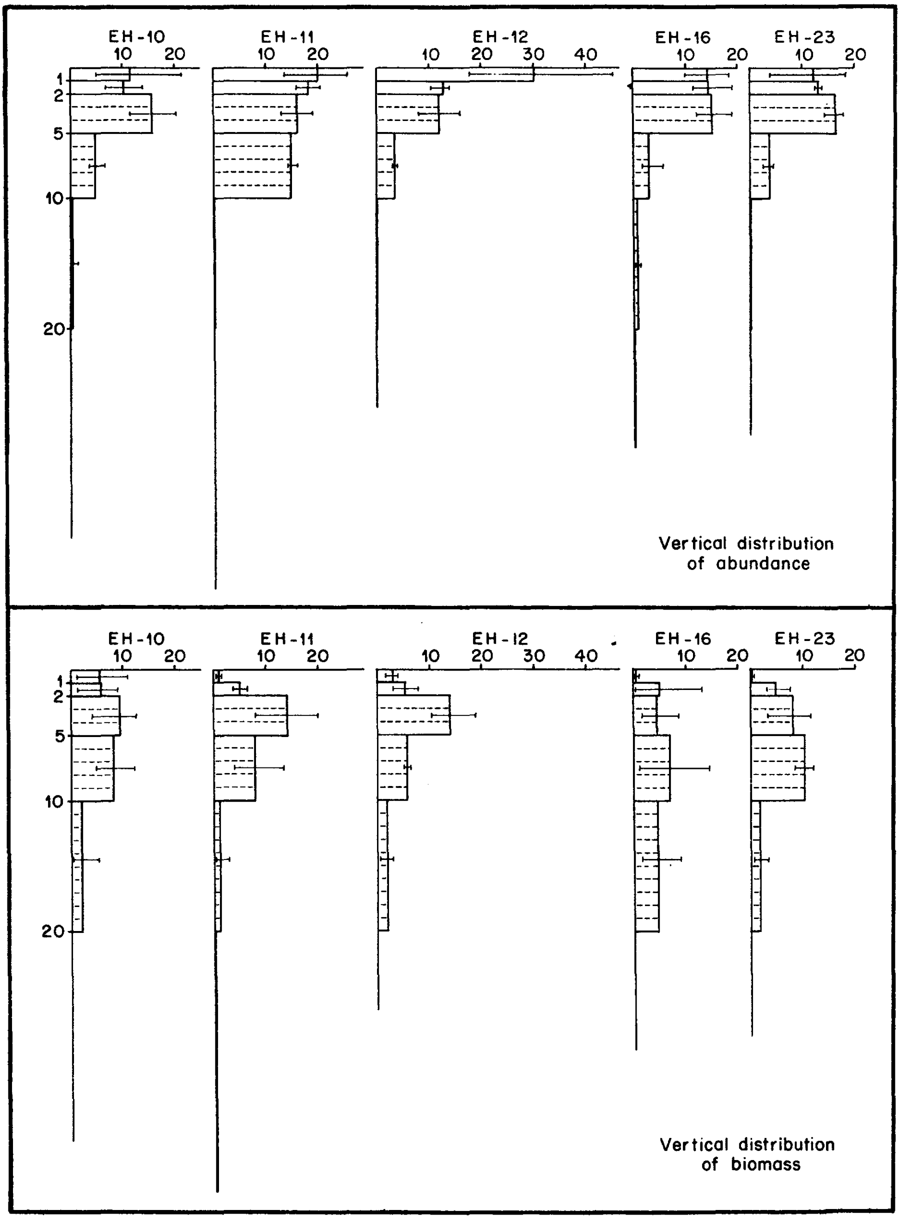

Fig. 5. Vertical distribution of polychaetes at the Eagle Harbor study sites as indicated by the percentage of total abundance or biomass within each centimeter stratum. Mean and range of the three replicate cores indicated for each depth stratum. Lengths of vertical axes indicate average depth of penetration (cm) of the replicates.

proportion of the polychaete biomass lay between 5 and 20 cm. On the basis of the abundance data, there appeared to be little evidence that polychaetes were occupying a progressively greater proportion of the sediment column along the Clam Bay organic gradient. There was, however, a pronounced increase in the proportion of the total polychaete biomass in the deeper strata with increasing distance from the net-cages. This trend was most evident in the proportion of biomass in the 5-10 cm stratum (4.8-8.7%$\cdot$cm^{-1} at CB-01 through CB-04; 14.4%$\cdot$cm^{-1} at CB-05) and in the 10-20 cm stratum (0.3%$\cdot$cm^{-1} at CB-01; 5.9%$\cdot$cm^{-1} at CB-04). Thus, while decreasing organic input did not result in proportionately more polychaetes in the deeper strata, the polychaetes in these deep strata were proportionately larger. The Clam Bay observations can be explained in terms of the mechanisms by which organic enrichment effects the benthos. The organic enrichment in Clam Bay and the subsequently high sediment oxygen demand in the vicinity of net-cages (Pamatmat, et al., 1973; Pease, 1977; Enell and Löf, 1983) would limit the vertical diffusion of oxygen into the sediments. This reduction in sediment dissolved oxygen concentrations would be particularly limiting to the large, deep-burrowing infauna, precisely the group lacking at CB-01, 02 and 03. Smaller organisms, because of their greater surface area to body volume ratio, would be less affected. If verified in further analysis of the data, the vertical distribution of biomass may prove a useful tool in pollution monitoring programs.

In the fine sediments of Eagle Harbor the corer was able to penetrate to greater depths than in Clam Bay (22-40 cm), yet very few organisms were present in these deeper strata (0.03-0.2%$\cdot$cm^{-1} at depths >20 cm). Most of the individuals were found in the upper 5 cm, yet sampling to a depth of 20 cm was necessary to collect most of the biomass (Figure 5). Unlike Clam Bay, there was no apparent trend in vertical distribution of abundance or biomass with distance from the source of creosote. Thus, based on the polychaete data alone, vertical distribution initially appears to be an insensitive indicator of PAH contamination. Further interpretation of the Eagle Harbor vertical distributions must await completion of the PAH analyses currently in progress.

Individual size

Pearson and Rosenberg's (1976; 1978) model predicts that the average macrofaunal individual will have a progressively larger body size as the rate of organic input to the sediment is decreased. Both the Clam Bay and Eagle Harbor data sets exhibit a trend towards increasing individual biomass with increasing distance from the pollution source (Figure 6). This trend is consistent with the model despite the fact that only polychaete data are presented

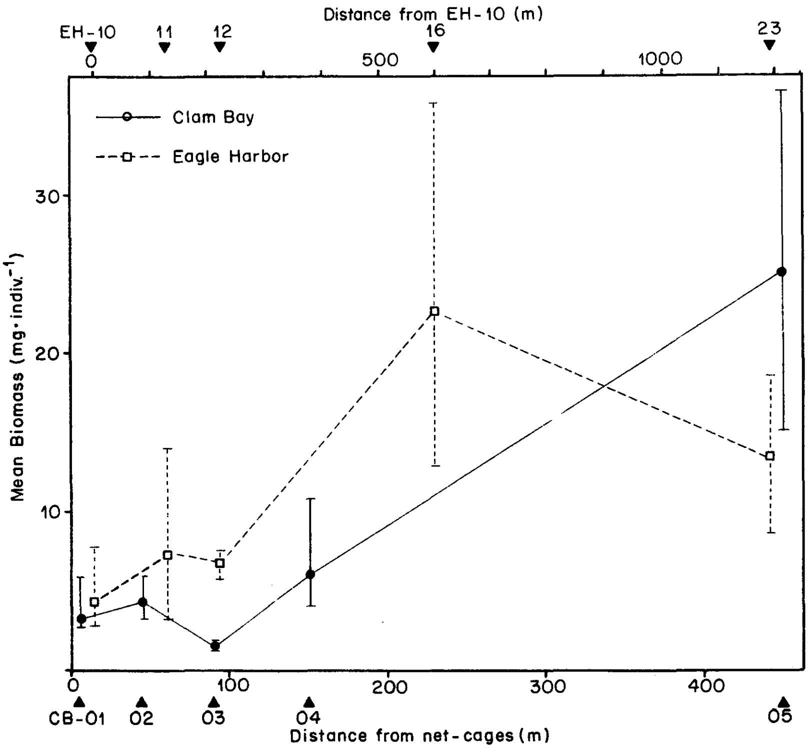

Fig. 6. Trends in average individual biomass of polychaetes along pollution gradients in Clam Bay and Eagle Harbor. Mean and range of three replicate cores indicated at each station.

from this study. The trend is likely to be even more apparent after all species identifications are completed, and some of the phyla having a larger average individual size are included in the analysis (e.g., bivalves, echiurans).

One potential explanation for reduced body size in organically enriched areas is the resulting maximization of the surface area to volume ratio, thereby enhancing absorption of oxygen across the body wall. Such an adaptation, however, would prove disadvantageous in a toxicant enriched environment if absorption through the body wall were a significant route of toxicant uptake. The fact that the sediments with the greatest visible creosote contamination contain polychaetes with the smallest body size indicates surface area related diffusion of toxicants is not the primary factor controlling body size. The mechanism for reduction of body size in toxicant polluted environments will be further explored following completion of the biological and chemical analyses.

Conclusions

This work will provide some of the information needed in making methodological choices such as the type of sampling device to use, the desired depth of sampling, and the choice of mesh size. These issues are incidental outcomes, however, and the primary goals remain to contrast the benthic impacts of organic enrichment and toxicants, and to improve the capacity to measure the effects of both types of pollution. Many traditional measurements such as gross structural characteristics of benthic communities (e.g., number of species and individuals) or integrating indices (e.g., diversity) are little more than the tip of the iceberg when it comes to understanding how benthic communites respond to pollution. When relied upon in pollution monitoring programs they can, at best, identify areas of near catastrophic disturbance. At worst, they will fail to detect more subtle changes when remedial action would be most advantageous. They may also create concern when in fact some environmental parameter totally unrelated to pollution may be the underlying cause.

It should be clear from this presentation of preliminary results that pollution impacts can be much better understood and potentially measured with a higher degree of sensitivity by considering under-exploited indicators of pollution disturbance. Individual body size is one such indicator. The vertical distribution of infauna is particularly intriguing not only as a pollution monitoring tool, but because of its implications for a wide variety of other areas of scientific investigation including bioturbation, nutrient regeneration, sediment-water exchange of toxicants, and resource partitioning. Shifts in the relative proportion of various feeding guilds are predicted by existing models to be still another potential indicator of pollution disturbance. While scarcely mentioned in this presentation, it will be the subject of detailed analysis at a later date.

Acknowledgements

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Using Bioindicators to Assess Contaminant Exposure in Flatfish from Puget Sound, WA§

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Introduction

Puget Sound is a large marine and estuarine ecosystem with several urban centers located along its over 2100 km of shoreline. Several areas of Puget Sound adjacent to these urban areas are contaminated with a variety of chemicals of anthropogenic origin, and a need exists for scientifically sound methods of monitoring this contamination and its effects on the biota of Puget Sound. Generally, environmental monitoring programs assess contaminant exposure and its effects on marine organisms primarily by direct measurement of contaminants in sediments and tissues and by determining associated biological damage (*e.g.* tumors, or neoplasms) in tissues of animals (Malins *et al.*, 1984). However, several types of chemical contaminants, such as aromatic hydrocarbons (AHs), are now known to be extensively biotransformed by many marine species after their uptake from sediment (Stein *et al.*,1987), and the myriad biotransformation products cannot be easily identified and quantitated by current methods (Varanasi *et al.*, 1987). Accordingly, efforts are now being made to evaluate the exposure of marine fish to xenobiotics by using appropriate biochemical methods which do not rely on detection of the parent compounds.

One bioindicator of interest is the induction, or increase, of mixed-function oxidases (MFO) in contaminant-exposed organisms. MFO are a group of diverse enzymes which both activate and detoxicate a variety of contaminants and are also involved in a wide

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§ Sections of this paper have been excerpted from - Collier, T.K. and U. Varanasi. 1987. Biochemical indicators of contaminant exposure in flatfish from Puget Sound, WA. *Proceedings OCEANS '87 Conference.* IEEE Ocean Engineering Society, Piscataway, N.J. pp. 1544-1549.

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range of normal physiological processes, such as steroid metabolism. Aryl hydrocarbon hydroxylase (AHH) is a MFO which is essential in the activation of AHs to metabolites which can modify, or damage, DNA. Such modification of DNA is believed to be an essential step in the initiation of neoplasia. Field studies worldwide have confirmed the induction of AHH activity both as an indicator of contaminant exposure and as one of the earliest biological effects seen after exposure of fish to contaminants (Payne *et al.*, 1987), and we have been studying this enzyme in Puget Sound flatfish (Collier *et. al.*, 1986, Varanasi *et al.*, 1986, Collier and Varanasi, 1988) The measurement of fluorescent aromatic compounds (FACs) in fish bile is another biochemical indicator which can be used to assess exposure of marine fish to organic contaminants (Krahn *et al.*, 1984). This method is currently in use in NOAA'S Status and Trends Program, as part of the National Benthic Surveillance Project, a multi-year effort to characterize the levels of contamination, and associated biological effects, in the Nation's coastal regions. Another useful indicator of contaminant exposure is the determination of DNA modification by a sensitive technique known as ^{32}P-postlabeling. The modified DNA measured by this method appears to be largely due to xenobiotic exposure (Varanasi *et al.*, 1988; Reichert and Varanasi, 1987). This paper describes the results obtained when each of these bioindicators were used to assess contaminant exposure and early biological responses in flatfish from Puget Sound, WA. The data demonstrate how these indicators can complement each other in monitoring programs aimed at assessing the environmental quality of our coastal environments.

Strategy and Design of Studies

Before using any indicators of contaminant exposure in field studies, it is important to first conduct rigorous and credible laboratory studies to determine if the proposed indicators are indeed responsive to known amounts of contaminant exposure. This can be most directly done by conducting dose-response studies, using the species of interest and suitable chemical contaminants. It is also important to determine the time response and reversibility of the indicators to be tested, especially if the indicators may be used in any studies of the effectiveness of remedial actions. Accordingly, we have performed several studies to determine the time- and dose-responses of the bioindicators described above, after exposure of Puget Sound flatfish to either an organic-solvent extract of a contaminated Puget Sound sediment, or a representative carcinogenic AH, benzo[a]pyrene (BaP). The results of some of these studies are given below.

The next step in developing suitable bioindicators of contaminant exposure is to test them in field applications. Puget Sound is an excellent ecosystem for such initial field studies, for the following reasons:

--Within a relatively small distance, areas of low, moderate, and high contamination of bottom sediments can be found, and accordingly easily sampled.

--One species of flatfish, the English sole, is found in virtually all areas of Puget Sound, and it appears that this species often carries out its entire life cycle within Puget Sound.

--A large data base is already available concerning the levels of chemical contamination found in many areas of Puget Sound, together with associated biological effects seen in English sole.

Accordingly, we have begun field studies in Puget Sound to test the usefulness of several bioindicators, and the design of these studies is also described below.

Current Results and Discussion

Figure 1 shows the dose-response curves for induction (*i.e.* increases) of hepatic AHH activity and increases in levels of FACs in bile of juvenile English sole exposed to an extract of sediment from the highly contaminated Duwamish Waterway in Seattle. Dose-response data for hepatic DNA modification are thus far only available for juvenile English sole exposed to BaP, and these results are also shown in Figure 1. It was interesting to note that AHH activity showed a good dose response at the lower doses used, whereas the levels of FACs in bile increased linearly with dose at the upper end of the curve. These data suggest that the combined use of both of these indicators in a monitoring program could provide useful information about contaminant exposure over a wider range of exposure levels than could be obtained using either indicator alone. As described below, field studies are being conducted which will allow us to test this hypothesis. The data for DNA modification show that this indicator is also dose responsive, but further work is needed to allow a comparison of this indicator with AHH activity and levels of FACs in bile.

The data shown in Figure 2 depict the time responses of each of these indicators after exposure of juvenile English sole either to an extract of a contaminated sediment (Duwamish Waterway), in the case of AHH activity and levels of FACs in bile, or to BaP, for the data on xenobiotic modification of DNA. Based on these data, it can be said that both AHH activity and levels of FACs in bile are very quick to respond to contaminant exposure,

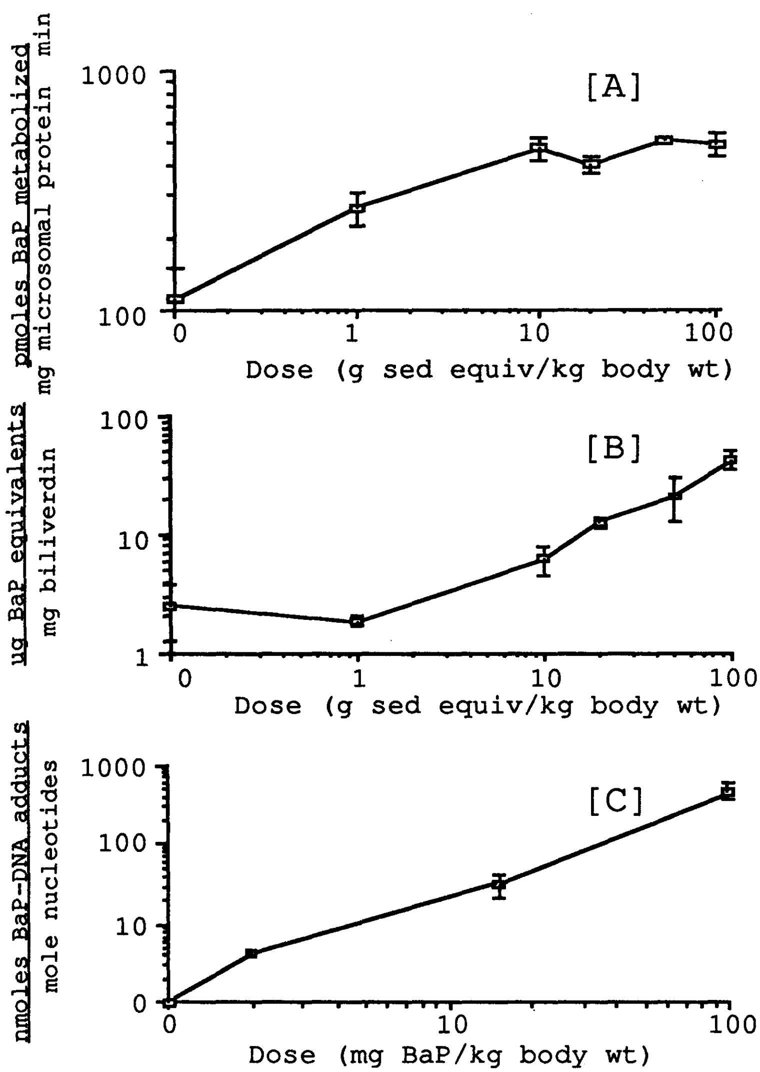


Figure 1. Dose-response curves for [A] hepatic aryl hydrocarbon hydroxylase activity, [B] levels of fluorescent aromatic compounds in bile, and [C] levels of hepatic DNA adducted with BaP metabolites, in juvenile English sole exposed via intramuscular injection to various doses of either an organic-solvent extract of sediment from the Duwamish Waterway ([A] and [B], sampled 7 days after exposure) or BaP ([C], sampled 1 day after exposure). Data from Collier and Varanasi, 1987 and Varanasi et al., 1988.

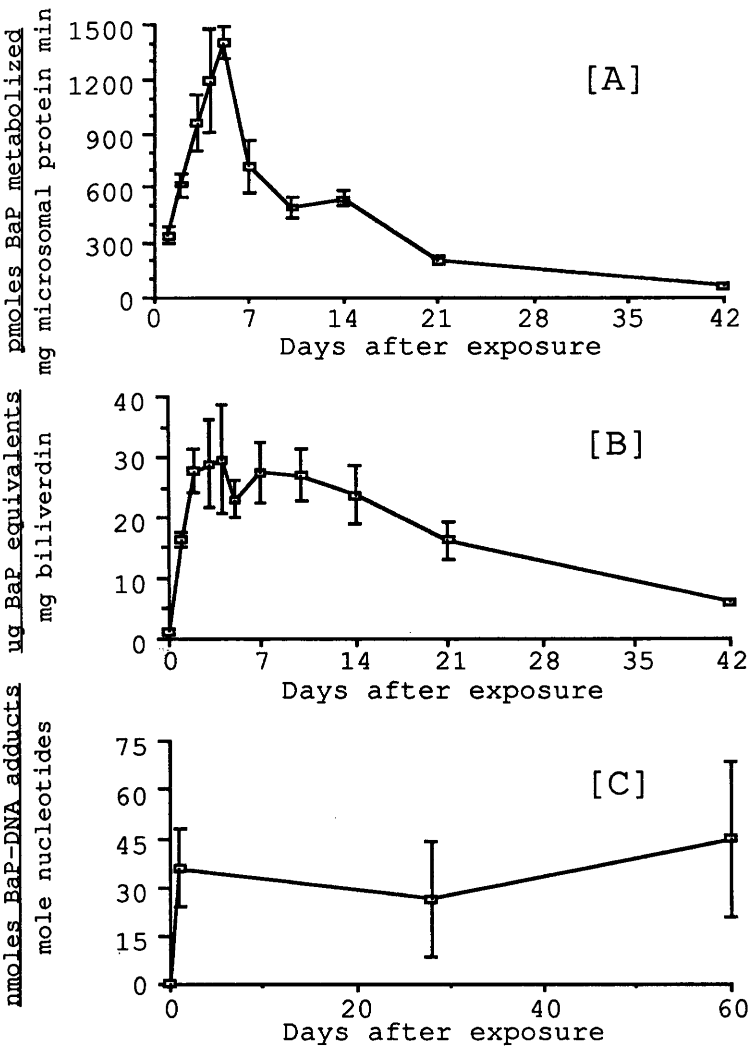


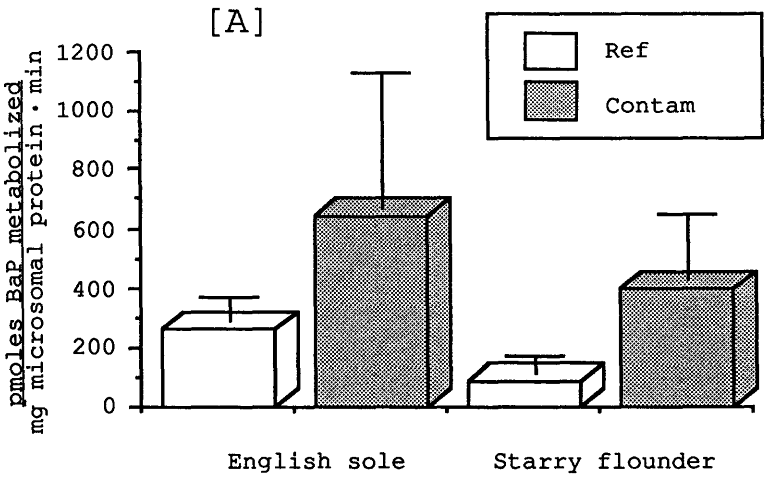
Figure 2. Time-response curves for [A] hepatic aryl hydrocarbon hydroxylase activity, [B] levels of fluorescent aromatic compounds in bile, and [C] levels of hepatic DNA adducted with a metabolite of BaP, in juvenile English sole exposed via intramuscular injection to either an organic-solvent extract of sediment from the Duwamish Waterway ([A] and [B], 50 g sediment extracted/kg body wt) or BaP ([C], 2 mg BaP/kg body wt. Data from Collier and Varanasi, 1987 and Varanasi et al., 1988.

but also that both of these indicators readily decrease within several days or weeks after exposure. Thus, if and when any remedial actions are undertaken in Puget Sound, measurement of these indicators both before and after a site is treated or cleaned up would potentially be useful in assessing the efficacy of the remedial action. The data for DNA modification show that, in contrast to the other two indicators described in this paper, ^{32}P-postlabeling of DNA may be a longer-term indicator of contaminant exposure. Taken together with the good dose response seen for this indicator, these data suggest that measurements of DNA modification may be a good dosimeter of cumulative xenobiotic exposure in bottomfish. Such information would be extremely useful in field studies of the biological effects of contamination of the marine environment.

Studies in Puget Sound

The Environmental Conservation Division of the Northwest and Alaska Fisheries Center is currently conducting a study aimed at developing and evaluating a suite of bioindicators of contaminant exposure in marine fish. This investigation, supported by the Office of Oceanography and Marine Assessment of NOAA, entails additional time- and dose-response studies of DNA modification resulting from contaminant exposure, together with a year-long field assessment of the bioindicators described in this paper. In addition, programs for quality assurance (QA) of data are being developed as part of the sample analyses. Such QA programs should include the use of standard analytical protocols, assessment of the reproducibility of the analytical methods, and the use of reference materials.

The results from an earlier study (Collier, 1988) of hepatic AHH activity and levels of FACs in bile of English sole and starry flounder have shown that, as expected, both of these indicators of contaminant exposure are higher in fish from a contaminated area compared to fish from a reference area. However, species differences were also seen in each of these measures, as indicated in Figure 3. It is also known that these indicators can be affected by factors other than contaminant exposure, such as spawning and feeding status (Collier *et al.*, 1986; Collier and Varanasi, 1987; Johnson *et al.*, 1988). Thus, field investigations of these parameters need to include sufficient data to allow an assessment of these factors. In our current study, we have been sampling, on a bimonthly basis, flatfish from several sites in Puget Sound. The sites, shown in Figure 4, were chosen to include a wide range of contaminant levels, and include Polnell Point, Useless Bay, Pilot



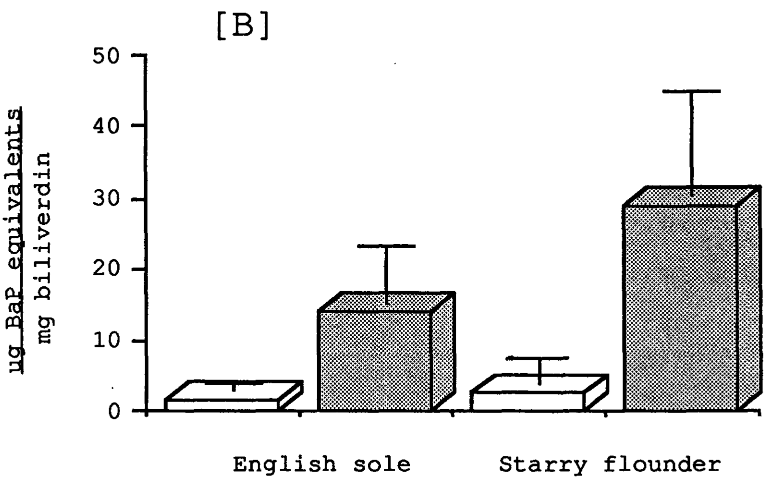


Figure 3. Hepatic aryl hydrocarbon hydroxylase (AHH) activities [A] and levels of fluorescent aromatic compounds (FACs) in bile [B] of English sole and starry flounder captured from a reference site [Saratoga Passage; (Ref)] and a contaminated site [Duwamish Waterway; (Contam)] in Puget Sound. Statistical analyses showed significant effects due to site of capture ($p<0.001$ for both AHH and FACs) and species ($p<0.01$ for AHH and $p=0.06$ for FACs). Data from Collier, 1988.

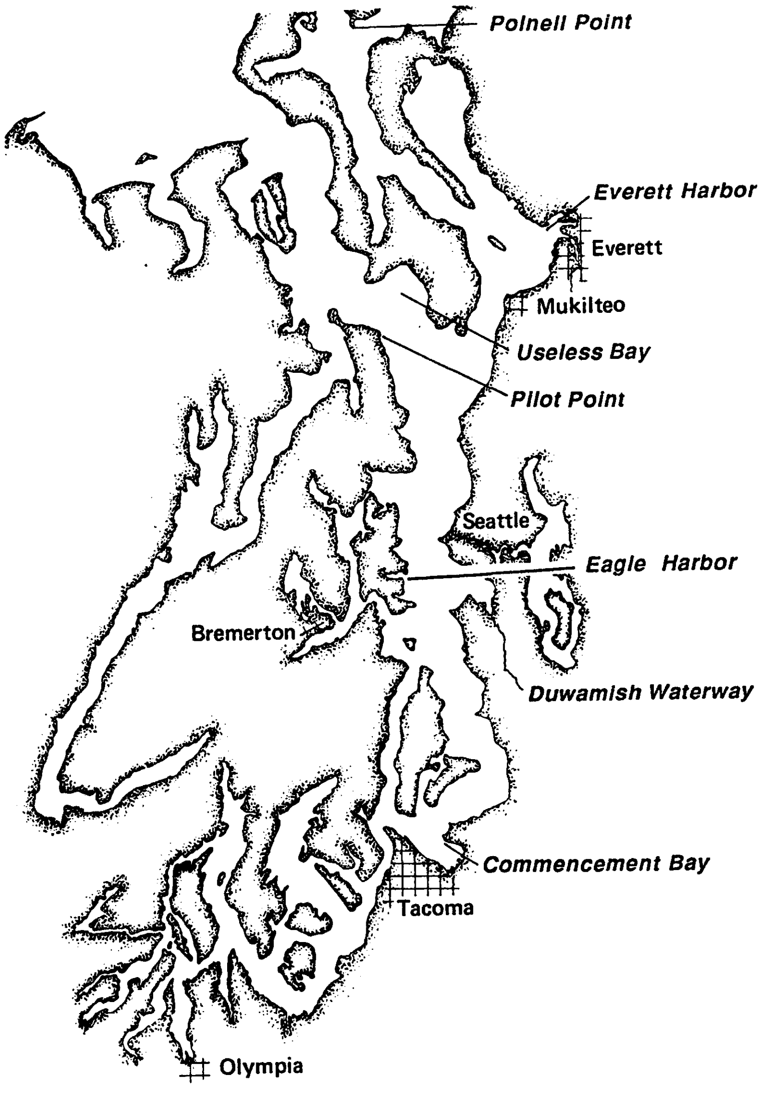


Figure 4. Map of Puget Sound, showing sites currently being sampled in field studies evaluating potential bioindicators of contaminant exposure.

Point, Everett Harbor, the Duwamish Waterway, Eagle Harbor, and Commencement Bay. English sole is the primary species being studied, but rock sole and starry flounder are also being sampled and analyzed as available. The data from these studies should allow us to determine which combinations of bioindicators provide the best assessment of contaminant exposure in bottomfish, together with providing information about possible variations in these measurements. The inclusion of additional species will also allow us to determine if any of these indicators are species-specific. Moreover, the information obtained in our studies of the contaminated areas should provide a very valuable data base in the event that remedial actions are undertaken to clean up any of these sites.

Acknowledgments

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Relating Field and Laboratory Studies: Cause-and-Effect Research

Michael H. Schiewe, John T. Landahl, Mark S. Myers, Paul D. Plesha, Frank J. Jacques, John E. Stein, Bruce B. McCain, Douglas D. Weber, Sin-Lam Chan, and Usha Varanasi*

Field surveys have played, and will no doubt continue to play, a critical role in assessing the health of Puget Sound. One major component of such field surveys is the measurement of concentrations of the myriad sediment-associated chemical contaminants often found in the Sound's urban embayments. Another major component of most field surveys focuses on direct and indirect measurement of biological effects. Among the more commonly used measures of effects have been determination of the prevalences of a variety of liver lesions in bottom-dwelling marine fishes (*e.g.* English sole), and assessment of sediment toxicity in bioassays using selected life stages of a variety of invertebrate species .

The theme of the present scientific session is the question of how good are biological effects measures as indicators of pollution-associated problems. One of the critical factors in addressing this question is knowledge of how strong is the evidence linking the effect (*e.g.* prevalence of neoplasms, significant toxicity in a sediment bioassay) to its cause(s). Unfortunately, available data on cause-and-effect relationships between many biological effects and environmental causes are, at best, relatively limited. The bright side is, however, that this situation is gradually changing. In our laboratories the findings from several recently completed studies are yielding data on cause-and-effect relationships that should be of considerable value to environmental managers in interpreting the significance of the findings of biomonitoring programs.

The use of histopathology to monitor prevalences of liver diseases in bottom-dwelling species has been used in field surveys in Puget Sound for over a decade (see Myers *et al.*, this volume). Indeed, it was the growing body of evidence

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from these studies that first suggested the relationship between liver tumors and exposure to sediment-associated contaminants (Malins *et al.*, 1984). Placing this relationship on an even more solid basis has been recent work in our laboratories using the methods of analytical epidemiology to further refine and strengthen the argument for this relationship. Using data from six field surveys conducted by our research group between 1979 and 1985, a model relating prevalences of hepatic neoplasms in English sole to fish size and sediment concentrations of selected classes of xenobiotics has been constructed. The proposed model (Figure 1), which is based on 66 collections of fish at 46 sampling sites in Puget Sound, accounts for approximately 35% of the observed variation in prevalences of hepatic neoplasms in English sole. Fish size exerted the greatest positive effect (15.3%), with observed neoplasm prevalences being greater when only large fish were selected than when the entire size range was employed. Moreover, neoplasm prevalences were positively correlated with sediment concentrations of both aromatic hydrocarbons (AHs, 12.1%) and with polychlorinated biphenyls (PCBs, 1.0%), and negatively correlated with sediment concentrations of chlorinated butadienes (2.4%). Interestingly, neoplasm prevalences were also negatively correlated with the interaction of AHs and PCBs (4.5%), suggesting a modulating effect of one or both of these classes of xenobiotics on the carcinogenicity of these compounds.

Notwithstanding the value of this model in contributing to a greater understanding of the etiology of contaminant related liver neoplasia in English sole, it is not *de facto* evidence for a cause-and-effect relationship. We are, however, addressing this issue in a series of controlled laboratory studies in which the tumorigenicity of organic-solvent extracts of urban sediments are being tested in this species. The results of this study, which will be reported in detail elsewhere (Schiewe *et al.*, manuscript in preparation), revealed a significantly higher ($p \leq 0.05$) incidence of a spectrum of lesions, including nuclear pleomorphism, megalocytic hepatosis, and preneoplastic foci of cellular alteration, in English sole exposed to a fraction of an extract of Eagle Harbor sediment enriched for high molecular weight AHs and nitrogen-containing aromatic compounds (Figure 2). English sole exposed to the model hepatocarcinogen benzo[a]pyrene (BaP) also exhibited elevated prevalences of these same lesions. Control groups of English sole, including the fish exposed to a similarly-prepared extract of West Beach sediment (a relatively uncontaminated reference site in northern Puget Sound), fish exposed to the carrier only, or fish that were not exposed, did not develop these same lesions. Moreover, lesions produced in the Eagle Harbor sediment extract-exposed sole were indistinguishable from those routinely

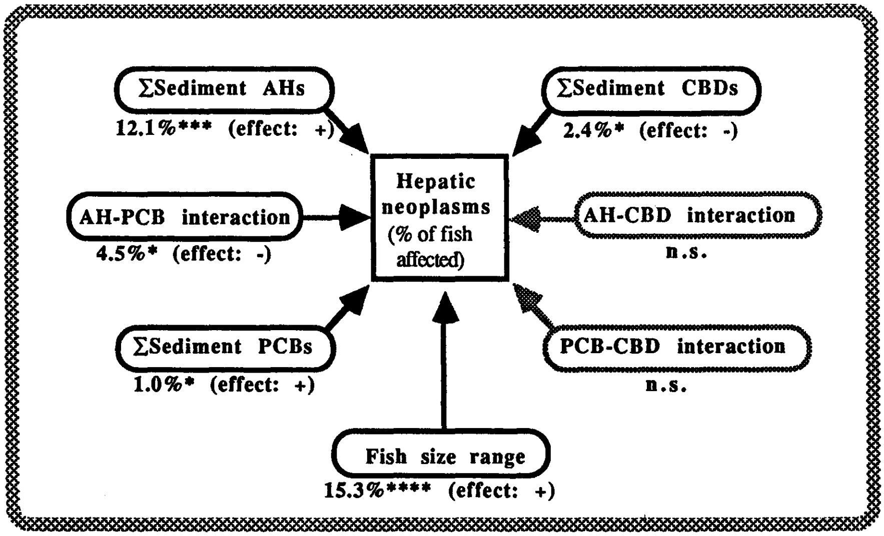

FIGURE 1. Model summarizing logistic regression analyses relating sediment concentrations of aromatic hydrocarbons (AHs), polychlorinated biphenyls (PCBs) and chlorinated butadienes (CBDs) to prevalences of liver neoplasms in English sole from Puget Sound. The model (AHs + PCBs + fish size - CBDs- [AH + PCB interaction] accounts for 35.3% of observed variation in neoplasm prevalence (% = percentage variation in neoplasm prevalence explained; n.s. = effect not significant [p > 0.05]; * = effect significant [p ≤ 0.01]; ** = effect significant [p ≤ 0.001]; *** = effect significant [p ≤ 0.00001]. An earlier version of this model has been previously published (Malins et al., 1988).

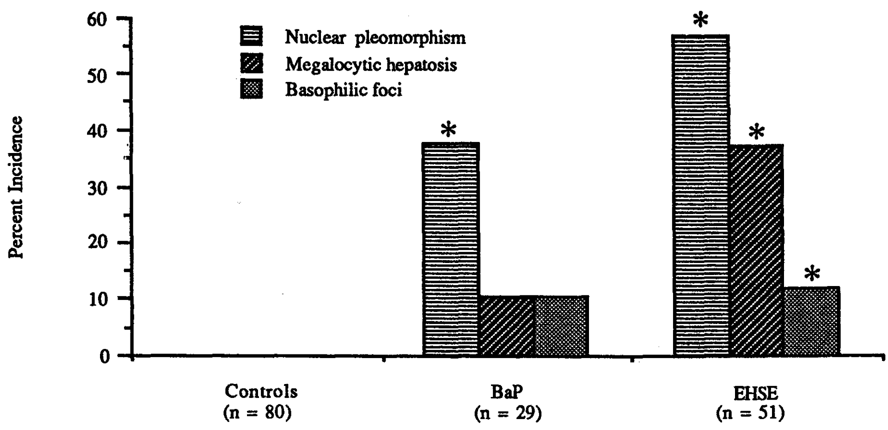

FIGURE 2. Incidence (%) of selected hepatic lesions in English sole (Parophrys vetulus) parenterally exposed to Eagle Harbor sediment extract (EHSE) or benzo[a]pyrene (BaP) for one year. (* indicates an incidence significantly different from control [P ≤ 0.05]).

observed in English sole captured in this creosote-contaminated embayment (Malins *et al.*, 1985; Myers *et al.*, 1987). Hence, these data directly implicate the sediment contaminants as the cause of the lesions and speak to the scientific soundness of using histopathology as a monitoring tool for adverse effects of chemical contaminants.

Bioassays have a long history of use in aquatic toxicology and in the past decade their use to assess sediment toxicity in field surveys in Puget Sound has become commonplace. Sediment bioassays also are used extensively in the evaluation of dredged sediments to evaluate disposal options. One of the more commonly employed sediment bioassays in the Puget Sound basin has been one measuring 10-day survival of the infaunal amphipod, *Rhepoxynius abronius*. Despite its widespread use, little was known until recently about the bioavailability of most of the more commonly encountered chemicals in sediments to, and their potential effects on, this species. In addressing the need for information about this species, we have been conducting a series of studies with *R. abronius* and sediments supplemented with defined mixtures of chlorinated and aromatic hydrocarbons. (Plesha *et al.*, Mar. Environ. Res., in press). These studies were designed to build on results of previous studies in our laboratories that demonstrated the bioavailabilty of a variety of 3 to 5 ring AHs from contaminated sediment to *R. abronius* as well as the capability of this species to metabolize and excrete these compounds (Varanasi, *et al.*, 1985).

For the amphipod investigations, a reference sediment from Hood Canal was supplemented with mixtures of 7 AHs or 4 chlorinated hydrocarbons (CHs). The compounds selected were all considered contaminants of concern based on their toxicity to selected marine species and their presence and persistence in urban areas of Puget Sound (Konasewich *et al.*, 1982). The nominal concentrations of the individual components were selected to approximate the highest concentration of each compound measured in sediment from either the Hylebos or Duwamish Waterways of Puget Sound (Malins *et al.*, 1982).

Results of the 10-day bioassays of the AH- and CH- amended sediments are summarized in Table 1. Significantly higher mortality, compared to controls, occurred among amphipods exposed to the CH-amended sediments at both the 5X level and at the level approximating that measured in contaminated sediments of Puget Sound. In contrast, significantly higher toxicity was observed with the AH-amended sediment only at the 5X level. Also assessed as part of these experiments was the degree of exposure of the amphipods to AHs and CHs based on uptake of representative radiolabeled compounds (*i.e.* naphthalene, BaP, PCBs).

TABLE 1. Survival of amphipods (*Rhepoxynius abronius*) following a 10-day exposure to sediments supplemented with mixtures of aromatic or chlorinated hydrocarbons. (Data from Plesha *et al.*, Mar. Environ. Res., in press).

Sediment	Percent Survival ($\overline{X} \pm$ s.d.)
Control (Untreated)	94.0 ± 5.5
Solvent Control (Acetone treated)	94.0 ± 4.2
1x Aromatic Hydrocarbons	95.0 ± 6.2
5x Aromatic Hydrocarbons	83.0[a] ± 2.8
1x Chlorinated Hydrocarbons	69.0[a] ± 7.4
5x Chlorinated Hydrocarbons	15.0[a] ± 13.7

[a] Significantly different from control (P = 0.05).

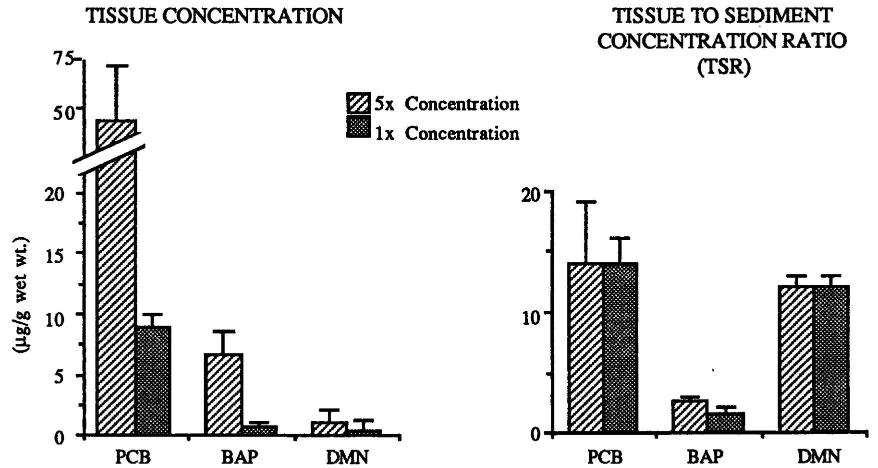

FIGURE 3 Concentrations of radiolabeled polychlorinated biphenyls (PCBs), benzo[a]pyrene (BaP)and 2,6-dimethylnapthalene (DMN) in the amphipod, Rhepoxynius abronius, and tissue to sediment ratios (TSRs) following a 10-day exposure to aromatic and chlorinated hydro-carbon-amended sediments. (Data from Plesha et al., Mar. Environ. Res., in press).

Measurement of radioactivity incorporated by *R. abronius* after the 10-day exposure period indicated a dose-related uptake of xenobiotic compounds from both the CH- and AH-supplemented sediments (Figure 3). While it is clearly a step forward to demonstrate a dose-related response to sediment concentrations of xenobiotics, it is an even greater step to relate sediment concentrations of xenobiotics to uptake and uptake, in turn, to effect. In these studies the latter was indeed the case. Although these data on uptake and toxicity of sediment-sorbed contaminants support the use *R. abronius* to assess sediment quality, there are several additional questions regarding, for example, influence of physical factors (*e.g.* grain size distribution; sediment redox potential) that are yet to be addressed. Clearly, a thorough understanding of all factors -- physical, chemical and biological -- that effect survival of a bioassay test species, is required for intelligent use of a bioassay in biomonitoring.

Although the list of sediment bioassays available to the environmental manager is gradually expanding, the majority of such bioassays focus on acute lethal effects, and to a lesser extent on acute sublethal effects. There are few sediment bioassays that focus on the potential of sediment to produce long-term or chronic effect. Hence there is a critical need to concentrate greater research effort in this area. In this regard, recent work in our laboratories include investigations exploring the feasibility of using juvenile sand dollars (*Dendraster excentricus*) and an aquarium fish, the Japanese medaka (*Oryzias latipes*) in chronic effects bioassays. This emphasis on chronic effects does not, however, obfuscate the need to continue examining cause-and-effect relationships with routinely used bioassay test species. For example, just as we have investigated the uptake and toxicity of selected sediment-sorbed contaminants in *R. abronius*, there is a need to conduct similar investigations with larvae of the Pacific oyster (*Crassostrea gigas*), a species and life-stage widely employed in sediment bioassays.

In summary, we have presented an overview of the results of two recently completed laboratory studies that provide insight into the utility of two of the more commonly used bioindicators in Puget Sound field surveys. In the case of histopathological monitoring of fish lesion prevalence, laboratory studies support a cause-and-effect relationship between selected liver lesions, including presumptive preneoplastic lesions, and exposure to sediment-associated organic contaminants, particularly aromatic compounds. In amphipod bioassays with *R. abronius*, these data demonstrate dose related uptake and toxicity of sediment-sorbed aromatic and chlorinated hydrocarbons. Such insight into cause-and-effect relationships clearly enhances the ability to more fully understand and interpret the significance of findings from intensive field surveys.

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Summary Statement

Edward R. Long*

All of the research reported by the speakers in this session provided important advances in the development of measures of biologic effects. These measures are important tools for assessing the consequences, if any, of the accumulation of contaminants in water, sediments, and biota.

Kocan and Landolt described a method now available for testing toxicity to herring embryos either in the field or in the laboratory. The laboratory tests can be performed with water samples or aqueous extracts of sediments. Tests performed with samples from Port Gamble and in the field in Port Gamble showed differences in incidences of physical defects (abnormalities) in the embryos following exposure to the sample sites. Severely deformed fish embryos in nature probably do not survive as juvenile fish, therefore, this test method may be a very sensitive measure of effects that are chronic and slow developing, yet ultimately fatal.

Anderson described a new proposed approach to interpretation and use of water bioassay data for single chemicals. The new approach couples data from laboratory tests of toxicity and data from field observations of survival in natural biological populations. The two types of data are coupled by a statistical model. An example of the application of the method to juvenile salmon was provided.

Weston described research currently underway to distinguish effects upon benthic communities attributable to organic enrichment of sediments from those attributable to accumulation of toxic chemicals in the sediments. A field experiment at sites in Clam Bay near Manchester and in Eagle Harbor on Bainbridge Island is underway to evaluate the effects of these two variables. Preliminary data for polychaetes are available and suggest differences between the two sites in the patterns of abundance with increasing distance from the pollution sources and with increasing depth in the sediments.

Collier et al. assessed the response of three biochemical measures of the exposure of fish to organic compounds in laboratory tests. The three measures of exposure involve quantification of the production of detoxifying enzymes by the liver, the concentrations of metabolites of organic compounds in the bile and the occurrence of modified DNA. All three measures showed responses related to increasing doses of a contaminant. All three tests could be used to complement or in place of traditional chemical analyses of tissues.

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Schiewe et al. reviewed research underway in their laboratory to link chemical concentrations to measures of effects in English sole and a sediment bioassay organism. The incidence of liver neoplasms was documented to be highly related to size of the fish and the concentration of polynuclear aromatic hydrocarbons in the sediments. Mortality in amphipods exposed to clean sediments increased significantly after chlorinated hydrocarbons or aromatic hydrocarbons were added to the sediments.

The answer to the question posed to the session speakers: "Do we have good indicators of problems?" is "YES." We have many good indicators. Many biological tests serve as excellent indicators of environmental "problems" as a result of exposure to contaminants. Since routine chemical analyses often do not measure many chemicals that are potentially toxic, biological tests become very important in environmental assessments. Test organisms will respond to many toxicants, whether or not they are quantified by the analytical chemists. However, the good indicators that we currently have available could be better. They could be more sensitive, less expensive, more easily attributable to specific contaminants, and more easily interpreted.

TOXIC CHEMICALS AND THEIR EFFECTS

WHAT ARE WE FINDING?

Bruce B. McCain,
National Oceanic and Atmospheric Administration
Marsha Landolt,
University of Washington
Session Co-Chairs

Introductory Statement

Bruce B. McCain * and Marsha L. Landolt**

Over the past twenty years an increasing body of evidence has developed to suggest that marine animals residing in polluted coastal areas are adversely affected by the presence of toxic chemicals in their environment. Examples of these effects include reproductive failures among marine mammal populations in the Baltic Sea, declines in marine bird populations in southern California, and epizootic neoplasms among bottom fish in several urban estuaries.

Studies in Puget Sound have clearly demonstrated the presence of high concentrations of a variety of toxic chemicals. These contaminants tend to be concentrated adjacent to urban areas, such as Commencement and Elliott Bays; however, high levels have also been found in some non-urban sites, such as Eagle Harbor, that are associated with industrial activity. Over the past decade, numerous field studies have also documented the fact that high levels of PCBs and other organic compounds are present in the tissues of resident invertebrates, fish and marine mammals. In addition, studies have demonstrated high prevalences of pathological conditions in some species of fish.

The papers in this session will present new data regarding contaminant levels in Puget Sound biota and they will expand our understanding of the resultant biological consequences. These reports will describe the results of investigations on marine mammals, birds and fish, and one study will address potential human health risks associated with consumption of freshwater organisms.

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Status of Puget Sound Harbor Seals: Trends in Population Size and Contaminant Concentrations

John Calambokidis*, Gretchen H. Steiger*, James C. Cubbage*, Suzanne Kort*, Sheryl Belcher*, and Maureen Meehan*

Introduction

The harbor seal (Phoca vitulina) is the most numerous marine mammal inhabiting Puget Sound and is the only pinniped that resides year-round and breeds in Washington State. Harbor seals were hunted until recent years and the State of Washington maintained a bounty on seals through the early 1960s because of purported conflicts with commercially valuable fish. Since 1972 marine mammals have been protected from killing and harassment under the federal Marine Mammal Protection Act (MMPA) of 1972. Though killing by fishermen with permits is still allowed, the MMPA has dramatically reduced the mortality of most pinnipeds.

High levels of stable chlorinated hydrocarbons have been recovered from from the tissues of marine mammals from different parts of the world, including Puget Sound (Calambokidis et al. 1984, Risebrough 1978). These contaminants have been linked to reproductive problems in several pinniped populations (Reijnders 1980, 1981, 1982, DeLong et al. 1973, Gilmartin et al. 1976, Helle et al. 1976a, 1976b, Helle 1980) and reproductive failure caused by ingestion of fish from contaminated areas has been demonstrated in a controlled experiment with harbor seals (Reijnders 1986).

The biology, abundance, and presence of toxic contaminants of Puget Sound harbor seals have been studied in recent years (Calambokidis et al. 1978, 1979a, 1984, 1985, Newby 1973a, 1973b, Johnson and Jeffries 1977, 1983, Arndt 1973). The purpose of this paper is to examine trends in Puget Sound harbor seal populations and contaminant levels based on results available from past research as well as new unpublished information.

Methods

Harbor seals have been censused by Cascadia Research from 1976 to present with the greatest amount of effort made in 1977 and 1984. Replicate aerial and land-based counts at the primary harbor seal haul-out sites in Puget Sound were conducted in these two years. In 1977, 615 land-based counts were made of seal haul-out sites and 19 aerial surveys flown. In 1984, 483 counts were made and 21 aerial

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surveys flown. Effort in other years was substantially less with counts made of only some of the haul-out sites. Pup production and mortality were also examined.

Tissues for contaminant analysis were collected during necropsies of stranded marine mammals. Blubber samples from 98 harbor seals were examined for concentrations of the industrial contaminant PCBs (polychlorinated biphenyls) and DDE (the environmental breakdown product of the pesticide DDT). Tissues from an additional 17 harbor seals were examined for a broader range of organic and inorganic chemicals by several outside laboratories. Most of the samples analyzed were recovered from harbor seals stranded in 1976-77 and 1984. A quarter of the samples (27) were from harbor seals that were collected in Grays Harbor and provided by Steve Jeffries and Dr. Murray Johnson.

Most of the samples were analyzed at the Environmental Analysis Laboratory of the Evergreen State College by the authors using methods described previously (Calambokidis et al. 1979b, 1984). In summary, blubber samples were digested in acid (Stanley and LeFavoure 1965), extracted with hexane, cleaned with concentrated sulfuric acid (Murphy 1972), and injected on Hewlett-Packard electron-capture gas chromatograph equipped with a 6' glass column. PCBs and DDE were quantified based on comparison to standards injected daily. PCBs were quantified by individual homologs based on percent weight figures (Webb and McCall 1973).

Results for different sites were grouped into three regions: southern Puget Sound (south of the Tacoma Narrows), Hood Canal, and north of Puget Sound (Strait of Juan de Fuca and San Juan Islands).

Results and Discussion

Harbor seal numbers at sites throughout Puget Sound have increased significantly from the middle to late 1970s to the mid-1980s. Changes in numbers of seals were measured using three values: 1) the highest number of seals seen at any one time during the year, 2) the average of the daily high counts of seals, and 3) the number of pups born at the site. At all sites examined, all three of these indicators were higher in 1984 than in 1977-79. Average counts were significantly higher in 1984 compared to 1978-79 at 9 of the 10 areas (t-test, $p<0.001$ in 7 cases and $p<0.05$ in 2 cases).

Figure 1 shows the annual percent increases in harbor seal numbers calculated from comparison of counts made in 1977-78 to 1984. A single regional 'growth index' was calculated by averaging the rate of increase indicated by the three measures of seal numbers for all the different sites in the region (weighted by seal numbers at each site). Southern Puget Sound showed the highest growth rate with a 'growth index' of 19% per year. This primarily reflects the 17% annual growth rate at Gertrude Island, the site with the highest numbers of seals in the region. The 'growth index' for sites north of Puget Sound was slightly lower, 14% increase per year, primarily reflecting the many adjacent haul-out areas in the San Juan Islands which were treated as one site. Increases in numbers of harbor

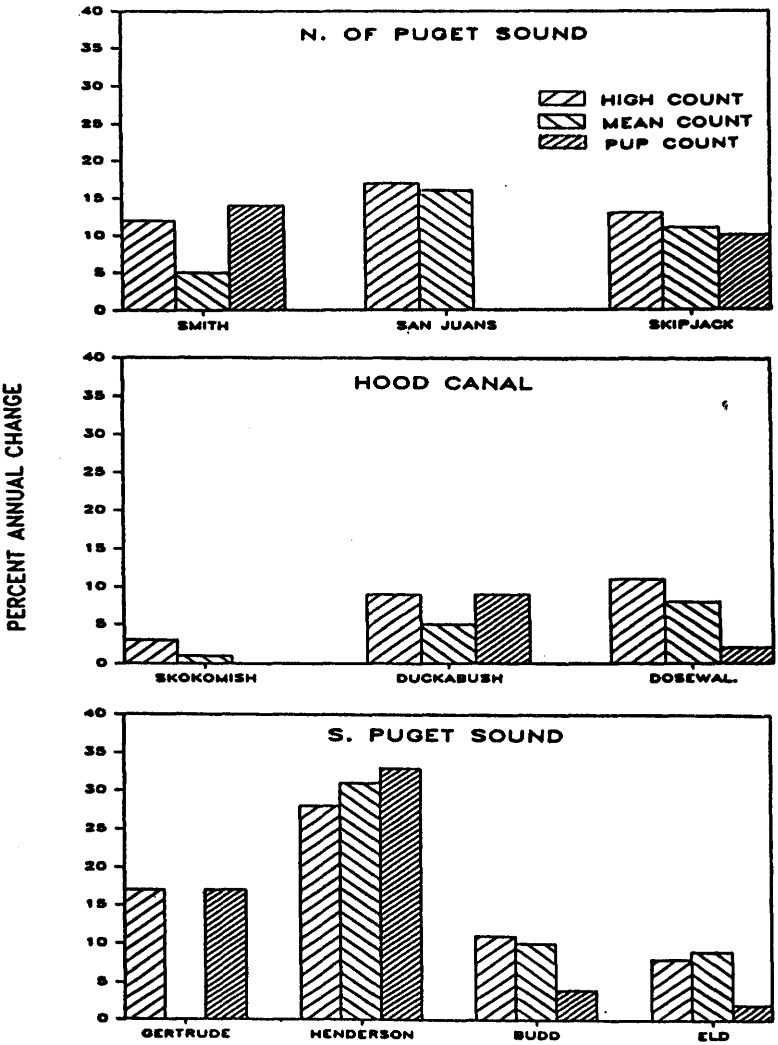

Figure 1. Percent annual increases in numbers of harbor seals at different haul-out sites based on comparison between counts made in 1977-79 and 1984. Data from Calambokidis et al. (1978, 1985) except San Juan Islands for 1978 (Everitt et al. 1979) and Gertrude Island for 1979 (Skidmore and Babson 1981).

seals in the Hood Canal has occurred at a much slower rate than the other areas. The 'growth index' was 5% per year for the region. Seal numbers at Skokomish River Delta, the site with the highest numbers of seals in the Hood Canal, were almost unchanged between 1977 and 1984. Beach et al. (1985) reported an annual increase of 19% in numbers of harbor seal pups and 10% for non-pups on the outer coast of Washington from 1976 to 1982.

Figure 2 summarizes the changes in the numbers of seals that have occurred at Gertrude Island, a site that has been monitored by a number of researchers since 1965. It is clear that increases in seal numbers at this site only began in the late 1970s. Unfortunately, data from other sites during 1960s or early 1970s are not complete. Newby (1971, 1973a) reported a population estimate for seals in Washington State, but with the exception of Gertrude Island, these are based on very limited data. These data and anecdotal reports from people living near seal haul-out sites, however, suggest seals numbers at most sites began increasing in the 1960s or early 1970s. This timing is consistent with the timing of the end to the bounty on harbor seals in the early 1960s and federal protection in 1972.

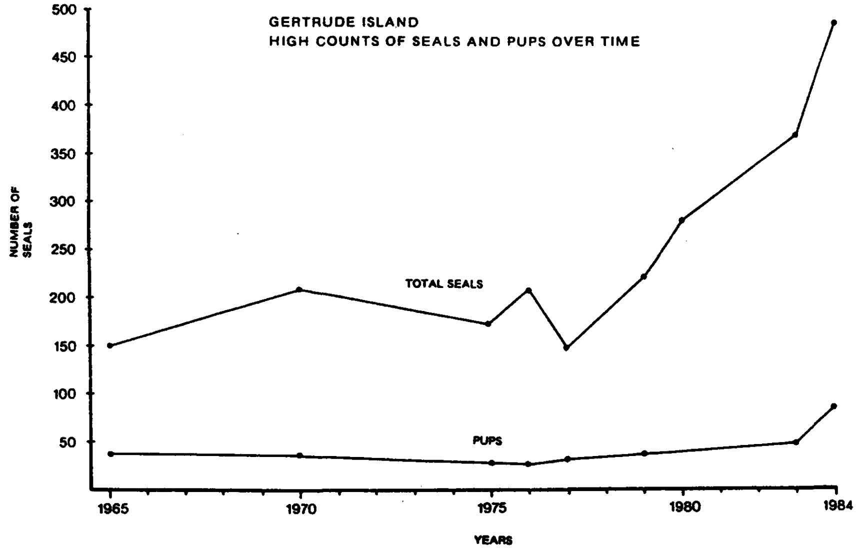

Figure 2. High counts of harbor seals and harbor seal pups at Gertrude Island, southern Puget Sound. Data are summarized from Calambokidis et al. (1978, 1979, 1985), Arnold (1968), Newby (1971), Johnson and Jeffries (1983), Skidmore and Babson (1981), and Gearin and DeLong (1984).

Previous reports have examined concentrations of PCBs and DDE in Puget Sound harbor seal tissues collected in 1972 (Arndt 1973) and 1975 to 1982 (Calambokidis et al. 1984). New data is also available for 1984 to 1985 (authors, unpublished data). Contaminant concentrations in Washington State harbor seals vary by region, age class, and in adults, by sex (Calambokidis et al. 1984). To determine temporal trends these other factors need to be isolated or controlled for. One way to do this is to limit comparisons to a single age class from one region. The blubber of 23 harbor seal pups from southern Puget Sound collected from 1972 to 1985 have been tested to date (Figure 3). A significant correlation (n=23, p<0.01) was found between both PCB and DDE concentrations and collection year. PCB concentrations in blubber (wet weight) have declined from an average of 174 ppm (n=4) in 1972 to 23 ppm (n=9) in 1984-85.

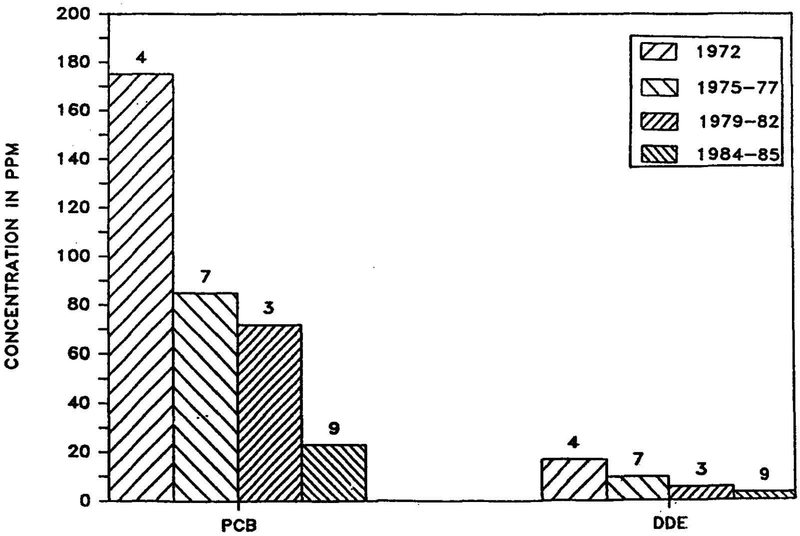

Figure 3. Concentrations (parts per million, wet weight) of PCBs and DDE in harbor seal pups from southern Puget Sound by year of collection. Data for 1972 is from Arndt (1972) and data for 1975-1982 is from Calambokidis et al. (1984).

Small sample sizes (10 or less) limited the value of statistical tests of changes in PCB and DDE concentrations in harbor seals from other areas or in other age classes. In two other cases (adult males from southern Puget Sound and pups from north of Puget Sound) multiple samples were available for different time periods. In both cases concentrations of both PCBs and DDE were lowest in samples from recent years (1984-86) compared to previous years. Concentrations of PCBs and DDE in adult male harbor seals from southern Puget Sound were highest in 1976-79 compared with either 1972 or 1983-85. This difference, as compared to the steady decline of contaminant levels in pups from this area, could reflect the accumulation of contaminants in males over their lifetime, delaying when peak levels would be seen in adult males compared to pups.

Limited data exist on concentrations of other contaminants in Puget Sound harbor seals (Calambokidis et al. 1984). They indicate a wide variety of other organic and inorganic contaminants present in seal tissues, however, sample size does not allow evaluation of trends or significance of these contaminants.

Contaminants do not appear to be currently affecting harbor seal numbers in Puget Sound based on increases in most areas. Reproductive problems in southern Puget Sound harbor seals, including the premature births and birth defects reported in the early 1970s (Newby 1971, 1973b), are not now occurring in this region (Calambokidis et al. 1985). Other sublethal contaminant impacts of contaminants on seals, however, cannot be ruled out. Circumstantial evidence indicates PCBs probably had an impact on Puget Sound harbor seals in the 1970s. During this period: 1) concentrations of PCBs in southern Puget Sound harbor seals were similar to those reported to be impairing reproduction in seals in other areas, 2) reproductive problems were reported at the largest seal rookery in Puget Sound, and 3) population trends indicate seal numbers did not begin increasing in southern Puget Sound until the late 1970s when PCB concentrations were declining.

Information on harbor seal populations and contaminant concentrations can provide valuable insights into the health of Puget Sound and changes over time. Problems harbor seals were experiencing in the early 1970s in Puget Sound may have been one the first indications of the accumulation of contaminants in the Sound. There has been relatively little funding for marine mammal research in Puget Sound. Many aspects of the biology and contaminant loads of harbor seals remain poorly undestood. Information on other marine mammal species in Puget Sound is even more lacking with often no data on population sizes. Given their position on the top of the food chain and the increasingly important role they play in the Puget Sound ecosystem, this research is needed.

Acknowledgments

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Puget Sound Glaucous-winged Gulls: Biology and Contaminants

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Introduction

Eggshell thinning has been documented in a wide variety of bird species, including many waterbirds, throughout the world (Ratcliffe 1970, Risebrough et al. 1970, Anderson and Hickey 1972) and has been primarily attributed to the extensive use since 1947 of the pesticide DDT, which transforms into its primary metabolite DDE (Hickey and Anderson 1968, Ratcliffe 1970, Bluss et al. 1972a, 1972b, Cooke 1973, Nisbet 1980, Henny et al. 1984). The inland marine waters of western Washington are used by a variety of waterbirds throughout the year (Jewett et al. 1953, Wahl et al. 1981, Wahl and Speich 1984, unpubl. obser.), and include sites in urban, industrial, and agricultural areas (Dexter et al. 1981, 1985). Recent research has revealed a wide variety of contaminants, including DDT, DDE, and PCB in Puget Sound marine waters and biota (Fitzner et al. 1982, Konasewich et al. 1982, Malins et al. 1982, Calambokidis et al. 1984, Dexter et al. 1985). However, to date only limited research has been conducted on possible adverse effects of contaminants on marine birds in the Puget Sound area (Riley et al. 1983, Calambokidis et al. 1985, Fry et al. 1987). We examined eggshell thickness of Glaucous-winged Gull (*Larus glaucescens*) eggs at different sites in western Washington in 1984, and report here the degree of thinning observed when compared to observed historical thicknesses. We also report on the PCB and total DDT residues found in Glaucous-winged Gull eggs. The implications of our observations to the breeding marine birds of Puget Sound are discussed.

Methods

Eggs were collected from Glaucous-winged Gulls nesting in western Washington. Study sites (Figure 1) in the Seattle-Tacoma-Olympia urban-industrial corridor were selected for their historical or potential pollution impacts and for comparison other sites were chosen in presumably relatively unpolluted areas. Complete clutches of three eggs and incomplete clutches were collected from five Glaucous-winged Gull colonies.

Because no pre-1947 eggshell thickness value for the Glaucous-winged Gull had been published, one egg from each of 52 Glaucous-winged

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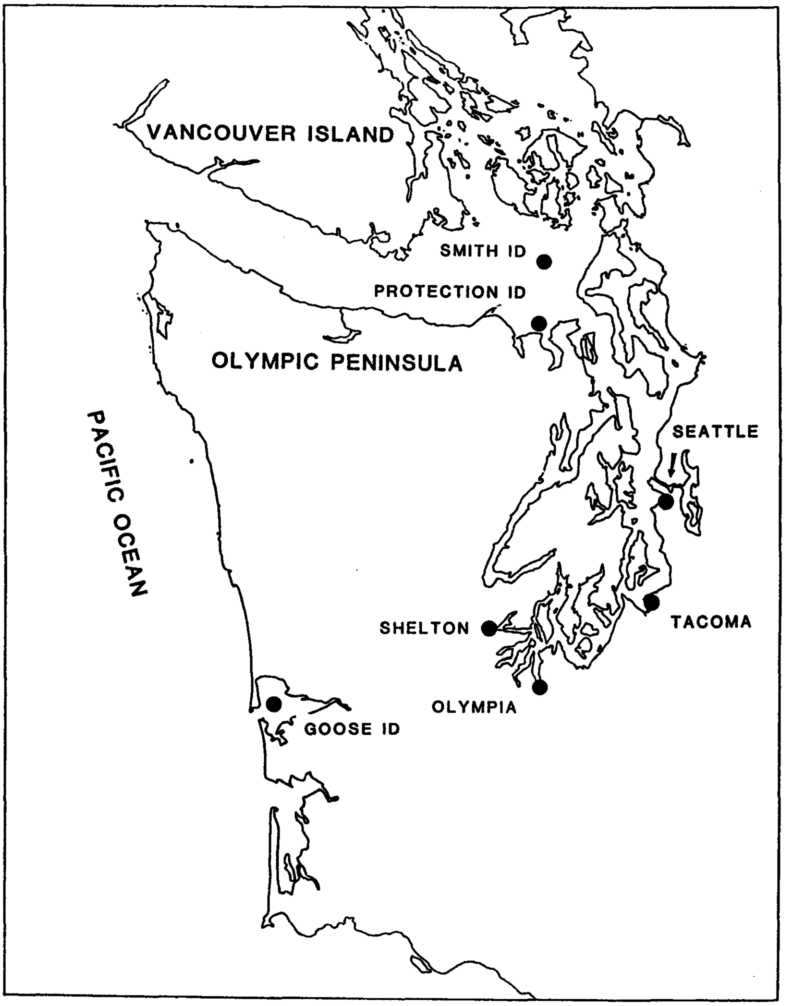

Figure 1. Locations of Glaucous-winged gull study sites in western Washington, 1984.

Gull clutches collected from colonies about the San Juan Islands prior to 1947 was measured (all eggs are in the Western Foundation of Vertebrate Zoology) to establish a pre-1947 Puget Sound Glaucous-winged Gull reference value.

All whole eggs and eggshell fragments collected were measured for shell thickness, including membrane, at or near the equator to the nearest 0.001 mm. Measurements were made by personnel of the Western Foundation of Vertebrate Zoology (WFVZ), using a modified Bench Comparator with a Federal Dial Indicator. All eggshells we collected are now deposited in either the WFVZ, Los Angeles, California, or the Burke Memorial Washington State Museum, University of Washington, Seattle. Liver weights were determined for 44 adult Glaucous-winged Gulls, mostly adult females taken from the nests sampled for eggs.

Glaucous-winged Gull eggs were wrapped in aluminum foil, placed in individual plastic bottles and refrigerated. The eggs were weighed, their length and width measured with calipers, and then opened via a small hole drilled at the eggs' equator. Empty eggs were filled with water and weighed to determine their volume. The eggs were then emptied, dried, and weighed to determine shell weight. Removed contents were examined for embryo development, homogenized and placed in glass containers previously rinsed twice with dichloromethane. Sample containers were then capped with aluminum foil previously heated to approximately 200° C, sealed with a screw cap and frozen at -60° C.

Single eggs from 17 clutches of Glaucous-winged Gulls were tested for PCBs and total DDT levels at The Evergreen State College, Olympia, Washington. Egg samples were homogenized prior to analysis. Approximately 20 g (wet weight) of eggs were used for the analysis. Analysis methods were as described in Mowrer et al. (1977) and Calambokidis et al. (1979). Samples were digested in BFM solution (perchloric and glacial acetic acid) and extracted with hexane (Stanley and LeFavoure 1965). A portion of the hexane-lipid extract was cleaned-up with concentrated sulfuric acid (Murphy 1972) and injected on a Hewlett-Packard electron-capture (63 Ni detector) gas chromatograph equipped with a 6' x 1/4" glass column packed with 10% DC-200 on gas chrom Q, 80-100 mesh. An alkaline (NaOH and KOH) precolumn converted all p,p'-DDT to p,p'-DDE and both are reported together as TDDT in text. Compounds were identified and quantified based on comparison to standards injected throughout the day. PCB peaks were quantified by individual homolog analysis using mean-weight percent figures reported by Webb and McCall (1973).

Results

There was significant variation in Glaucous-winged Gull eggshell thickness among the five sites (Table 1; ANOVA, $p<.05$). The mean thicknesses of eggs from Smith Island and Goose Island were greater than those from Seattle, Tacoma, and Shelton.

Comparison of our findings between sites and with historical values is complicated by the hybridization of Glaucous-winged Gulls with

Western Gulls (*L. occidentalis*). It has long been recognized that coastal Washington is the primary area of contact and hybridization between the two species (see Hoffman et al. 1978). The hybrid zone is also evident in Puget Sound, though birds in Puget Sound and especially in the San Juan Islands are clearly more similar to the Glaucous-winged Gull "morph", while birds nesting on Goose Island are more similar to the Western Gull "morph" (Speich, unpubl. obser.).

Table 1. Eggshell thickness (with membranes) of whole eggs of Glaucous-winged Gulls from western Washington.

Location	n	Mean thickness mm (SD)	Percent difference pre-1947
Smith Island	16	0.384 (.035)	-3
Seattle	13	0.354 (.032)	-10
Tacoma	20	0.360 (.028)	-9
Shelton	29	0.362 (.025)	-8
Goose Island	33	0.388 (.022)	-2
All sites (above)	111	0.372 (.029)	-6
Pre-1947, Puget Sound	52	0.395 (.025)	---

We compared the eggshell thickness for each site to the pre-1947 Puget Sound Glaucous-winged Gull value to determine percent eggshell thinning. The mean eggshell thickness values of gulls nesting in Seattle, Tacoma, and Shelton were significantly smaller than the pre-1947 measurements (t-tests, $p<.05$), 10%, 9%, and 8% respectively (Table 1).

The concentrations of TDDT in eggs of Glaucous-winged Gulls from three study sites (Table 2) were determined. There was a significant correlation between TDDT levels and Ratcliffe Index ($n=17$, $r=-.515$, $p<.025$, 1-tailed t-test), but not of TDDT levels and eggshell thickness ($n=14$, $r=-.449$, $p>.05$, 1-tailed t-test). Elevated TDDT levels did not appear to account for all the eggshell thinning observed. Even eggs containing less than 1 ppm (wet weight) TDDT had significantly thinner eggshells than pre-1947 values (t-test, $p<.05$). Mean eggshell thickness and Ratcliffe Index were less in eggs containing greater than 1 ppm TDDT compared to those with less than 1 ppm, but the differences were not significant (t-tests, $p>.05$ in both cases). The concentrations of PCBs in the eggs of Glaucous-winged Gulls from three sites were also determined (Table 2). There was not a significant correlation between PCB level and Ratcliffe Index ($n=17$, $r=-.274$, $p>.10$, 1-tailed t-test), nor of PCB level and eggshell thickness ($n=14$, $r=-.252$, $p>.10$, 1-tailed t-test).

Table 2. Observed concentrations (wet weight) of PCB and TDDT in Glaucous-winged Gull eggs from Puget Sound, Washington, 1984.

	PCB			TDDT		
Location	n	Mean ug/g	(SD)	n	Mean ug/g	(SD)
Tacoma	6	3.10	(1.22)	6	0.50	(0.19)
Smith Island	6	1.45	(0.70)	6	0.55	(0.11)
Seattle	5	5.04	(2.07)	5	1.29	(0.70)

Gulls in this study had marked variations in the size of their livers, which ranged from 21.3 g to 52.0 g (244% variation) in birds whose total body weight varied less than 20%. Average liver weights varied significantly among sites (ANOVA, $p<0.001$). Birds examined from Seattle had significantly larger livers than birds from any other site (t-test, $p<0.05$ in all cases), with weights ranging from 42.8 g to 52.0 g ($n=6$, 47.1 g average or 50.8 g/kg body weight). Birds from Tacoma had small livers averaging 28.8 g or 30.3 g/kg of body weight ($n=9$). The other three sites had some birds with large and some with small livers. No relationship was found between adult liver weights and PCB or TDDT concentrations in the eggs of the adult.

Discussion

The Glaucous-winged Gull is thought to be only moderately sensitive to DDE (Hickey and Anderson 1968, Peakall 1975, Nisbet 1980). The degree of eggshell thinning observed in the eggs of Puget Sound Glaucous-winged Gulls (Table 1) suggests that gulls were exposed to significant amounts of DDE. The levels of DDE observed are similar to those found in eggs of Western Gulls from coastal Oregon (Henny et al 1982), where only a moderate amount of thinning was observed, and well below the levels observed in eggs of Herring Gulls (*Larus argentatus*) from Great Lakes colonies (Gilman et al. 1977) where larger amounts of thinning were observed. Eighteen Western Gull eggs collected between 1974 and 1977 from coastal Washington colonies by Henny et al. (1982) had mean eggshell thickness 7.6% smaller than the pre-1947 Western Gull value.

In this study we did not find evidence of current reproductive failure, high incidence of eggshell breakage, eggshell flaking, or low hatching success as observed in studies of Herring Gulls in Lake Ontario colonies (Gilbertson 1974, Gilman et al. 1977). Egg breakage and flaking were associated with 9-16% eggshell thinning (Gilbertson 1974) in Lake Ontario, but were not observed when thinning had decreased to 2-8% (Gilman et al. 1977). The thicknesses of eggs of Glaucous-winged Gulls from the Seattle colony (Table 1) were generally greater than those of Herring Gulls experiencing reproductive difficulties.

Highest levels of PCBs and TDDT were found in eggs from Seattle and lesser levels in eggs from Tacoma and Smith Island. The levels in gull eggs likely reflect the distance the adults are feeding from areas of greatest contamination. Few gulls were observed feeding in the industrial waterways, while many birds were observed feeding in the open bays and passages of Puget Sound (Wahl and Speich 1984, Speich unpubl. obser.).

The Glaucous-winged Gull population of western Washington has increased since the turn of the century (Speich and Wahl ms, Eddy 1982). Recognized colonies in Seattle and Tacoma appeared about 1940. Observations of banded adults (band numbers read) indicate that many birds in these colonies were hatched on Protection Island and other northern Puget Sound colonies and they moved into southern Puget Sound colonies (unpubl. obser.). Such movement could maintain colonies even if they were experiencing reproductive failure.

A variety of environmental contaminants have been shown to alter liver weights of birds (ie. Miller et al. 1978, Peakall and Lincer 1970). A number of hypothesis may be put forward to explain the differences in liver weights we found, but without additional data the final causes of the marked variation cannot be explained.

The weak correlation between eggshell thickness or Ratcliffe Index and levels of TDDT in eggs suggests that other factors may be acting to reduce eggshell thickness. It has been shown that the presence of TDDT in gull eggs can cause feminization of male embryos and altering of female embryos in a dose dependent fashion (Fry and Toone 1981, Fry et al. 1987). Severely feminized males may be unable to reproduce. Female malformations include the persistence and enlargement of right oviducts, and abnormal development of the left oviduct which may lead to eggshell thinning and abnormal eggs when exposed birds become adults and attempt to reproduce (Fry et al. 1987, Greenwood and Blyth 1938).

The occurrence of a high proportion of supernormal clutches in 1979 in the Seattle colony, the continued presence of supernormal clutches in Seattle, and presence of right oviducts in adults from several colonies (Fry et al. 1987) suggest the past exposure of the population to TDDT. Current eggshell thinning (Table 1) and the presence of TDDT in eggs (Table 2) supports this contention. The gulls of Puget Sound likely experienced maximum exposure to TDDT several years ago. We may be observing the long term residual effects of that exposure, expressed as persistence of right oviducts and developmentally induced eggshell thinning.

Conclusions

The observations of this study and of others cited above show that PCBs and TDDT are widespread contaminants in the western Washington marine system. Expectedly, the highest levels found in eggs largely correspond to the urban-industrial centers of Puget Sound and the known contamination of sediments and resident biota. The eggshell thinning observed in Glaucous-winged Gulls also largely correspond to known contamination and use patterns.

Significant historical reproductive failure is unknown in the Glaucous-winged Gull. However, Glaucous-winged Gulls nesting in the urban-industrial areas may have experienced some reproductive failure, and many individuals upon close examination exhibit probable sub-lethal pollution-induced effects in their reproductive organs and eggs.

Acknowledgments

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POLYCHLORINATED BIPHENYL CONTAMINATION OF CRAYFISH AND FINFISH IN LAKE UNION

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Introduction and Historical Studies

Lake Union is a highly urbanized lake located in the center of Seattle, receiving relatively good quality water from Lake Washington and discharging to Puget Sound by way of the industrialized Ship Canal and Hiram Chittenden Locks. Since the City's inception, Lake Union has been a receptacle for sewage, stormwater and industrial pollutants. In 1985, during a comprehensive review of its Shoreline Master Program, the City of Seattle identified a need for an environmental management plan for the Lake Union and Ship Canal area from the Montlake Cut to the Locks. Studies conducted by Metro (Tomlinson et al., 1977) and the U.S. Environmental Protection Agency (Hileman et al., 1985) revealed that Lake Union and the Ship Canal exhibit water quality problems and contaminated sediments. Toxic pollutants such as metals, polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) were identified in Lake Union sediments at high concentrations when compared to other lakes and to interim sediment quality values proposed for Puget Sound.

The Lake Union and Ship Canal Water Quality Management Program was established in late 1985 with the City of Seattle's Office for Long-range Planning (then the Land Use and Transportation Project) in the lead role. The program is a cooperative effort involving participation by many City departments, local, state and federal environmental agencies, and interested citizens. Work performed to date has included compilation and analysis of existing data on water quality, sediment quality and biota in the study area; identification of data gaps and key issues; design and implementation of new site investigations to fill in some data gaps; and development of an Interim Action Plan. This plan represents a first step in a comprehensive effort to clean up Lake Union and the Ship Canal and focuses on source controls to reduce the amount of pollution entering

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these water bodies. Implementation and refinement of the Interim Action Plan will begin this year.

Established City goals include both maintenance of the historical working character of Lake Union and the continued public recreational use of the lake. It is the intent of the Lake Union and Ship Canal Water Quality Management Program to achieve the goals of the Federal Clean Water Act, i.e. fishable and swimmable navigable waters. One of the key program issues focuses on whether fish in Lake Union contain contaminants at levels exceeding those found in other commonly harvested fish or exceeding U.S. Food and Drug Administration (FDA) tolerances.

Concern was focused on crayfish (Pacifasticus leniasculus) because of their close contact with sediments and because they are harvested both commercially and recreationally from Lake Union. In 1984, the Seattle-King County Health Department took the lead in collecting crayfish from three sites (see Figure 1): offshore of Gas Works Park, the Montlake Cut, and the Ship Canal. Raw whole crayfish and cooked tail tissue were analyzed for metals, PCBs and PAHs (Frost et al., 1985). In 1986, as part of a pilot project designed to obtain an overview of environmental conditions in the south end of Lake Union, the City's Office for Long-range Planning took the lead in collecting crayfish from two sites (see Figure 1): offshore of the City Light Steam Plant and near the site of the City's proposed South Lake Union Park. Raw and cooked crayfish tail tissue were analyzed for metals and PCBs. These contaminants were selected because they were found at relatively high levels in south Lake Union sediments during the pilot project (City of Seattle, 1987).

The results of the 1984 study showed that raw whole crayfish had higher levels of contaminants than cooked tail tissue. None of the metals or PCBs in any of the samples were found at concentrations that would present a significant health hazard (according to FDA tolerance levels) from the consumption of crayfish. PAH contamination was found only in Gas Works Park samples at levels exceeding those found in other shellfish and finfish but not beyond the range found in many commonly consumed foods such as smoked or barbecued foods (Frost et al., 1985).

The results of the 1986 study showed that cooked crayfish tail tissue had higher levels of contaminants than raw crayfish tail tissue. Metal levels were within the range expected for crustaceans from urban waters and were not of concern to the Seattle-King County Health Department. The PCB levels in both raw and cooked crayfish tail tissue were well below the 2000 ppb FDA tolerance for PCBs in shellfish and finfish. The 1986 PCB level in cooked tail tissue of crayfish harvested near the City Light Steam Plant was 4 to 6 times as high as the levels measured in cooked tail tissue of crayfish harvested from offshore of Gas Works Park and the Ship Canal in 1984. The 1986 PCB level in cooked tail tissue of crayfish harvested near the City Light Steam Plant was also 7 times as high as the level in raw tail tissue of crayfish harvested from the same area in 1986. There is a lack of information to determine if the raw and cooked

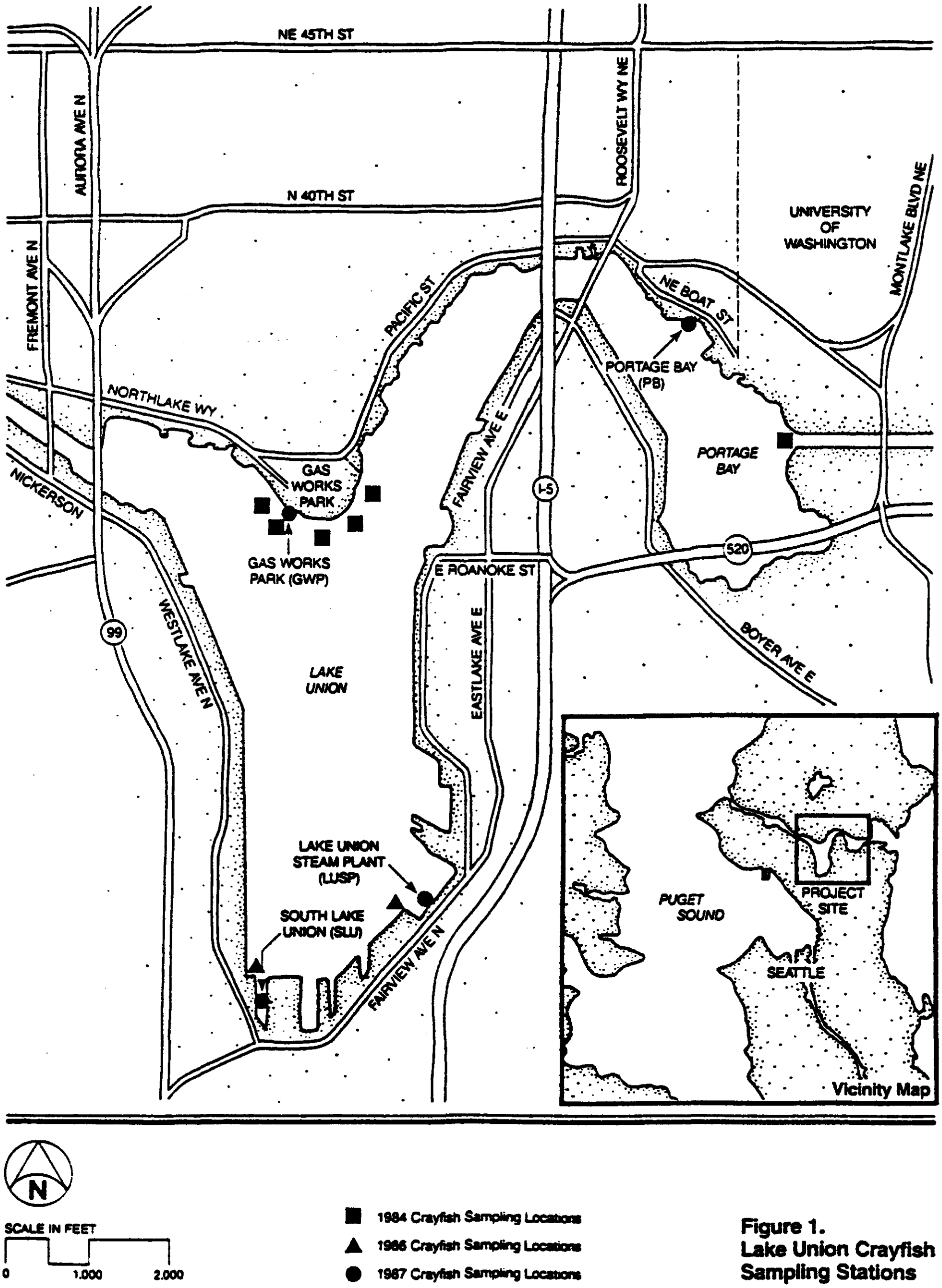

Figure 1.
Lake Union Crayfish Sampling Stations

samples were equivalent in terms of total crayfish weight and age distribution (City of Seattle, 1987).

The Seattle-King County Health Department recommended further PCB analyses including replicate samples at each location and establishment of comparability between raw and cooked tissue. Concerns were also raised about the possibility of PCB contamination arising from past activities at the Lake Union Steam Plant. Because of these concerns and because previous sampling was limited in scope, Seattle City Light and Seattle Office for Long-range Planning undertook a more comprehensive study of crayfish contamination. The study described herein focuses specifically on PCB levels in crayfish as well as in water and sediment samples and in finfish collected from Lake Union.

Lake Union: 1987 Crayfish PCB Study

Methods

This study examined the concentration of PCBs in crayfish, finfish, water, and sediments from four areas of Lake Union; (1) Lake Union Steam Plant (LUSP), (2) South Lake Union (SLU), (3) Gas Works Park (GWP), and (3) Portage Bay (PB) (Figure 1). The complete study is presented in a final report to Seattle City Light and City of Seattle Office for Long-range Planning, titled "Lake Union: 1987 Crayfish PCB Study" (Trial and Bailey, 1988).

Wire mesh crayfish traps (30 cm dia x 60 cm) were baited and set out overnight at each of the four sampling stations. At the laboratory, crayfish were equally divided among size classes for analysis of cooked and uncooked tail muscle tissue. For each station, whole crayfish were cooked (boiled) for 15-20 minutes in a Pyrex flask. Tail muscle tissue was dissected then pooled for analysis as a composite sample. Raw tail muscle tissue was also dissected from crayfish and analyzed as a composite sample for each sampling station.

To capture fish for this study, fyke nets and Oneida traps were set at each sampling site, left overnight and collected the next morning. At the laboratory, fish were fileted, the skin removed and the filets divided for analysis of cooked and uncooked tissue. Filets were wrapped in aluminum foil and cooked in a 105°C oven for 30 minutes. Samples were then analyzed using U.S. EPA method 8080.

Water and sediment samples were collected in the immediate vicinity of the crayfish traps. Surface water was collected using laboratory sample containers and a simple grab technique. Sediments were collected using a Van Veen grab ($0.04m^2$) at each site.

Results and discussion

Crayfish and fish tissues. At GWP and the LUSP site, fish captured and analyzed (six from each site) were yellow perch, *Perca flavescens*. At the SLU and PB site pumpkinseed (*Lepomis gibbous*) were also included in the fish tissue analysis for these two stations. These two fish species are pelagic and may be considered similar in their feeding habits.

Yellow perch ranged in length from 18 to 21 cm at all sites. Pumpkinseeds from SLU and PB (five fish from each site) ranged from 10 to 14 cm in length. Crayfish captured at all sites ranged in weight from 21 to 74 grams. Approximate numbers of crayfish captured at each site and submitted for analysis were: GWP, 30; SLU, 10; LUSP, 40; and PB, 10.

The highest PCB levels in Lake Union crayfish tissue were observed at LUSP and SLU. The highest PCB concentrations in fish were found at SLU and GWP (Table 1). Levels of total PCBs (mean of cooked and raw tissue) were not significantly different (two sample t-test; $p>0.20$) between Lake Union fish and crayfish muscle tissue.

For fish and crayfish tissue, PCB levels were, in all but one case, lower in raw than in cooked tissue. For fish, the PCB levels in raw tissue from all four stations were not significantly different (two sample t-test, $p>0.50$) from that in cooked tissue. However, in crayfish, the levels of PCBs were significantly different ($p<0.05$) for cooked (215 ± 91; $x \pm SD$) versus raw (75 ± 38) tissue.

Past studies of Lake Union crayfish have also reported higher PCB levels in cooked than in raw crayfish tissue (Table 2). Possible explanations for these observations have been offered by several reviewers (City of Seattle 1986c) and include; (1) evaporation of water from tail tissue during cooking results in higher PCB concentrations, (2) possible migration of PCBs from hepatopancreas to muscle tissue during cooking, and (3) age differences between cooked and uncooked specimens.

Evaporation does not appear to be a plausible explanation since the difference in percent moisture between cooked and raw crayfish tissue in this study was shown to be less than 2 percent. Age differences also do not appear to explain these findings since in this study they were accounted for by equally dividing size classes of crayfish for cooked and uncooked tissue.

Migration of PCBs from crayfish hepatopancreas to tail muscle tissue may provide a possible explanation for the differences observed in raw versus cooked tissue PCB levels. Previous studies have shown whole body crayfish (raw) PCB levels in Lake Union to be from 1.5 to more than 13 times the level found in cooked tail muscle tissue (Frost et al., 1985).

The PCB level in control crayfish (raw) collected from nearby Lake Sammamish was below the method detection limit (i.e., <20 ppb) thus reflecting this lake's relatively non-industrialized and apparently uncontaminated drainage area.

<u>Sediments and water</u>. Surficial sediment (0-5 cm) concentrations of total PCBs were highest at LUSP. At the other three stations, PCBs were below or only slightly above detection limits (Table 2). Previous sediment investigations showed higher (and lower) PCB levels

Table 1. Total PCBs recovered from Lake Union crayfish, fish, water column, and sediments. Samples collected August 12 and 13, 1987. Crayfish were also collected from a control site, Lake Sammamish, on August 12, 1987.

	Total PCBs, ug/Kg dry weight and (wet weight)				
	Lake Union Steam Plant (LUSP)	South Lake Union (SLU)	Gas Works Park (GWP)	Portage Bay (PB)	Lake Sammamish
Crayfish (raw)	90(16)	<40(<7)	120(21)	<50(<9)	<20(<4)
Crayfish (cooked)	240(47)/200[a](39)/310[a](60)	280(50)	<50(<10)	210(42)	--
Fish (raw)	70(16)/108[b](27)/132[b](33)	240(52)	360(84)	115(23)	--
Fish (cooked)	105(34)	330(104)	400(120)	150(41)	--
Water	<1	<1	<1	<1	--
Sediment	2300	<20	<100	10	--

[a]Replicate results

[b]Replicate results from second laboratory

Table 2. PCB levels in Lake Union crayfish tail muscle tissue observed in this and previous studies.

	Total PCBs, ug/Kg (dry weight)[a]					
	Lake Union Steam Plant Crayfish		South Lake Union Crayfish		Gas Works Park Crayfish	
	Raw	Cooked	Raw	Cooked	Raw	Cooked
1984 Data[b]	---	---	---	---	1310	850
1986 Data[c]	450	3300	<200	300	---	---
1987 Data[d]	90	250	<40	280	120	<50

[a]1984 and 1986 data converted from wet weight to dry weight assuming 80 percent moisture in crayfish tail muscle tissue as measured in this study.

[b]Reference: Table 5 in Frost et al., 1985. Raw value is for whole crayfish; cooked value is for tail muscle tissue.

[c]Reference: Table 6 in City of Seattle, 1987.

[d]Reference: This study. LUSP cooked value is the mean of three composites.

in LUSP sediments, higher levels in SLU sediments, and similarly low levels at GWP (Hileman et al., 1985; City of Seattle, 1986a; 1986b; 1987).

The measured PCB concentration (2300 ug/Kg) in sediment from the Lake Union Steam Plant exceeded the proposed U.S. Environmental Protection Agency (EPA) screening level concentration (290 ug/kg) for freshwater sediments (Neff et al., 1986). Screening level concentrations are based on chemical and biological synoptic field data and represent the estimated highest sediment concentration of a contaminant that can be tolerated by 95 percent of benthic infauna. Based on this and previous Lake Union sediment PCB investigations it appears that benthic infauna, at least near the Lake Union Steam Plant, may be perturbed by the elevated PCB concentrations present.

Water concentrations of total PCBs were below the method detection limit (<1 ppb) for all Lake Union stations sampled thus reflecting their low solubility in water.

Human health concerns. In order to protect public health, the U.S. Food and Drug Administration (FDA) has established a regulatory tolerance of 2000 ppb PCBs in fish and fishery products (Title 21, Code of Federal Regulations). Based on this PCB tolerance level, crayfish and fish collected from Lake Union during this study meet U.S. FDA regulatory requirements for human consumption. (Because the FDA tolerance level is based on the total PCB concentration in wet or undried tissue, the wet weight PCB concentrations noted in Table 1 are used for comparison.)

Differences in consumption rates and edible muscle tissue concentrations of PCBs may alter the health risk factors involved. Because FDA levels are intended to regulate toxicants in fish and fishery products traded in interstate commerce as part of a national market, such levels may be too lenient for protecting the health of a local population that consumes large quantities of fish and crayfish from Lake Union.

Seasonal differences in PCB accumulation for fish and crayfish have not been evaluated in this or in previous Lake Union studies. Given possible seasonal variations, the PCB values reported here may be low when compared with tissue samples collected in the spring of the year (Edgren et al., 1981). Thus, while PCB levels did not exceed FDA tolerance limits for the Lake Union fish and crayfish tissues examined in this study, seasonal differences may alter such findings.

Conclusions

Lake Union crayfish and finfish collected from four sites in August 1987 exhibited measurable levels of PCB contamination in their edible (muscle) tissue. Dry weight tissue concentrations of total PCBs ranged from <40 to 280 ppb (<7 to 50 ppb, wet weight) in raw and cooked crayfish and from 70 to 400 ppb (16 to 120 ppb, wet weight) in raw and cooked finfish. These levels are lower than those observed in previous studies of Lake Union crayfish and remain well below the U.S. FDA tolerance level established for human consumption of fish and fishery products. However, ingestion of large amounts of fish and crayfish from Lake Union may increase the health risk factors involved.

Implications of the PCB Data for the Lake Union and Ship Canal Water Quality Management Program and Recommendations for Further Research

Data from the 1984 and 1986 studies of lake Union crayfish contamination and from the 1987 PCB study indicate that Lake Union sediments, crayfish and finfish are contaminated with PCBs. PCBs are primarily a historical Lake Union pollutant as a result of their use in electrical transformers, plasticizers, lubricating fluids, fluorescent light ballasts, adhesives, printing ink, carbonless copy paper, and pigments. Although the manufacture of PCBs was banned by EPA in the late 1970s, these chemicals are persistent in the environment (Washington Department of Social and Health Services, 1987).

PCBs are one of the pollutants of concern in Lake Union. Their levels are relatively high in sediments from the south end of the lake, i.e., up to an order of magnitude higher than the EPA screening level concentration for freshwater sediments (Neff et al., 1986). Although PCB levels in Lake Union crayfish and finfish do not exceed the FDA tolerance level of 2000 ppb, this does not mean that there are no potential human health risks from the consumption of crayfish; the tolerance level is not based on regular consumption of crayfish from local waters.

However, PCBs are not the only or the major contaminant group in Lake Union. Heavy metals such as arsenic, chromium, copper, lead, nickel, and zinc enter the lake on an ongoing basis from sewage, stormwater and industrial waste. PAHs such as benzo(a)pyrene, fluoranthene, naphthalene and pyrene are found at very high concentrations up to thousands of parts per million, in the sediments offshore of Gas Works Park as a result of the former operations of the Seattle Gas Plant (Hileman et al., 1985; City of Seattle, 1986a; City of Seattle, 1986b). Overall metal concentrations and overall PAH concentrations are twice as high in Lake Union sediments as in sediments from Lake Washington which is less industrialized (Galvin et al., 1984). Concentrations of individual metals in sediments from south Lake Union and offshore of Gas Works Park are up to 22 times as high as in sediments from Chester Morse Lake which is located within the protected watershed of the Seattle Water Department. Elevated metal and PAH concentrations in sediments offshore of Gas Works Park have been associated with high amphipod mortality and decreased abundance and diversity of benthic infauna (Yake et al., 1986).

The Interim Action Plan for Lake Union and the Ship Canal focuses on source control actions to reduce the amount of pollution currently entering the lake from sewage, stormwater, industries and boats. The Action Plan also addresses management of contaminated sediments from past pollution. Lake Union PCB data alone do not indicate the need for remedial action. In conjunction with data on other pollutants such as metals and PAHs, the PCB data support the concept of a pilot or demonstration capping project in an area of Lake Union where there is known contamination. As part of the Action Plan, the City of Seattle intends to investigate the feasibility of such a pilot capping project. Candidates for this pilot project include sediments in south lake Union as well as sediments offshore of Gas Works Park. Placing a clean layer of sand over the contaminated sediments would isolate toxics including PCBs from aquatic biota and from recreational users of the lake.

If a pilot capping project were found to be feasible and were implemented, followup monitoring would be required to determine its effectiveness. If, over time, the sediments deposited above the cap showed PCB contamination, this would indicate ongoing migration of PCBs into Lake Union.

In addition to investigating the feasibility of a pilot capping project in Lake Union, two other areas are recommended for further research. Because there may be seasonal variations in uptake and bioaccumulation of PCBs by crayfish and finfish, sampling crayfish and finfish on a seasonal or monthly basis would present a more complete picture of PCB contamination in Lake Union and associated risks to aquatic biota or human health.

PCBs are known to bioaccumulate in the food chain even when sediment and water column concentrations are low (Wilson and Forester, 1978; Washington Department of Social and Health Services, 1987). More work is required to evaluate the potential for crayfish to bioaccumulate PCBs from Lake Union water and sediments and for finfish to bioaccumulate PCBs from crayfish.

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MODES OF TOLUENE UPTAKE IN ADULT CHINOOK SALMON[1]

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Abstract

To determine the rate and modes of toluene uptake in pre-spawning chinook salmon (*Oncorhynchus tshawytscha*), fish subjects were exposed for 3 hours to sublethal tritium-labeled toluene concentrations (1.0, 0.2, 0.01 mg/l) in salt water in a flow-through (non-recirculating) system. Blood samplings from the heart, dorsal aorta, and caudal vein were performed at selected time intervals to determine the following parameters: toluene accumulation in blood relative to exposure concentration; rate of toluene uptake across gill lamellae;rate of toluene uptake across the mucus/skin; and toluene tissue burdens after 3 hours of exposure. At each exposure concentration, toluene reached levels of at least 400 times the exposure concentration in the blood after several minutes of exposure. Gill lamellae uptake was extremely rapid, and was the primary mode of uptake. A secondary mode of uptake was across the skin, with toluene appearing in the blood after one to two hours of exposure. Relative tissue burdens of toluene after 3 hours were in the order of blood > gall bladder > liver > muscle. The findings indicate the rate of toluene uptake is concentrtion dependent, with significant amounts of toluene and other MAH potentially accumulating in salmon in a short period of time at concentrations commonly found near oil spills and urban areas.

Background

Planned oil exploration along the continental shelf off the Washington coast in the late 1980's increases the probability of petroleum hydrocarbons contaminating a relatively pristine environment and potentially having a negative impact on the fisheries in the region. Any oil production activity will involve the risk of an oil spill which may occur from refineries, ships, pipelines, and wells.

Monoaromatic hydrocarbons comprise ten to thirty percent of crude oil (McAuliffe, 1977; Cohen et al. 1980), and are employed in industry as reagents in the production of a

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variety of organic compounds, such as polymers, detergents, pesticides, and degreasers. They are also natural components in gasolines. Benzene, toluene, and xylene are three of the most important MAH used in industry. MAH are classified as petrochemicals because they are derived from petroleum or natural gas. Some physical characteristics making them deserving of more attention in pollution research is that they are both highly toxic and water-soluble when compared with other classes of petrochemicals (Whipple et al. 1981). Marine and freshwater measurements of MAH have been generally lacking due to the assumption that they are too volatile to persist in the aquatic environment for significant periods of time and therefore pose little threat to aquatic organisms.

However, some solubilization will occur, and because of their high toxicity, the impact of MAH on the marine environment is greater than simple mass balance considerations would imply. Data from an American Petroleum Institute report (API, 1978) indicate that total MAH measured in refinery effluents can range from only trace amounts up to 100 ppb. The data also indicate that MAH concentrations in intake water are sometimes higher than in effluent water. The latter fact suggests that there may be chronic levels of MAH in water in the 100 ppb range near many urban areas. Another study (Blumer and Sass, 1972) has shown that MAH did not completely volatilize in an actual petroleum spill. Rather, MAH entered the water column to be transported to the sediments and deposited there, leaching into the water column over several seasons.

Since marine organisms are often contaminated by MAH after oil spills, biologists have undertaken a variety of field and lab studies to determine the toxicokinetics (rate and modes of uptake, accumulation, and depuration) of these compounds. To achieve this, radiolabeling MAH with isotopes such as tritium and carbon-14 and measuring the radioactivity resulting from accumulation of isotope labeled compounds in tissue or fluid has been the common practice.

The purpose of this investigation was to approximate conditions for one highly toxic component (toluene) of the water-soluble fraction (WSF) of crude oil as would be observed in the water column under a crude oil spill in marine waters, and to determine in part the toxicokinetics of this substance in salmon.

Methods

Fifty Fall Snake river chinook salmon of 1983 brood year were obtained from the University of Washington Big Beef Creek Hatchery in June of 1987. Fish were raised in freshwater but were adapted to seawater (smolted) six weeks before exposure tests. The chinook were held indoors in 2000 gallon circular tanks under dark conditions in sand-filtered seawater (particulates > 200 µm removed). Fish used in the study were females and were fed daily with a dry, pelleted formula (Cenex, Tacoma, WA).

Figure 1 is a schematic of the flow-through exposure system which solubilized and diluted ^{3}H-toluene (specific activity: 1.11 x 10^6 pCi./ml, New England Nuclear) to obtain approximate ^{3}H-toluene concentrations of 1000, 200, and 10 ppb (w/v). Exposure water was sampled for analysis by gas chromatography (GC) using the gas equilibrium techniques developed by McAuliffe (1971). A separate aquarium next to the exposure tank was used as an anesthesia chamber. Using exposure water from the second mixer, a 1:60,000 concentration of tricaine methane sulfonate (MS-222, Argent Chemical Laboratories, Redmond, WA) was used to anesthetize fish prior to their transfer to the sponge wedge fish holder for blood sampling.

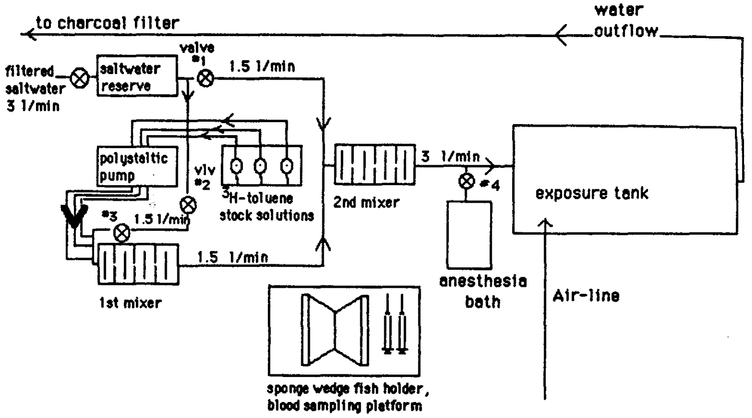

Figure 1. Schematic of exposure system.

Blood samples (0.2 ml) were taken from the dorsal aorta, heart, and caudal vein by a 1 cc glass tuberculin syringe with heparinized needle (23 guage). Dorsal aorta blood was sampled at 0, 1, 2 min, and 1, 2, 3 hr, with procedures following Shiffman (1959). Heart blood was sampled at 0, 1, 2 min, with procedures following Klontz and Smith (1968). Caudal vein samples were at 1, 2, and 3 hr, and were accomplished by a 45° anteriorad needle insertion at a point two scales below the lateral line in the region of the peduncle. Blood was transferred from the syringe into glass crew cap scintillation vials containing 2.4 ml of NCS tissue solubilizer (Amersham-Searle). The sample was heated at 50°C for 20 minutes to facilitate solubilization. After the samples cooled, 0.8 ml of a saturated solution of benzoyl peroxide in toluene was added as a decolorizer and the sample was heated at 60°C for 20 minutes. Upon cooling, 18 ml of Aquasol scintillation cocktail (New England Nuclear) was added to each vial, and the vial was placed in complete darkness for at least 24 hours before being counted on a Beckman 200 liquid scintillation counter.

Once the exposure was completed, fish were sacrificed by suffocation and immediately frozen at -70°C until the tissues could be dissected. Upon dissection, 1 g each of liver, gall bladder, and muscle tissue was weighed and solubilized using the blood solubilization procedure. Care was taken during dissection to minimize potential cross-contamination among target tissues. All solubilized tissue samples were acidified to pH 5 with glacial acetic acid to improve tritium counting efficiency by liquid scintillation. After accounting for photon quenching, total tritium activity (pCi) in blood and tissue (pCi/ml or pCi./g) was converted to µl/ml or µl/g.

Results

The measured ^{3}H-toluene water concentrations for the exposure studies are listed in Table 1. Oxygen levels, water temperature, and behavioral observations of the fish subjects are also listed in Table 1.

Levels of ^{3}H-toluene in blood reached almost constant levels within 5 minutes of exposure (Fig. 2). Blood levels of ^{3}H-toluene were 400-1100 times that of exposure concentrations. Subsequent analyses of blood samples showed hematocrit contained 70% of the total toluene in

Table 1. Summary of exposure levels, exposure tank oxygen levels and temperature, and observed fish behavior to ^3H-toluene. Three replications per concentration.

Hour	Exposure Concentrations (ppb)	(S.D.)	Oxygen (mg/l)	Temp. (C)	Opercular Movements/min	Behavior
	1000					
0	1040	(26)	7.2	13.6	60	shuddering, darkening
1	986	(32)	6.6	13.0	100	of color disequilibrium,
2	971	(24)	6.5	12.5	85	lack of motion
3	924	(13)	6.1	12.5	85	disequilibrium, lack of motion
	200					
0	270	(12)	7.2	14.5	60	
1	235	(9)	6.7	13.7	110	hourly behavior same as
2	222	(13)	6.3	12.8	90	in 1000 ppb study
3	219	(6.2)	6.2	12.6	85	
	10					
0	19	(2)	7.2	13.3	60	
1	N.D.[a]		6.4	13.0	110	hourly behavior
2	N.D.		6.6	12.9	85	same as in 1000
3	N.D.		6.3	12.7	85	ppb study

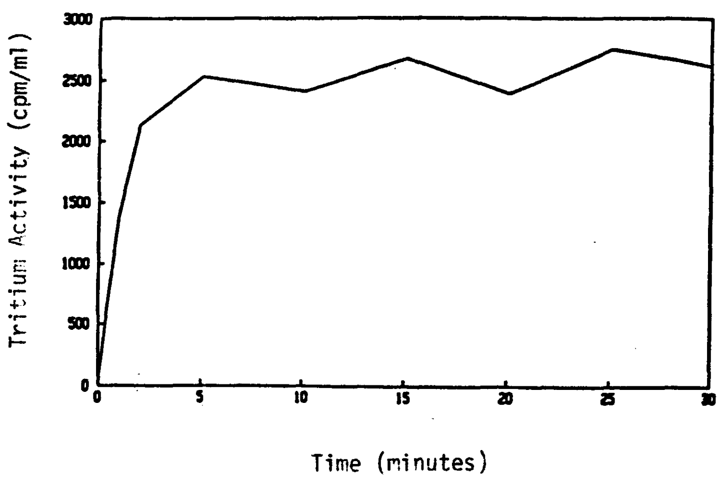

Figure 2. Dorsal aorta ^3H-toluene levels at 1000 ppb in the first half-hour of exposure as measured by liquid scintillation.

[a]Not detectable.

the blood, with plasma having the remainder. Statistical analyses showed that ^{3}H-toluene blood concentrations were of the following order: 1000 ppb > 200 ppb > 10 ppb.

A regression analysis was performed on the 0, 1, and 2 minute dorsal aorta ^{3}H-toluene concentrations to obtain an approximation of the ^{3}H-toluene uptake rate, which was the slope of the regression line. The ^{3}H-toluene uptake rates were calculated to be 0.56 µl/min at 1000 ppb, 0.12 µl/min at 200 ppb, and 0.005 µl/min at 10 ppb.

Levels of ^{3}H-toluene in the three locations sampled in the salmon's circulatory system showed that heart and dorsal aorta blood concentrations were equivalent after 5 minutes exposure. Caudal vein concentrations were significantly larger ($P<0.05$) than dorsal aorta concentrations after one to two hours of exposure.

Of the tissues analyzed for ^{3}H-toluene, the following distribution (total organ/tissue weight) was found: blood, 91%; gall bladder, 6%; and liver, 3%. The concentration of ^{3}H in muscle was below detection limits.

Discussion

Chinook salmon appear to be extremely efficient accumulators of MAH, accumulating toluene in their blood 400 to 1100 times that of water concentrations. The only other study (Benville et al. 1980) to look at blood MAH levels in fish showed striped bass (*Morone saxatilis*) in 1000 ppb of benzene accumulating this compound to levels of up to 100 times that of water concentrations over a period of several hours. Benzene uptake rate in striped bass was lower than toluene uptake in chinook salmon; the bass requiring up to one-half hour to achieve blood equilibrium levels of benzene, while the chinook required only a few minutes to achieve ^{3}H-toluene blood equilibrium levels. Differences in accumulation of these two MAH are possibly due to interspecific physiological differences between striped bass and chinook salmon and differences in the molecular structures of benzene and toluene.

Rate of toluene uptake appeared concentration dependent, as was indicated by the significantly higher rate at 1000 ppb, while the rate at 200 ppb was significantly higher than that at 10 ppb. This may be due to sheer availability of the compound in the water at higher concentrations.

At what concentration maximum rate of uptake occurs remains to be answered.

One purpose for sampling at different points in the salmon's circulatory system was to try to gain evidence of the importance of the various routes by which toluene will enter the fish. Since the greatest surface area to water in the fish are the gill lamellae, a strong possibility exists for diffusion across lamellae to be the major mode of uptake. Since toluene is highly lipophilic, the partition coefficient would favor rapid movement of toluene from water into lipid-rich salmon blood, separated from the water by only one or two cell layers in the gill lamellae. Preliminary results from this study indicate rapid diffusion through the lamellae.

Another mode of uptake but one which is likely to be of secondary importance in terms of total volume of toluene accumulated in the fish is via the lateral line scale pores. Most fish in this study had higher concentrations of ^{3}H-toluene in the caudal vein (muscle blood output) than the dorsal aorta (muscle blood input) after two hours exposure. The extra toluene most probably came from the peripheral lymphatic duct, which also connects to the caudal vein. The peripheral lymphatic duct is connected to the lateral line scale pores via lymphatic tubules, and therefore would possibly receive some ^{3}H-toluene through direct contact with toluene-tainted water. Diffusion through the skin appears to be much slower than gill uptake; ^{3}H-toluene from the lateral line scale pores was only detectable after two hours of exposure.

The apparent rapid removal of toluene from salt water as shown in this study leads to the conclusion that salmon would only require several minutes under an oil spill to accumulate MAH to a degree which may lead to fish tissues being rendered unpalatable (off-odor and/or off-flavor). Although MAH appear to be rapidly depurated from fish tissues after termination of exposure (Roubal et al. 1977; Whipple et al. 1981) this process can take several weeks to achieve complete depuration.

Acknowledgments

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Uptake of Toxic Chemicals by Salmon in an Urban Estuary

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INTRODUCTION

Estuaries provide important habitats for juveniles of all five of the most important Pacific salmon species (Healey, 1982). A number of urban estuaries in Puget Sound are used during the down-stream migration of wild and hatchery-reared juvenile salmonids, including the Duwamish Waterway in Seattle, the Puyallup Waterway in Tacoma, and the Snohomish River in Everett. Municipal and industrial wastes enter these urban estuaries from point and non-point sources and accumulate in bottom sediment and associated ecosystems.

The Duwamish Waterway is perhaps the most chemically contaminated of these estuaries. The bottom sediments contain hundreds of toxic chemicals, including aromatic hydrocarbons (AHs) and polychlorinated biphenyls (PCBs) (Malins et al. 1984). Bottomfish and bottom-dwelling invertebrates in this waterway bioaccumulate substantial levels of PCBs and AHs. In addition, a number of bottom fish species have a variety of diseases which are not found, or are found at significantly lower prevalence, in individuals of the same species from minimally contaminated areas of Puget Sound.

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Of all the species of Pacific salmon, chinook salmon (Oncorhynchus tshawytscha) are most dependent upon estuaries as a feeding ground (Thom, 1987). The usual residence time for juveniles of this species in the Duwamish Waterway is about two months (early April to early June) (Meyer et al. 1981). Most of these juveniles are released into the Green River/Duwamish Waterway system from the hatchery at Soos Creek operated by the Washington State Department of Fisheries (Meyer et al. 1981). During their time in the estuary, they feed on a variety of prey organisms, including epibenthic crustaceans, such as copepods, gammarid amphipods, mysids, cumaceans, and insects (Healey, 1982, Simenstad et al. 1982).

Therefore, it is possible that juvenile chinook salmon during their residency in the Duwamish Waterway bioaccumulate substantial levels of toxic chemicals and exposure to these chemicals could lead to adverse effects on the health and survival of this important marine resource. We report here the results of a study which addresses the first part of this hypothesis: "Do juvenile chinook salmon in the Duwamish Waterway bioaccumulate toxic chemicals?"

METHODS

Fish Sampling

Similarly aged juvenile chinook salmon were collected at four sites: the Duwamish Waterway, the Nisqually River, the Kalama Creek hatchery of the Nisqually River and the Soos Creek Hatchery on the Green-Duwamish River system . The Nisqually River Estuary, located in a rural region of southern Puget Sound, served as a reference site. Concentrations of AHs and PCBs in sediments from this estuary have been reported to be at or below the limits of detection (McCain et al. 1988). Fish at the estuarine sites were captured with a 30 m beach sein. Samples of liver, stomach contents (food organisms), and bile were collected for chemical analyses. Due to the small size of the fish (mean lengths of 78 to 96 mm), all samples were composited, 10 to 30 specimens per composite. Prior to chemical analysis, selected composites of stomach contents were examined with a dissecting microscope in order to estimate their taxonomic composition.

Chemical Analyses

Stomach contents and liver tissue were analyzed using capillary column gas chromatography with mass spectrometry (MacLeod et al. 1985). A high-pressure liquid chromatographic/fluorescence detection technique (Krahn et al. 1986) was employed to measure metabolites of aromatic compounds in bile. This technique was used because analyses of AHs (e.g., components of fossil fuels and their combustion products) in tissues are of limited value due to the extensive metabolism of these compounds, especially in the liver (Varanasi et al., 1987).

RESULTS

The mean concentrations of AHs and PCBs in the stomach contents (primarily cumaceans, amphipods and fish tissue) of salmon from the Duwamish Waterway were 650 times and 4 times, respectively, higher than those in salmon from the Nisqually River (Figures 2A and 2B). Similarly, the mean concentration of bile metabolites of aromatic compounds which fluoresce at benzo(a)pyrene wavelengths was 24 times higher in the urban salmon compared to salmon in the Nisqually River, and 10 times higher compared to salmon from the Nisqually Hatchery (Figure 2C). The mean concentrations of PCBs in livers of fish from the Nisqually Hatchery and the Nisqually River Estuary were not significantly different ($p \leq 0.05$) (Figure 2D). The mean concentration of PCBs in livers of Duwamish Waterway salmon (2,600 ± 560 ng/g dry weight), however, was approximately 10 fold higher than that for livers of fish from the Green River Hatchery and significantly ($p \leq 0.05$) higher than that of salmon from the Nisqually River Hatchery or Estuary (Figure 2D).

DISCUSSION

The results of this investigation clearly demonstrated that juvenile chinook salmon captured in the Duwamish Waterway had been exposed to substantially higher concentrations of toxic chemicals than had juvenile chinook salmon from the Nisqually River estuary. However, we have not had the opportunity to study the second part of the above-mentioned hypothesis: "Is the bioaccumulation of toxic chemicals by juvenile chinook salmon in the Duwamish Waterway related to adverse biological effects?" A large number of variables can influence the survival of juvenile salmon, making it very difficult to assess the effects of estuarine pollution on these salmon. For example, the growth and survival rates of juvenile salmon could be lowered by such factors as predation and availability of food organisms, as well as environmental contamination. Nevertheless, additional investigations on the effects of estuarine pollution on juvenile salmon, including chinook salmon, in the Duwamish Waterway as well as in other contaminated estuaries in Puget Sound are needed. These investigations should include both laboratory and field studies. Laboratory studies would be necessary for further definition of the rates of uptake and depuration and the effects of chemicals identified in this study. Field studies should involve an assessment of the health status of exposed juvenile salmon. Indicators of health status would include changes in growth rate and behavior, the presence of pathological conditions, and changes in parameters indicative of organ dysfunction. These types of studies will be technically very difficult to perform because the effects on juvenile salmon of exposure to estuarine pollution for relatively short periods of time (i.e. one to two months) are likely to be subtle. Therefore, these investigations would have to evaluate if any of these subtle changes can affect the long-term survival of juvenile salmon. The results of these laboratory and field studies may increase our understanding of whether polluted estuaries may be one of the contributing factors to mortalities in juvenile salmon. Information from these studies should also aid environmental managers in making sound decisions regarding the regulation of discharges of industrial and domestic wastes into estuaries which are important salmon habitats.

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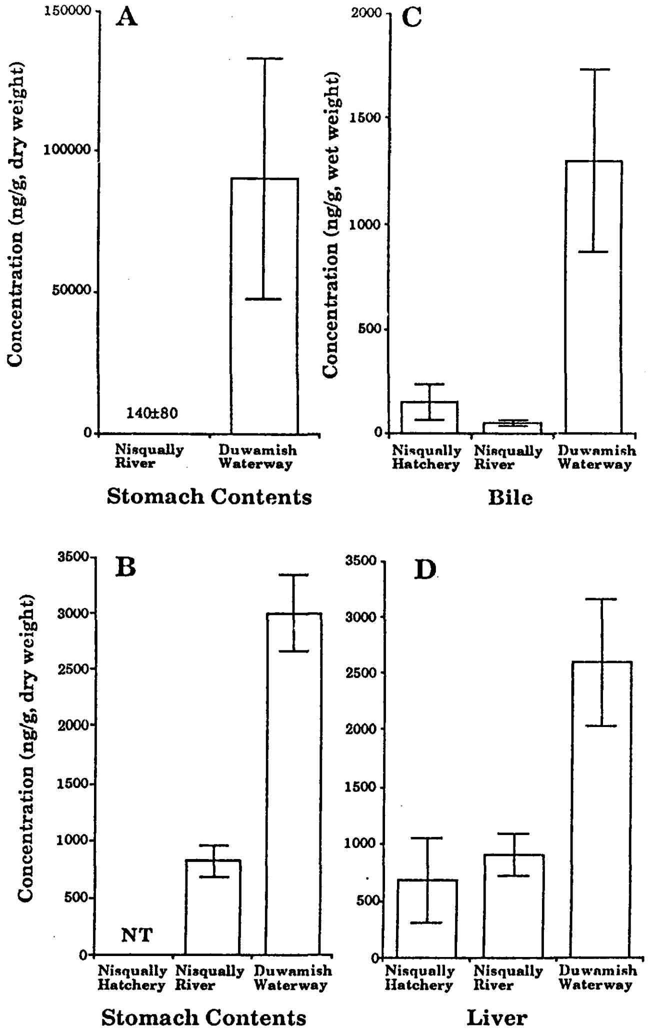

Figure 1. Mean concentrations of chemicals in juvenile chinook salmon from various Puget Sound sites: (A) aromatic hydrocarbons [summed concentrations of 17 individual compounds] in stomach contents, (B) PCBs in stomach contents, (C) metabolites (measured at benzo [a]pyrene fluorescent wavelengths) in bile, and (D) PCBs in livers. Each value is the mean of analyses of two to three composite samples; each composite contained specimens from 10 to 35 individual salmon.

LIVER CARCINOGENESIS IN ENGLISH SOLE FROM PUGET SOUND: THE IMPORTANCE OF NEOPLASIA-ASSOCIATED HEPATIC LESIONS AS INDICATORS OF CONTAMINANT EXPOSURE

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Introduction

It is extremely difficult to reliably assess the effects of toxic chemicals through field studies on animals in the marine environment because of the inherent lack of control over a multitude of factors, including natural variations in population and community structures, and the genetic background and diet of potential indicator organisms. However, field studies utilizing microscopic examination and disease diagnosis of fish liver as a tool for assessing the deleterious effects in bottom-dwelling fish species resulting from exposure to chemical contaminants have been, and continue to be, instrumental in identifying the geographic areas and resident fish populations in Puget Sound adversely affected by chemical pollution.

Extensive, long-term studies in Puget Sound (McCain et al., 1982; Malins et al., 1984,1985a,1985b,1987a; Myers et al., 1987) have comprehensively assessed and described the broad spectrum of hepatic lesions affecting several bottom-dwelling marine fish species from over 50 sites that show a broad range of sediment pollutant profiles and concentrations, while focusing on the English sole (*Parophrys vetulus*) as the indicator organism. The evidence from these studies has provided a firm basis for the

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hypothesis that liver tumors and other idiopathic (of un-unknown causation) liver lesions in English sole are the result of exposure to sediment-associated chemical contaminants, especially polycyclic aromatic hydrocarbons. In fact, these studies along with data on uptake and metabolism of certain classes of chemical contaminants (Krahn et al., 1984, 1986; reviewed in Varanasi et al., 1987) have served to increase public awareness of pollution problems in Puget Sound.

The liver of mammals and especially fish is well known as a target organ for the effects of xenobiotic toxicants and carcinogens. Consequently, our recent efforts have concentrated on the detailed evaluation and analysis of a spectrum of lesions in the liver of fish species exposed to toxicants in the marine environment, concentrating on the English sole as a model. Until recently, there has been a preoccupation with hepatic neoplasms in English sole as the primary condition indicative of exposure to toxic and carcinogenic chemicals, to the exclusion of other types of hepatic lesions that 1)can result from experimental exposure to hepatocarcinogens and 2)also may be involved in the stepwise progression toward hepatic neoplasms. The objective of the study reported here was to determine if certain lesions that precede neoplasia in experimental mammalian and fish liver carcinogenesis models can be reliably used as early indicators of biological damage in feral fish species, especially in younger fish and in those species that show high prevalences of these lesions but low prevalences of hepatic neoplasms.

Background

By applying analytical epidemiologic techniques to the study of English sole neoplasia and its relation to chemical contaminant exposure, we have constructed a model that incorporates neoplasm prevalence, sediment contaminant levels, interactions between different classes of chemicals, and fish size (an indirect measure of fish age) from multiple studies in Puget Sound. This model is discussed by Schiewe et al. in this volume. These results strengthen the hypothesized role of aromatic hydrocarbons (AHs) in sediment as a major etiologic agent of liver neoplasms in English sole and point out the significant positive influence of fish age on neoplasm prevalence.

A basic tenet of analytical epidemiology holds that, while the strength of a particular statistical association in a study is important in establishing a relationship, of perhaps greater importance is the consistency of that relationship among multiple studies of a similar type. Accordingly, we have demonstrated a consistent, positive association between liver neoplasm prevalence and sediment-associated aromatic hydrocarbons across six separate field studies (Malins et al., 1988, Schiewe et al., this volume). A strong, consistent association of the same type is also established for the hepatic lesion types megalocytic hepatosis (MH) and foci of cellular alteration (FCA). This latter finding further supports the value of these neoplasia-associated lesion types as indicators of pollution exposure.

We have also previously documented significant associations between levels of flourescent aromatic compounds in bile of English sole and prevalences of idiopathic liver lesions (Krahn et al. 1984; 1986). Although the positive correlation between bile metabolites and hepatic neoplasms has received more attention, equally strong correlations exist between these metabolites and the lesions, MH and FCA.

In addition to the studies discussed above, we recently published an epidemiologic analysis of a multiyear study (1979-84) investigating the significance of certain potential risk factors associated with liver lesion prevalences in English sole, including age, gender, and site, season and year of capture (Rhodes et al., 1987). The results of this study are summarized in Table 1, showing odds ratios for particular risk factors as applied to the major liver lesion categories. The results of this study suggest that certain risk factors inherent to the Duwamish Waterway (e.g. chemical contamination) significantly influence the prevalence of not only neoplasms, but also preneoplastic FCA, and the degenerative conditions of nuclear pleomorphism(NP) and MH. Other less polluted urban sites in Elliott Bay (N. Seattle waterfront, S. Seattle waterfront) show elevated odds ratios only for NP and MH. Fish age emerges as a significant risk factor for all lesion categories with the exception of NP/MH. The importance of other neoplasia-associated lesions as indicators of pollutant exposure is especially clear considering the higher prevalences of these non-neoplastic lesions in fish at the younger age classes (Rhodes et al, 1987). The consideration of fish age and liver lesions other than neoplasms is, therefore, of obvious importance in any investigation designed to identify appropriate bioindicators of environmental pollution exposure.

Considering this background information relating to hepatic lesions in English sole and their relation to chemical contaminant exposure and fish age, we conducted a detailed analysis of the process of liver neoplasia in English sole in order to further develop a model of hepatocarcinogenesis and to identify the significance of lesions that may precede neoplasms in this species. This knowledge may also be useful in assessing the utility of these lesion types as indicators of chemically induced stress in young English sole and in other species of fish.

Results and Discussion

Liver lesions: geographic distribution and patterns of lesion co-occurrence in English sole

The English sole displays a broad spectrum of multiple co-occurring liver lesions (Table 2) that closely parallels (Myers et al., 1987) the degenerative, regenerative, preneoplastic and neoplastic liver lesions induced experimentally by chemical hepatotoxins and hepatocarcinogens in the mouse, rat (Farber and Sarma, 1987) and certain fish species (Hendricks et al., 1984; Couch and Courtney, 1987). These lesions are found almost strictly, or at significantly higher prevalences in English sole captured from polluted embayments and estuaries in Puget Sound

(Malins et al., 1984). This pattern is closely paralleled by similar lesions in several other bottom dwelling species, including rock sole (Lepidopsetta bilineata) and starry flounder (Platichthys stellatus) (Myers and Rhodes, 1988). However, the English sole is the most effective indicator species due to the higher prevalences of toxico- pathic liver lesions detected in this species relative to other fish species resident to Puget Sound.

To clarify the role of these lesion types within the context of liver neoplasia in English sole, we began with the following general facts and observations: 1)most hepatocarcinogens are also cytotoxic; 2)many of the observed lesion types co-occur in individual fish; 3)certain lesion types in addition to neoplasms tend to primarily affect fish captured from polluted waters; and 4)some lesion types are found mainly in adult fish while other types also affect younger fish. By statistical analysis of lesion type co-occurrence in English sole from Eagle Harbor, we were able to show that certain lesions may be temporally related to one another, based on the assumption that lesions which tend to occur together are more likely to precede or follow one another in a sequence of lesions progressing toward hepatic neoplasms (Myers et al., 1987). This type of analysis represents our best attempt to make some temporal sense of the profusion of multiple hepatic lesion types we typically encounter in feral English sole, a pattern which is consistent with what one would expect under a situation of continuous exposure to chemical hepatocarcinogens.

The significant patterns emerging from this type of analysis are shown in Figure 1. These results are consistent with the experimentally determined temporal histogenesis of hepatic neoplasia in the mouse and rat (Farber and Cameron, 1980; Farber and Sarma, 1987) and certain hatchery fish species (Hendricks et al., 1984). Within this scheme, megalocytic hepatosis (MH) is viewed as the initial degenerative lesion resulting from the cytotoxic effects of the hepatocarcinogens English sole are exposed to in the sediment. This is because it appears first in fish less than one year in age, is associated with other degenerative conditions of the liver, is the most common idiopathic lesion detected in sole from polluted waters, and appears identical to the initial lesions resulting from experimental exposure to certain hepatocarcinogens in the mouse, rat, and certain fish species. MH is strongly associated with hepatocellular regeneration, manifesting the compensatory proliferative response to the liver degeneration and necrosis seen in and associated with MH, and which is an essential step in neoplasia in chemically induced, experimental hepatocarcinogenesis in rodents (Columbano et al., 1981). MH is also associated with all three major types of foci of cellular alteration (FCA) (clear cell, eosinophilic, and basophilic focus), which are considered to be preneoplastic lesions in rodent hepatocarcinogenesis models, from which true neoplasms may develop under the correct conditions (Farber and Cameron, 1980). These focal lesions, in rodent models, represent an obligatory precursor step in the induction of hepatocellular neoplasms. They generally occur earliest in sole at least one year in age, and are found at higher prevalences in older adults. Based on these strong associations, the age of affected fish, and the established temporal histogenesis of liver neoplasia in rodents, we believe that the regenerative and preneoplastic focal lesions follow the initial hepatotoxic lesion, MH.

Strong, consistent associations also exist among all of the preneoplastic FCA, and between these focal lesions and hepatic neoplasms, especially the liver cell adenoma. These patterns strongly suggest that the focal lesions in English sole are, in fact, preneoplastic lesions. The lack of a strong association between MH and the hepatic neoplasms also suggests that these lesions are sufficiently separated in the temporal sequence of lesion progression that they tend not to co-occur together.

In summary, this analysis strongly suggests that the co-occurring lesion types observed in English sole from polluted areas of Puget Sound comprise a temporal sequence of lesions progressing towards hepatic neoplasms that parallels the same process in experimental mouse and rat hepatocarcinogenesis. These results are the first confirmation, in any wild vertebrate population exposed to hepatotoxins and carcinogens in the environment, of the experimentally derived histogenesis of chemically induced hepatic neoplasia (Myers et al., 1987). This evidence also suggests that xenobiotic chemical hepatotoxins and carcinogens in the environment are the inducers of these lesions in English sole liver. The need to consider hepatic lesion types other than neoplasms as indicators of contaminant exposure when conducting fish liver histopathology monitoring studies is also strongly indicated by this model.

<u>Chemical induction of idiopathic hepatic lesions in English sole</u>

The types of evidence discussed above supporting a xenobiotic chemical etiology for hepatic neoplasms and other associated lesions in English sole, however, are essentially epidemiologic and rely merely on the demonstration of correlations and associations. This type of evidence is not generally interpreted as proof of a direct cause and effect relationship. We have recently addressed this question directly by completing a series of laboratory exposure studies with English sole, in which sole were injected with various extracts from polluted sediments, a model hepatocarcinogen (benzo[a]pyrene), with appropriate controls. The results of this study are reported, with different emphasis, by Schiewe et al. (this volume) and will be fully reported in subsequent manuscripts (Schiewe et al., Myers et al., in preparation). To summarize this work, only injection of an extract from Eagle Harbor and the model hepatocarcinogen, benzo[a]pyrene, induced high incidences of the unique hepatotoxic lesions, NP (up to 83%) and MH (up to 60%), and associated hepatocellular degenerative lesions. The most exciting result of this study was the induction, again only in the exposure groups mentioned above, of basophilic foci. This lesion type is an essentially proven preneoplastic lesion in rodent hepatocarcinogenesis models (Solt et al., 1977; Farber and

Cameron, 1980), and is thought to be the immediate precursor of hepatic neoplasms in these models, as well as in the model for rainbow trout (Hendricks et al., 1984). All of the lesions indicative of hepatotoxicity and the preneoplastic focal lesions induced in this study were morphologically identical to those observed in sole from our field studies in Puget Sound. These lesion types also showed associations, in a lesion co-occurrence analysis, that paralleled our results from field studies in Eagle Harbor, providing a more firm experimental basis for our proposed scheme of liver neoplasm histogenesis in feral English sole (Myers et al., 1987). In summary, the results from this laboratory exposure study directly impugn certain chemical contaminants in the sediments as the agents of hepatocellular NP/MH, regeneration, preneoplastic FCA, and, by association, hepatic neoplasms in English sole from Puget Sound. Further emphasized is the prime importance of monitoring indicator lesions other than hepatic neoplasms when conducting liver histopathology field studies on the potential effects of xenobiotic chemicals on fish.

Summary and Conclusions

Research over the past decade in Puget Sound utilizing the histopathology of feral English sole liver as a biological indicator of chemical contaminant exposure, in conjunction with other chemical indicators of contaminant exposure and uptake, has produced a reliable, predictive model of chemical hepatocarcinogenesis in an apparently sensitive species. Although hepatic neoplasms have received the most attention as an indicator lesion in fish, our research and the research of others in this field clearly show that other, often more prevalent and earlier appearing lesions in feral fish are also good indicators of the biological effects of pollutant exposure. This is especially true when these lesion types are considered within the context of the stepwise histogenesis of hepatic neoplasia in vertebrates. We have found these non-neoplastic lesions in demersal species of fish other than English sole from Puget Sound (Myers and Rhodes, 1988) and elsewhere on the

North American west coast (Malins et al., 1987b; McCain et al., 1988; Myers et al., in preparation). For example, in Puget Sound, the pattern of geographic distribution of fish, such as rock sole and starry flounder, affected by these lesions closely parallels the same pattern for English sole (Myers and Rhodes, 1988), with the exception that these species show relatively low prevalences of frank hepatic neoplasms. By further understanding the process of hepatic neoplasia in English sole, it is hoped that this model can be extrapolated to other species of marine fish, thus providing reliable early biological indicators of chemical contaminant exposure in the marine environment.

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Table 1. Calculated odds ratios of significant ($p \leq 0.05$) risk factors for four categories of idiopathic hepatic lesions. Odds ratios for the site of collection are interpreted relative to the reference site of Port Madison. Odds ratios for age represent the effect of each additional year of age on the odds of lesion occurrence.

Lesion	Risk factor	Odds Ratio
Neoplasms	Upper Duwamish Waterway	8.7
	Lower Duwamish Waterway	8.2
	Age	1.6
Foci of cellular Alteration	Upper Duwamish Waterway	4.2
	Lower Duwamish Waterway	2.4
	Age	1.4
Nuclear pleomorph./ Megalo. hepatosis	Upper Duwamish Waterway	4.9
	Lower Duwamish Waterway	3.4
	North Seattle Waterfront	4.0
	South Seattle Waterfront	2.7
Steatosis/ Hemosiderosis	Upper Duwamish Waterway	2.9
	Age	1.4
	Winter season	0.001

Adapted from: L.D.Rhodes et al., J Fish Biol 1987; 31:395-407.

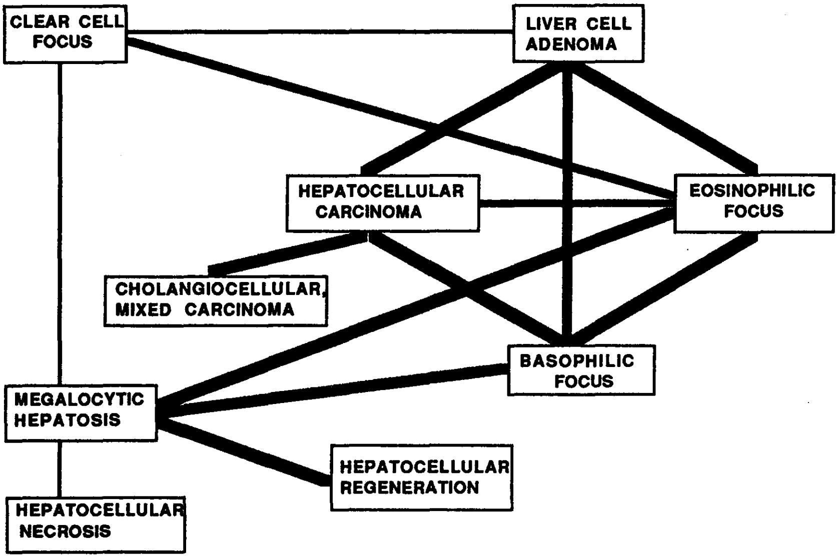
Table 2. Important Idiopathic Hepatic Lesions in English Sole

Lesion type/category

1) hepatocellular necrosis/degeneration
   (nonspecific, degenerative)
2) nuclear pleomorphism
   (unique,degenerative)
3) megalocytic hepatosis
   (unique,degenerative)
4) hepatocellular regeneration
   (proliferative,non-neoplastic)
5) clear cell focus of cellular alteration
   (preneoplastic)
6) eosinophilic focus " "
   (preneoplastic)
7) basophilic focus " "
   (preneoplastic)
8) liver cell adenoma
   (benign neoplasm)
9) hepatocellular carcinoma
   (malignant neoplasm)
10) cholangioma
   (benign neoplasm)
11) cholangiocellular carcinoma
   (malignant neoplasm)
12) mixed hepatobiliary carcinoma
   (malignant neoplasm)
13) cholangiofibrosis
   (proliferative, non-neoplastic)
14) hemosiderosis
   (storage disorder)
15) steatosis/fatty change
   (storage disorder)

Figure 1.

Histogenetic Relationships (Associations Determined by 3-way Contingency Analysis) Among Idiopathic Lesions Involved in Hepatic Neoplasia in Feral English sole



Lines between lesion types indicate significant associations between those lesions, in terms of their frequency of co-occurrence. Thickness of lines shows the relative consistency of that association.
Diagram adapted from Myers et al., J. Natl. Cancer Inst. 1987;78:333-363

Epidemiology of Diseases in Fish Populations, Especially of the English Sole Liver Tumor in Puget Sound

Vincent F.Gallucci and John R. Skalski

The objective of this presentation is to introduce into the environmental literature some of the concepts and methods employed by epidemiologists and demographers concerned with disease and its dispersal in human populations. It has become clear that to achieve the objectives sought by epidemiologists, significant modifications in the methods of analysis used in ecological studies are needed. We have coined the expression "ecological epidemiology" to describe the methods and the application.

In order to make these principles as specific as possible we describe a potential application to the epidemiology of the tumor that occurs in the liver of English sole in Puget Sound. The most obvious difference between working with a population of fish and with a human population is that the interview with a human subject to collect life history data is considerably easier for the patient and the interviewer. Nevertheless, certain data must be collected. Information about age, growth, and sex, including fertility or fecundity, is important. In addition, where one was born, places one has live, and thus, the types of food consumed, are also important.

Presuming that these data are available, a population dynamicist/epidemiologist would proceed to try to identify and quantify a relationship between the diseases of interest and the demographic data. Although the ecological literature is rich in various methods that estimate mortality rates or survival rates (e.g. the Robson-Chapman technique), relatively little effort has gone into the estimation of survival functions as a consequence of a particular health hazard. In contrast, epidemiologists have constructed a body of methodology based on classical responses of populations to disease and have effectively used this methodology for the statistical evaluation of cause and effect in disease-related investigations.

For example, a survival function S(t) is defined mathematically as:

a) $S(t) = P(T > t)$ = probability an individual lives longer than time t where T is a random variable

b) $\hat{S}$ = number surviving longer than t/total number of individuals

A hazard function or age specific failure rate or, as it is known, a force of mortality, is:

c) $h(t) = \lim_{\Delta t \to 0} P(\text{an individual of age t dies in } (t, t+\Delta t))/\Delta t$

d) $\hat{h}$ = number dying in a Δt starting at t/(number surviving at t)(Δt)

And associated with these is a probability density function f(t):

e) $f(t) = \lim_{\Delta t \to 0} P(\text{individual dying in } (t, t+\Delta t)/(\text{total\#})(\Delta t)$

These three functions S(t), h(t), f(t) are most frequently associated with a particular disease and a population's response to the disease. These functions are all conceptually related to each other and are expressible as functions of each other.

Hazard functions are usually plotted versus time or age and can take several classical forms. For example, exponentially increasing might refer to acute leukemia if initial treatment fails; exponentially decreasing if an operation is performed on bullet wounds, since the principal risk is the operation, after which survival is quite probable; a constant h(t) is risk of death for individuals aged 18-40 who are healthy, and a bathtub or bowl function describes life in general: high infant mortality rates and after age 40 rates, but relatively constant between ages 4 and 40, and so on.

Epidemiologists use the mathematical formalism to great advantage in defining both parametric and non-parametric hazard functions and a variety of methods of parameter estimation. An example of the flexibility that even parametric models offer is seen in the Weibull.

distribution, a distribution rarely seen in the ecological literature but common in epidemiology. It is a very flexible distribution defined with two parameters, one of these, λ is the scale parameter and the other, γ is called the shape parameter.

In a general way the methodology of epidemiology is partitionable into several parts:

a) Identification of the prospects of survival due to some treatment involving survival curves and hazard functions. Survival functions are conveniently classified as non-parametric and parametric. Examples of non-parametric models are life tables and curve fitting regressions where parametric models would be mathematically linked to specific probability models.

b) Comparison of survival functions to compare treatment vs non-treatment cases. These methods of comparison are also of the parametric and non-parametric types.

c) Identification of the risk variables.

d) Design of experiments to obtain the appropriate data for analysis.

To make distinctions between hazards is then to construct a theory of competing risks. Now if the probability of mortality in the Δt after time T is specified with respect to CVR (cardiovascular-renal disease), cancer, TB, etc. and, if all causes act independently, then the average "time to death" could be computed. Intuition, however, tells us that successive diseases weaken resistance making one more susceptible to other life-threatening hazards. Therefore, it is not quite proper to treat several disease processes and the risk of death from any one disease as independent from the others. To cope with this realistic situation, the theory of competing risks allows the assignment of probabilities of death to each of the risks.

The transition to fish is now straight forward. Proceeding by analogy, a working hypothesis is that the tumors that occur in the liver of English sole are of ecological origin. To understand the etiology of the hepatic neoplasms is one problem in itself because the first step may not be pollutants in the water, in the sediments, or in the prey they fed on. Another problem, however, is what is the cause of mortality, is it the neoplasms or do neoplasms of a certain density or type make the fish more susceptible to risks of death.

The epidemiology of this tumor should be viewed in the context described above. The likelihood of survival to age T for English sole in"polluted" bays and in"clean" bays need first be determined. Few conclusions would be drawn, however, because the life history of fish in any bay will require that they are occasionally outside the bay. Therefore, it is important to know what competing risks are present to survival for an old fish about to go to fish heaven. Such prospective sources of mortality competing for a victim such as predators, parasites, harvests, etc. need to be explored to determine their roles acting independently and concurrently.

Multidimensional contingency tables can be used to relate the counts of individuals with exposure to specific ecological hazards and from that, multivariate probability statements can be made. For example, if "+" means "yes" and "-" means "no", contingency tables for exposure to risks 1 and 2 lead to probability statements such as the prevalence odds ratio (POR) and the relative risk (RR). Relative risk describes the ratio of rates of newly diseased fish arising in exposed and unexposed populations. The odds ration is particularly suited to cases where one characteristic of a situation is antecedent to another. For example, the exposure to a pollutant is statistically related to the risk of experiencing the outcome of concern, a liver tumor (B), by the use of the odds ration. The odds are expressible in terms of conditional probability, P(B/A) and thus in terms of contingency tables.

At this stage of the analysis, a quantitative link between causative factors and the probability of occurrence of disease will be drawn and the linkage between the risk of increased disease rates and changes in the frequency of occurrence of apparently associated pathological conditions. Further, it is now possible to associate dollar costs with the removal or cleanup of some aspect of society's impact on the bay which appears to be related to the occurrence of a specific disease. Then it becomes a societal question.

The epidemiology of diseases in fish is at an early stage of development. One of the principal ways in which this new field will be advanced is by the development of improved life history information.

CONTAMINANT EFFECTS ON OVARIAN MATURATION IN ENGLISH SOLE (*Parophrys vetulus*) FROM PUGET SOUND, WASHINGTON[1]

Lyndal Johnson, Edmundo Casillas, Tracy Collier, Bruce McCain, and Usha Varanasi*

Introduction

The ability to reproduce is critical to the survival of any species. However, it has become increasingly clear that exposure to xenobiotic chemicals in the environment may alter reproductive processes in a variety of organisms, including teleost fish. In several areas of Puget Sound, the sediments are contaminated with toxicants, including aromatic hydrocarbons (AHs), polychlorinated biphenyls (PCBs), and metals (Malins *et. al.* 1984). Moreover, recent studies have demonstrated that exposure to these compounds may have a detrimental effect on fish health. A high proportion of bottomfish residing in contaminated areas of Puget Sound, for example, have pollution associated liver diseases, including liver neoplasms (Malins *et. al.* 1984,1985; Myers *et. al* 1988). It seems likely that contaminant exposure could be causing other problems as well, including reproductive impairment. Unfortunately, little is known about the impact of environmental contaminants on the reproductive processes in natural fish populations in Puget Sound.

To address this problem, in 1985 the EC Division began a research program to examine possible effects of environmental exposure to contaminants on reproduction in Puget Sound flatfish. English sole was chosen as the primary experimental animal for this research project because previous studies had shown that this species was particularly sensitive to contaminants, and because it

[1]Sections of this paper have been excerpted from a more detailed manuscript, Johnson, L.L., E. Casillas, T.K. Collier, B.B. McCain, and U. Varanasi. 1988. Contaminant Effects on Ovarian Maturation in English Sole (*Parophrys vetulus*) from Puget Sound, WA, in preparation.

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could be found in a wide range of areas in Puget Sound, including both urban and non-urban sites. In addition, hepatic lesions are particularly common in English sole, so by working with this species, we have the opportunity to examine the impact of liver damage on reproduction. Several aspects of the reproductive process are being investigated in our reseach program, including gonadal development, steroid hormone production and metabolism, spawning ability, fertilization success, and larval viability. We have recently completed a two year field study concerning contaminant effects on ovarian maturation in English sole. The results are summarized in this paper.

Ovarian development in female English sole typically begins in the fall. One of the major events in this process is the development of yolked eggs, or vitellogenesis. Like many other aspects of the reproduction, vitellogenesis is hormonally controlled.. The events involved are summarized in Figure 1. First, gonadotropin releasing hormone is secreted by the brain; this stimulates the pituitary to produce gonadotropin. The gonadotropin then acts on the ovarian follicle cells, causing them to secrete estradiol. The estradiol, in turn, stimulates the liver to produce yolk protein, or vitellogenin, which is released into the bloodstream and is then incorporated into the developing oocytes. Estradiol is later broken down by hepatic enzymes, some of which are similar to the enzymes which metabolize xenobiotic compounds. Once vitellogenesis is intiated, oocytes continue to absorb yolk and increase in size for several weeks, until migration to the spawning ground occurs. This generally takes place in

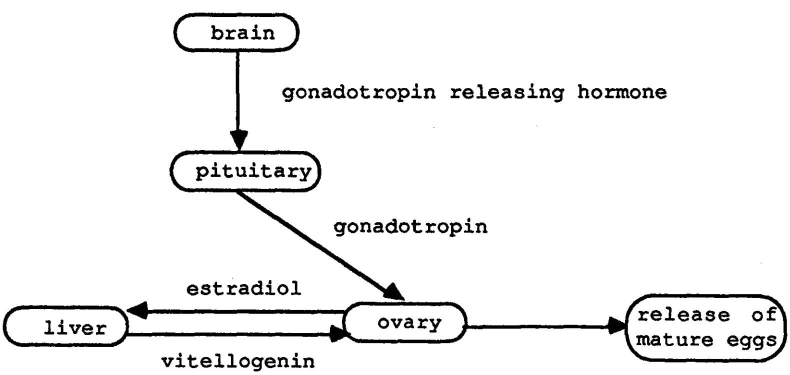

Fig. 1. Endocrine control of ovarian maturation process in teleost fish

late December or early January. After the animals have reached the spawning areas, the oocytes ,undergo hydration and ovulation, and are released from the body cavity. Contaminant exposure could disrupt the ovarian maturation cycle by altering or interfering with any one of the processes described above. Our goal in this study was to determine whether such effects on ovarian development could be found in English sole exposed to environmental contaminants.

Methods

Adult female English sole were collected from four sites in Puget Sound (Port Susan, Sinclair Inlet, Eagle Harbor, and the Duwamish Waterway), just prior to the winter spawning seasons of 1986 and 1987, when ovarian maturation was at its peak, but before migration to the spawning areas had occured. These sites were chosn because previous sampling had shown that they had a wide range of levels of contaminants in the sediment (Malins, *et al.* 1984,1985). Eagle Harbor has high levels of aromatic hydrocarbons (AHs) in the sediment, Duwamish Waterway has high levels of both AHs and polychlorinated biphenyls (PCBs). Sinclair Inlet has low levels of AHs and moderate levels of PCBs, while Port Susan is minimally contaminated

To assess the contaminant exposure level in individual fish, concentrations of fluorescent aromatic compounds (FACs) in the bile (Krahn *et al.*, 1984) and PCBs in the liver were determined. In addition, hepatic aryl hydrocarbon hydroylase (AHH) activity was measured. AHH is a mixed function oxidase enzyme which metabolizes aromatic hydrocarbons. The activity of this enzyme increases with exposure to a variety of xenobiotic compounds, making it a useful indicator of contaminant exposure marine species (Payne *et al.* 1977, Collier *et. al.*, 1988). Also, livers were examined histologically for the presence of contaminant associated lesions including as neoplasms, foci of cellular alteration, megalocytic hepatosis, and storage disorders (Myers *et al.*, 1988).

Reproductive activiy was assessed by examining the ovaries of the sampled English sole histologically to determine their developmental stage, and for the presence of ovarian lesicns that would be indicative of oocyte resorption Other parameters associated with reproductive activity were also measured, including plasma vitellogenin and estradiol levels, and gonadosomatic index (ovary wt/gutted body wt x 100). Relationships between ovarian maturation, plasma

Table 1. Mean levels of indicators of contaminant exposure in female English sole by site of capture. Bile for determination of fluorescent metabolites of aromatic compounds (FAC) were collected in 1986 and 1987, and AHH and hepatic PCB data were collected in 1987 For bile FACs, hepatic AHH activity, and hepatic PCB concentrations, significant differences between the means were determined using analysis of variance and Fishers least significant difference multiple range test; for prevalences of hepatic lesions, the G-statistic was used. Significant levels for both tests were set at p=0.05.

	FACs		hepatic		percentage of
	BaP wavelength (ng/g)	NPH wavelength (ng/g)	AHH activity (pmol/mg/min)	hepatic PCBs (ug/g wet wt)	animals with 1 or more hepatic lesions[1]
Port Susan	36+ 41[c,d] (n=25)	15208 + 34808[b,c,d] (n=25)	221 + 214[b,d] (n=25)	0.52 + 0.95[d] (n=10)	10.00[c,d] (n=50
Sinclair	155 + 364[d] (n=45)	56313 + 88844[a,c,d] (n=45)	72 + 74[a,c,d] (n=24)	3.21 + 1.86[d] (n=10)	6.06[c,d] (n=99)
Eagle Harbor	546 + 407[a,b,d] (n=47)	97074 + 64164[a,b] (n=47)	313 + 242[b,d] (n=25)	0.61 + 0.22[d] (n=10)	67.74[a,b] (n=93)
Duwamish	218 + 206[a,c] (n=47)	87052 + 52976[a,b] (n=48)	492 + 275[a,b,c] (n=25)	14.2 + 10.6[a,b,c] (n=22)	77.48[a,b] (n=111)

[1]Contaminant-associated lesions include hepatic neoplasms, foci of cellular alteration, specific degnerative necrosis, or storage disorders

[a]significantly different from Port Susan

[b]significantly different from Sinclair Inlet

[c]significantly different from Eagle Harbor

[d]significantly different from Duwamish Waterway

estradiol, plasma vitellogenin, and contaminant exposure were then evaluated.

Results and Discussion

Levels of contaminant exposure in English sole sampled from Port Susan, Sinclair Inlet, Eagle Harbor, and the Duwamish Waterway are shown in Table 1. Generally, contaminant exposure was greatest in sole from the Duwamish Waterway and Eagle Harbor. Animals from both of these sites had high levels of FACs in the bile and elevated hepatic AHH activity, and were more likely than sole from the other two sites to have contaminant-associated hepatic lesions. English sole from the Duwamish Waterway also had high concentrations of PCBs in the liver, compared to English sole from the other sites.

Several abnormalities were observed when ovarian development was monitored in English sole from the four sampling sites. The most striking finding concerned the apparent effect of contaminant exposure on ovarian

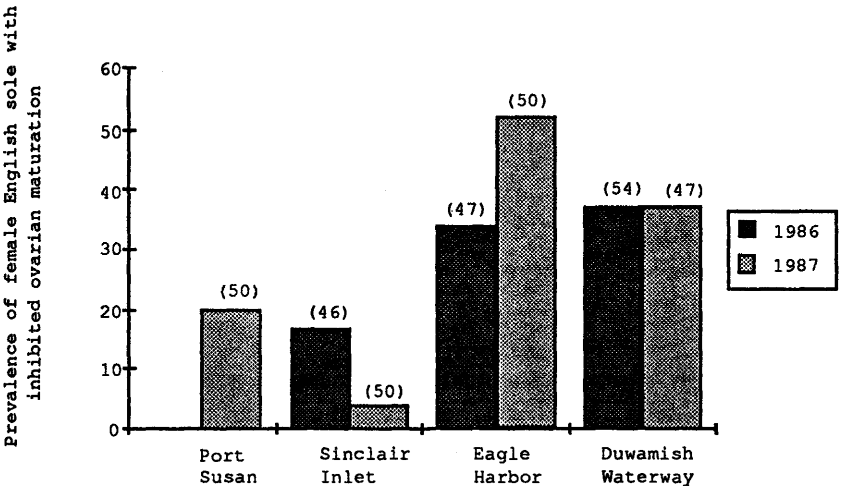

Figure 2. Prevalences of adult female English sole with inhibited ovarian development, as determined by histological examination, from four sites in Puget Sound. Numbers in parentheses indicate the number of animals sampled. For both 1986 and 1987, prevalences of female English sole with inhibited ovarian development were significantly higher (G-statistic, $p = 0.05$) at the most heavily contaminated sites, Eagle Harbor and the Duwamish Waterway, than at Port Susan or Sinclair Inlet.

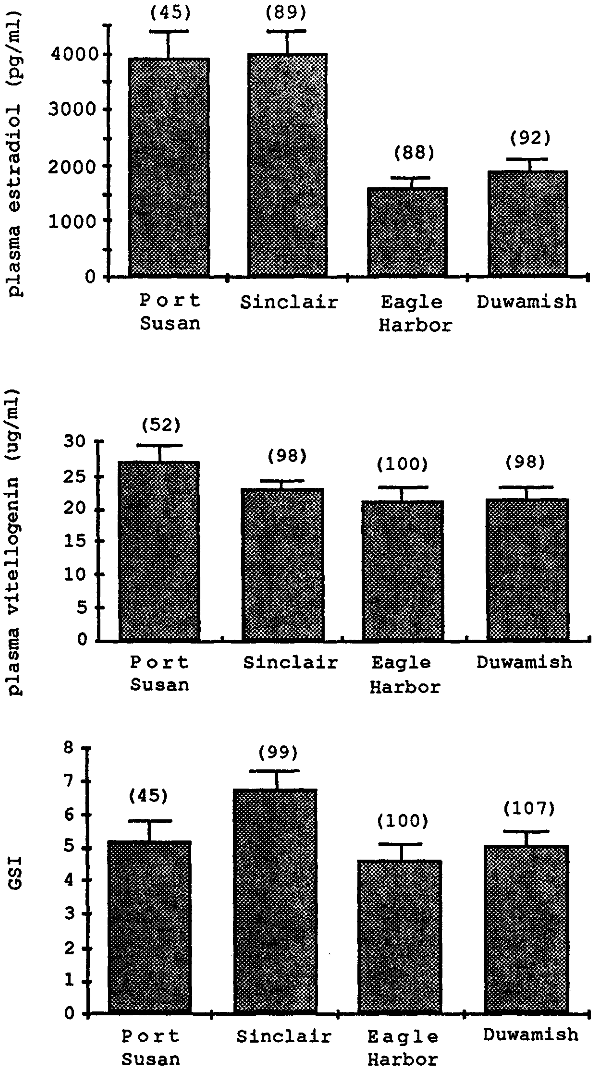

Figure 3. Mean levels of plasma estradiol, plasma vitellogenin, and GSI in female English sole from four sites in Puget Sound. Plasma estradiol levels in sole from Port Susan and Sinclair Inlet are significantly higher than in sole from Eagle Harbor or the Duwamish Waterway; GSI is significantly higher in sole from Sinclair Inlet than in sole from Eagle Harbor or the Duwamish Waterway. (ANOVA, $p < 0.05$).

maturation. In both 1986 and 1987, it was found that the prevalence of female English sole failing to undergo ovarian maturation (i.e. to enter vitellogenesis) was significantly higher ($p \leq 0.05$, G-statistic) at the two heavily contaminated sites, Eagle Harbor and the Duwamish Waterway, than at Port Susan or Sinclair Inlet.(Fig. 2). Female English sole from Eagle Harbor and the Duwamish Waterway also had lower levels of plasma estradiol than female English sole fom Port Susan or Sinclair Inlet, and reduced GSI compared to female English sole from Sinclair Inlet (Fig. 3). No significant intersite differences were found in plasma vitellogenin levels, however (Fig. 3). Intersite differences in GSI could be accounted for by the lower proportion of females undergoing gonadal maturation in the Duwamish Waterway and at Eagle Harbor. In the case of plasma estradiol, however, even in vitellogenic females, levels were lower in English sole from the Duwamish Waterway and Eagle Harbor than in English sole from Port Susan or Sinclair Inlet, suggesting that in

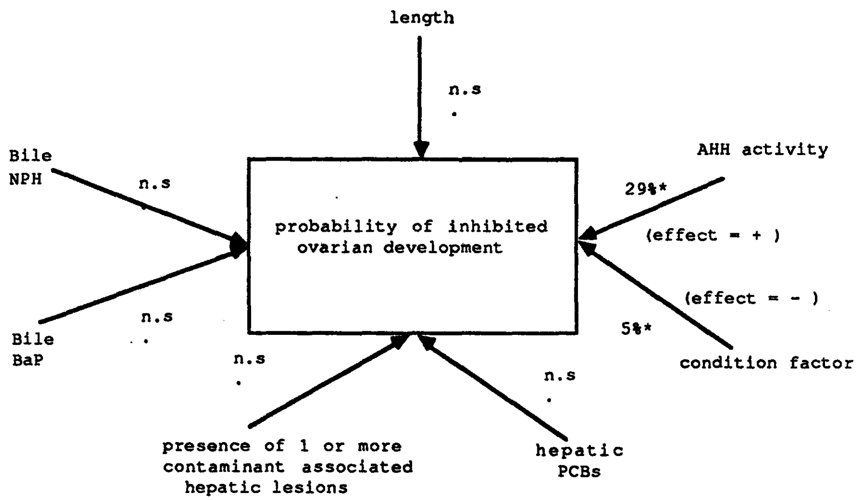

Figure 4. Predictive model relating indicators of contaminant exposure to the probability of inhibited ovarian development.in English sole from Puget Sound. The model (AHH activity - condition factor) accounts for 34% of the observed variation in the occurrence of inhibited ovarian development. (% = % of variation explained; n.s. = effect not significant at $p \leq 0.05$; * = effect significant at $p \leq 0.05$; ** = effect significant at $p \leq 0.001$).

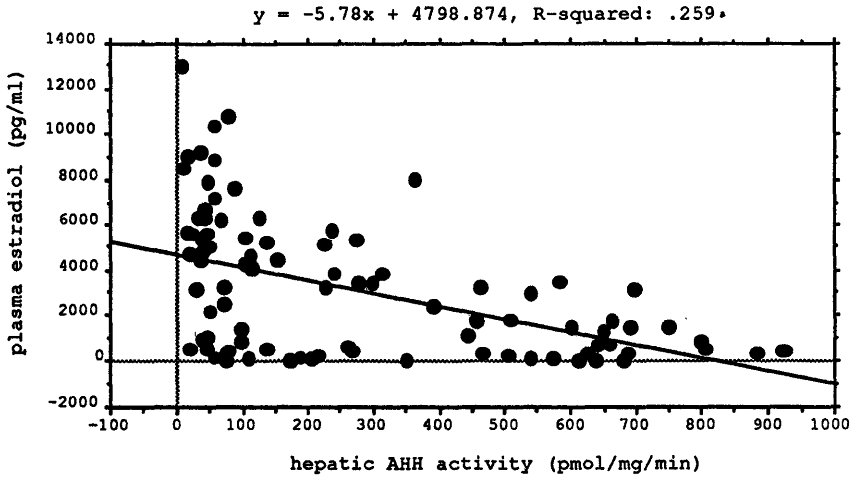

Figure 5. Relationship between hepatic AHH activity and plasma estradiol level in female English sole from Puget Sound.

sole from contaminated sites, there is either decreased production or increased breakdown of estradiol.

Of the various indicators of contaminant exposure which we measured (i.e., FACs in bile, hepatic AHH activity, hepatic PCB concentrations, and contaminant associated hepatic lesions), we found that hepatic AHH activity was the best predictor of whether or not English sole would undergo normal ovarian maturation (Fig. 4). Other investigators have also found reproductive abnormalities in fish with increased activity of AHH and other xenobiotic metabolizing enzymes, including reduced fertilization success (Spies *et al.*, 1985) and depressed levels of plasma steroid hormone levels (Sivarajah, *et al.*, 1978).

The exact role that AHH and other hepatic xenobiotic metabolizing enzymes may play in generating reproductive abnormalities is not clear. Several classes of these enzymes appear to be involved in the metabolism of estradiol (Hansson *et al.*, 1980; Forlin and Hansson, 1982, Snowberger and Stegman, 1986). A contaminant induced increase in the activity of any of these enzymes could increase the rate of estradiol breakdown, leading to reduced levels of estradiol in the plasma, and perhaps impaired ovarian maturation. In our study we found that hepatic AHH activity was closely related to plasma estradiol level; female English sole with elevated AHH activity had depressed levels of plasma estradiol (Fig. 5). However, other steroid metabolizing enzymes that are affected by contaminant exposure may also be having an impact on estradiol metabolism in English sole. Studies investigating the effects of contaminant exposure on the activity of steroid metabolizing enzymes in this species, and consequent effects on plasma estradiol level and ovarian maturation, are currently in progress in our laboratories.

Hepatic lesions did not appear to have a direct impact on ovarian maturation. No significant differences were found in plasma vitellogenin levels in English sole sampled from the contaminated and reference sites, in spite of clear differences in lesions prevalences. Moreover, when the effects of other indicators of contaminant exposure were taken into account, the presence of hepatic lesions did not result in a significantly incrased risk of inhibited ovarian development (Fig. 4), as one would expect if the lesions had a direct effect on vitellogenesis. Apparently, the liver damage associated with these lesions was not extensive enough to impair hepatic vitellogen productions.

Of the two types of contaminants (AHs and PCBs) whose levels we measured, AHs appeared to be the most closely related to inhibited ovarian development. At the two sites with the highest proportion of females failing to mature, Eagle Harbor and the Duwamish Waterway, average levels of FACs in the bile were greater than 500 ng/g for metabolites fluorescing at BaP wavelengths, and greater than 80000 ng/g for metabolites fluorescing at naphthelene wavelengths. Hepatic PCB concentrations did not show any clear relationship with inhibited ovarian maturation; the prevalence of English sole with this problem was highest at Eagle Harbor, a site with high AH levels but no appreciable PCB contamination.

Although the specific mechanisms responsible for inhibited ovarian development and depressed estradiol production in English sole are not clear, the present findings suggest that exposure to environmental contaminants may interfere with the ability of female English sole residing in localized contaminated sites in Puget Sound to complete the normal reproductive cycle. At this point, the impact of this impairment on recruitment and population growth in English sole is not known. However, our findings suggest that contaminant effects on reproduction should be taken into account in stock assessment of marine fish.

Acknowledgements

We thank Bob Bunnell and Pat Manning for processing histological samples; Mark Myers, Paul Olson, and Linda Rhodes for their assistance in microscopic examination of the liver and ovary tissues; Peggy Krahn and Leslie Moore for performing HPLC analyses of bile; Walt Dickoff, Penny Swanson, and Melinda Bernard for performing plasma estradiol assays; Ethel Blood for perfoming plasma vitellogenin assays; and Bill Gronlund, Dick Hughes, and Doug Weber for their assistance in capturing fish. These studies were partially supported by the Office of Oceanography and Marine Assessment, National Ocean Services, NOAA.

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Summary Statement

Bruce B. McCain * and Marsha L. Landolt**

This session, titled "Toxic Chemicals and Their Effects: What Are We Finding?", contained eight presentations which described the results of studies designed to identify, characterize, or predict pollution-related problems in the Puget Sound Basin. These papers can be grouped into three categories with regard to problems in the Basin: (1) those that described an approach which did not demonstrate a current problem; (2) those that identified potential problems whose biological significance or relationships to pollution are unclear; and (3) those that characterized the mechanisms and impacts of recognized pollution-associated problems.

The papers in the first category included those by Calambokidis *et al.* and Solomon *et al.* Calambokidis *et al.* examined relationships between the reproductive capacity and population size of Puget Sound harbor seals (Phoca vitulina) and concentrations of PCBs and DDE in harbor seal pups. Although they hypothesized that diminished population sizes of harbor seals in the early 1970s may have been related to increased tissue concentrations of PCBs, population sizes have increased in recent years and tissue levels of PCBs have declined. Solomon *et al.* investigated levels of PCBs in edible tissues of crayfish (Pacifasticus leniasculus) and various finfish from Lake Union, Seattle, and found the levels to be "well below the U.S. FDA tolerance level".

The papers by Speich *et al.*, Lee *et al.*, and McCain *et al.* fell into the second category. Speich *et al.* examined eggshell thickness and measured PCB and DDT residues in glaucous-winged gull (Larus glaucescens) eggs at sites near Puget Sound. They found eggshell thickness values to be lower and concentrations of PCBs to be higher in eggs from Seattle/Tacoma, compared to eggs from a site north of Puget Sound; however, no clear relationship between these two parameters was found. In studies with chinook salmon (Oncorhynchus tshawytscha), Lee *et al.* found that this species readily bioaccumulates waterborne toluene, and McCain *et al.* found that juvenile downstream migrants in the Duwamish Waterway, Seattle, bioaccumulated high levels of aromatic hydrocarbons and PCBs compared to juveniles from a reference estuary.

Of those papers addressing commonly accepted pollution-associated problems in Puget Sound, Myers *et al.* presented data on the stepwise progression of lesion types leading to liver neoplasms in English sole (Parophrys vetulus). Gallucci and Skalski discussed the principles of "ecological epidemiology" which may be used to investigate changes in English sole populations due to the possible harmful effects of liver neoplasia. Johnson *et al.* examined the question of whether English sole in polluted waterways may have impaired reproductive processes. They reported that mature English sole females from the Duwamish Waterway and Eagle Harbor, Bainbridge Island, had inhibited ovarian development compared to females from reference sites.

Although many of the papers in this session showed that pollution-related effects exist, or may exist, in certain urban bays, they also highlight the challenges facing researchers in Puget Sound.

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CONTAMINATED SEDIMENTS

HOW ARE THEY IDENTIFIED AND WHAT CAN WE DO WITH THEM?

Keith Phillips,
Washington Department of Ecology
Session Chair

Introductory Statement

Keith E. Phillips *

Over the last ten years, the focus of aquatic pollution management activities has shifted from water quality and the water column to sediment quality and the benthic environment. In hindsight, there is no great surprise in this shift. Many of the chemicals we discharge and manage in the aquatic environment are not all that soluble, and would rather be bound to particles than remain in the dissolved state. We have learned over time that sediments can act as a persistent repository of chemical contaminants. Contaminated sediments have been associated with toxic responses in laboratory bioassays, with degraded benthic communities and even with diseases found in bottom fish. The risk to human health from consuming animals found in these areas is also being debated.

Studies conducted by federal agencies such as NOAA and EPA first identified the presence of contaminated sediments in the Sound. The Commencement Bay nearshore and tideflats area was one of the first marine sites designated for remedial action via the federal Superfund program. Similar action has been undertaken by Superfund and the Puget Sound Estuary Program in Eagle Harbor, Elliott Bay, Everett Harbor, Budd Inlet and Sinclair Inlet. This attention contributed to the initiation of the Puget Sound Dredged Disposal Analysis, where the relationship between dredging and contaminated sediments was addressed. As outlined in the Authority's 1987 Puget Sound Water Quality Management Plan, the State of Washington is moving to establish sediment quality standards for the State by regulation. Confined disposal standards and sites, and guidelines for remedial action, for contaminated sediments are also being addressed.

The speakers today will provide select insights into current contaminated sediments issues and activities -- beginning with the identification of what is contaminated and what is not. The first two papers address the development of sediment quality values or criteria to identify contaminated sediments, and ultimately to facilitate management decisions.

Once you've identified a contaminated sediment, the last three speakers each take a look at different responses. First, what would happen to the sediments if we turn off the sources of contamination and just let them sit and recover naturally? How long will it take? Second, if a sediment to be dredged is too contaminated for unconfined, open-water disposal, a confined disposal method is required. What happens to the contamination in sediments placed behind a dike along the shoreline? Do the chemicals leach over time? And last, what in-place methods can we use to isolate or treat contaminated sediments? How do we effectively cap the contamination?

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USE OF SEDIMENT QUALITY VALUES TO ASSESS SEDIMENT CONTAMINATION AND POTENTIAL REMEDIAL ACTIONS IN PUGET SOUND

Robert Barrick, Robert Pastorok, Harry Beller, and Thomas Ginn
PTI Environmental Services

INTRODUCTION

Sediments are a primary reservoir of contaminants released by industrial, commercial, and residential activities in Puget Sound. The management of contaminated sediments requires that biological or chemical criteria be developed to distinguish effects or concentration levels above which the sediments are considered to be a problem. Regulatory sediment criteria for defining "clean sediments" have not yet been adopted, nor is it likely that a single biological test or single concentration for a chemical can always define "acceptable contamination". To meet the needs of ongoing sediment management programs, an ideal approach for sediment criteria would perfom well on both of the following two tests of reliability based on actual field data on the occurrence of biological effects:

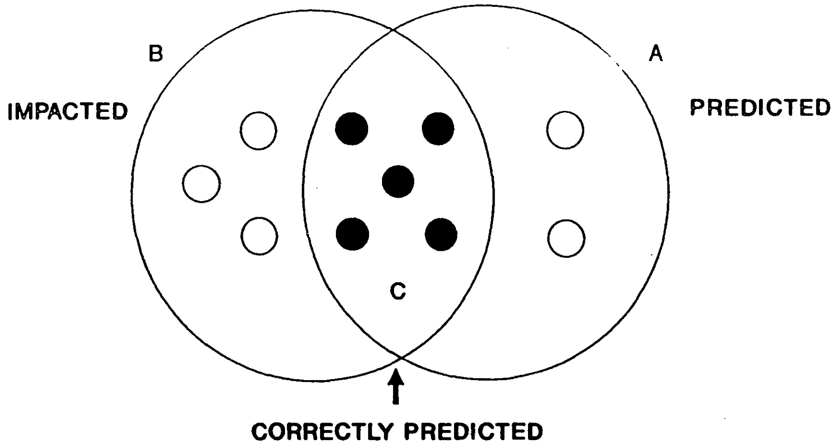
- Sensitive detection of environmental problems (i.e., are <u>all</u> sediments exhibiting biological effects identified using the predictive approach?)

- Efficient screening of environmental problems (i.e., are <u>only</u> sediments exhibiting biological effects identified using the predictive approach?).

The concepts of sensitivity and efficiency are illustrated in Figure 1. Sediment quality values that are highly sensitive may be environmentally protective but are not necessarily cost-effective. Sediment quality values that are highly efficient may be cost-effective and defensible in pursuing high priority remedial action but are not necessarily protective. The application of sediment quality values in Puget Sound, including how these tradeoffs are addressed, is described in this paper using examples from the Commencement Bay Superfund Project, the Puget Sound Dredged Disposal Analysis (PSDDA), and the Puget Sound Estuary Program (PSEP). This work has been supported by several contracts from the Washington Department of Ecology/U.S. EPA, PSDDA (coordinated by U.S. Corps of Engineers, Seattle District), and the PSEP (managed by U.S. EPA, Region 10). Many of the evaluations described in this paper were conducted over the past 4 years at PTI Environmental Services and Tetra Tech, Inc.

METHODS

The assessment of toxic effects associated with contaminated sediments has been approached by environmental scientists in two general ways (Figure 2). The first general approach is based on empirical relationships



SENSITIVITY = C/B x 100 = 5/8 x 100 = 63%

EFFICIENCY = C/A x 100 = 5/7 x 100 = 71%

FOR A GIVEN BIOLOGICAL INDICATOR:

A ALL STATIONS PREDICTED TO BE IMPACTED

B ALL STATIONS KNOWN TO BE IMPACTED

C ALL STATIONS CORRECTLY PREDICTED TO BE IMPACTED

Figure 1. Measures of reliability.

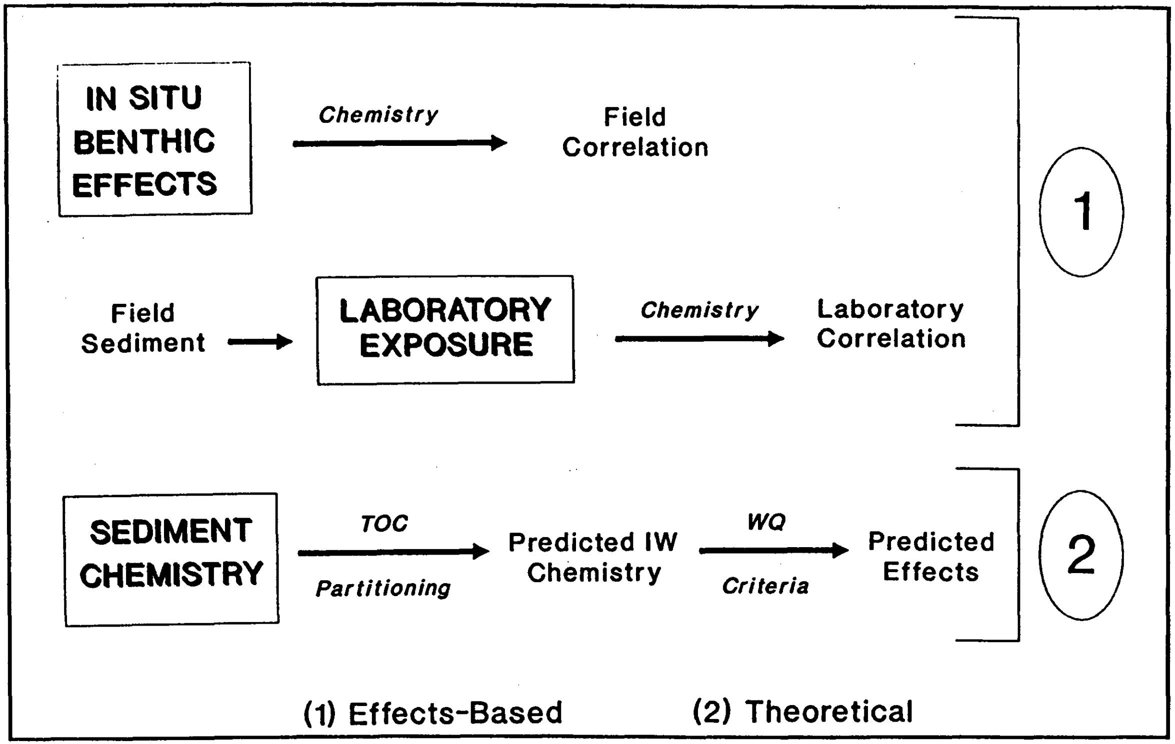


Figure 2. Assessment of sediment quality.

between laboratory sediment bioassays, *in situ* biological effects observed in organisms associated with sediments, and chemical concentrations measured in sediments. Examples of this "effects-based" approach include the Sediment Quality Triad (Long and Chapman 1985), the Apparent Effects Threshold (AET) (Barrick et al. 1985), and the Screening Level Concentration (SLC) (Battelle 1986). The second approach emphasizes theoretical models to predict the partitioning of sediment contaminants to interstitial water (a major exposure pathway for organisms associated with sediments). The predicted interstitial water concentrations are then compared to water quality criteria based on laboratory measurements of biological effects (e.g., the Equilibrium Partitioning approach) (Kadeg et al. 1986). None of the available approaches is fully capable of addressing all concerns over interactive effects among chemicals; hence, field verification using diverse environmental samples is critical to the evaluation of each approach.

EPA is working at the national level to develop sediment criteria that can be used similarly to existing water quality criteria. These efforts have focused on the theoretical approach shown in Figure 2 and are not yet completed. Of the effects-based approaches, the AET technique has been applied and tested in multiple areas of Puget Sound. The approach assumes a dose-response relationship between increasing chemical contamination and biological effects (Barrick et al. in prep). Specifically, the AET is the chemical concentration in sediments above which statistically significant biological effects are always expected for one or more biological effects indicator. AET were originally developed to identify problem sediments in the Commencement Bay Nearshore/Tideflats Remedial Investigation (Barrick et al. 1985). The reliability of AET were subsequently tested using biological and chemical data from a 190-sample database compiled from multiple Puget Sound studies (Beller et al. 1986). Additional tests incorporating recent data from PSEP investigations in Elliott Bay and Everett Harbor are in progress.

RESULTS

Although each of the approaches evaluated have advantages and disadvantages, the AET approach was selected as the currently preferred approach for assessing sediment quality in Puget Sound (Beller et al. 1986). Based on a comparison of three recommended approaches using available biological effects data, the AET approach had the highest sensitivity for detection of a range of biological problems, and was as efficient as the alternative approaches tested.

AET were developed for each chemical of concern for multiple biological indicators because different kinds of biological indicators respond in different ways to the same chemical exposure. For example, an assessment of acute lethal toxicity to contaminated sediments is expected to result in different sediment quality values than an assessment of acute or chronic sublethal toxicity to the same sediments. Acute or sublethal responses by different biological species also can differ. For example, the lowest AET (LAET) for one chemical [e.g., PCBs] may be established by the Microtox bioassay and for another chemical (e.g., lead) by benthic infaunal analyses. The LAET is expected to be protective of a range of biological effects. Used in combination, the multiple AET can also provide a preponderance of evidence for associating environmental effects and chemical contamination. Above the highest AET (HAET) for a range of biological indicators there is a high degree of confidence that sediments will fail biological testing regardless of the test. In recent evaluations with an expanded PSEP/PSDDA database, existing Puget Sound LAET were from 87 to 94 percent sensitive in correctly predicting all known biological effects (depending on the particular biological test). HAET were 61 to 100% efficient in only predicting actual problem sediments (i.e., 0 to 39% of the sediments did not actually have the predicted effect, depending on the particular biological test).

Sediment quality values based on this tool were recommended for the following uses:

- Determination of the extent and relative priority of potential problem areas to be managed
- Identification of potential problem chemicals in impacted sediments
- Prioritization of laboratory studies for determining cause-effect relationships
- With appropriate safety factors or other modifications, for use in regulatory programs and as "trigger levels" for screening decisions on the need for further chemical or biological testing of sediments.

Sediment criteria based on definitive laboratory cause-effect studies and field verification studies will continue to be active research issues for many years. In the interim, field effects-based approaches using the AET concept provide decision tools that have the following characteristics: (1) developed empirically from field data; (2) provide chemical-specific values; (3) supported by a variety of biological indicators including acute lethal and sublethal bioassays and *in situ* benthic infaunal analyses indicative of acute and chronic effects; (4) driven by statistically significant adverse effects; (5) supported by non-contradictory evidence of adverse effects within a given data set.

DISCUSSION

Sediment quality values have been integrated into several Puget Sound programs (Figure 3). These chemical values are not used in isolation; in all cases site-specific biological testing is used to supplement or verify the predictions based on sediment quality values. Problem area and problem chemical identification was the focus of the Commencement Bay Remedial Investigation, and is currently a major aspect of the urban bays program of PSEP. This problem identification establishes a basis in toxics action plans for prioritizing potential remedial actions according to the environmental significance of contamination. Problem identification requires sensitive sediment quality values to ensure that all potential problems are considered.

The purpose of subsequent sediment remedial action is to mitigate contamination in problem areas and thereby to eliminate associated adverse biological effects. In the Commencement Bay Feasibility Study, sediment quality values based on the lowest AET for a range of biological indicators were developed as target cleanup goals. These goals correspond to those sediment

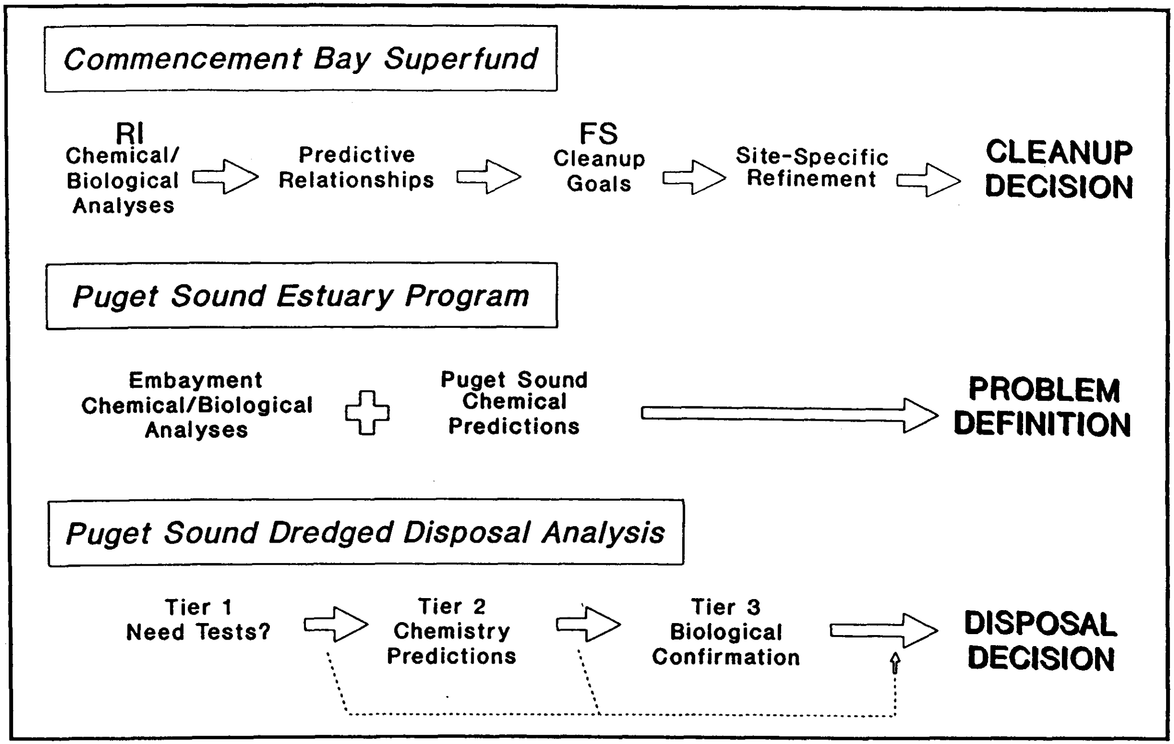


Figure 3. Use of sediment quality values in remedial action programs.

contaminant concentrations that are not predicted to result in adverse effects according to any of the biological effects indicators used to generate AET. Although the goals may not be totally protective of all potential environmental problems, they are sensitive to currently measureable effects, including effects originally used to identify problem areas in the remedial investigaiton.

A higher (i.e., less stringent) level for cleanup was identified as an alternative to the target cleanup goal. This alternative cleanup level was recommended for use should the target goal be infeasible at a particular problem area. The higher concentration alternative would be more technically or economically feasible than target goals because it would tend to require smaller volumes of sediment for remedial action. Instead of using an arbitrary multiple of the target goals, this alternative was generally based on the highest AET for the range of biological indicators (i.e., the concentration of each chemical above which all biological effects accounted for by AET are predicted to occur). This alternative cleanup level is expected to be efficient in addressing major contaminant problems.

Dredged material disposal guidelines developed by PSDDA incorporate sediment quality values to address both sensitivity and efficiency concerns (PSDDA 1988). PSDDA guidelines establish a chemical screening level (SL) above which biological testing must be performed to establish the suitability of dredged material for disposal at unconfined, open-water sites. The SL is lower or equal to the lowest AET for a range of biological indicators and is intended to be sensitive, fully protective of the environment. Contamination below the SL is assumed to be acceptable without confirming biological tests. A maximum level (ML) was also established by PSDDA as the highest AET for a range of biological indicators. The ML was intended to indicate a level of chemical contamination above which there was a preponderance of evidence for adverse effects. Biological testing above the ML is always expected to confirm the prediction of unacceptable biological effects, and is not required,

It is recognized that site-specific factors could anomalously influence predictions of biological effects based on sediment quality values. Therefore, in evaluating final requirements for sediment remedial action, selected verification of predicted effects is recommended (extensive biological testing of each sample may not be feasible). For example, in the Commencement Bay Feasibility Study an option is provided to appeal the site-specific prediction of biological effects. This optional biological testing program is consistent with the intent of other regional contaminated sediment management programs, including PSDDA disposal guidelines. Comparable tests and test protocols are recommended, and site-specific biological information overrides predictions of biological effects based on chemical data. Some specific differences between regional programs in the interpretation of biological test results may exist because of differing program goals (e.g., cleanup of nearshore sediments in a multiuse environment vs. assessment of the suitability of potentially contaminated material for disposal at a designated deepwater site).

RECOMMENDATIONS

Sediment remedial action is controversial because few objective criteria exist for quantitatively assessing more than the economic feasibility of remedial actions. A commonly expressed concern is that cleanup or disposal guidelines based solely on biological effects would likely be economically or technically infeasible. In any case, there is a strongly perceived need for a preponderance of evidence to implement remedial action. To address these concerns, the following policy questions must be recognized in research efforts:

- What degree of environmental protection is desired or required? What biological indicators are appropriate to assess the attainment of environmental goals of "no adverse effects" in Puget Sound sediments?

- Should the degree of environmental protection vary among programs or among sites that are used for different purposes?

- What should determine appropriate remedial action guidelines? Should economic and technical feasibility be incorporated into the selection of these guidelines? If so, what procedures should apply to ensure technical consistency among sites?

A range of sediment quality values such as incorporated into existing Puget Sound programs is recommended to address the complexity of these questions. In addition, tradeoffs required by balancing environmental protection and remedial action feasibility are reflected in the sensitivity and efficiency of sediment quality values, which should both be evaluated in each research study.

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CANCER RISK ASSOCIATED WITH SEDIMENT QUALITY

HENRY LEE II AND ROBERT RANDALL [1]

Establishing sediment quality criteria based on tissue residues in benthos has been limited by the lack of validated bioaccumulation models. One possible model is the thermodynamic partitioning bioaccumulation model that predicts tissue residues in the benthos based on sediment pollutant concentration, TOC, lipids, and a constant (accumulation factor). Recently, three EPA laboratory and field studies showed that this partitioning model generates reasonably consistent predictions of tissue residues for 13 PCB congeners and other slowly metabolized compounds. For rapidly metabolized compounds, it may be necessary to use a toxicokinetic model rather than a partitioning model.

Using tissue residues for regulatory purposes is limited because of the paucity of end-points. We suggest that sediment criteria, for certain compounds, can be derived by establishing the acceptable human health risks and then back-calculating for the corresponding sediment concentrations. Using the guideline tissue concentrations in the Elliott Bay Toxics Action Program, acceptable cancer risk of 10^{-5} (20 g daily consumption rate), and accumulation and/or bioaccumulation factors for benzo(a)pyrene (BaP), PCBs, hexachlorobenzene (HCB), DDE, and DDD, we back-calculated the corresponding sediment concentrations.

The sediment pollutant concentration calculated to maintain the benthos below the guideline tissue concentration was found to be up to 16,800 times lower for BaP than the values obtained by other approaches (i.e., application of water quality criteria to interstitial water, or the "apparent effects threshold" (AET) approach using amphipod, oyster, microtox or benthic effects). The differences were not constant for all toxics or between treatments, however. This approach and the benthic (AET) method were the same for DDD, for example. Calculated tissue residues using the equilibrium partitioning approach consistently exceeded the tissue guidelines by factors of 3 for DDD to 16,800 for BaP.

These results suggest that to protect human health from carcinogens, sediment concentrations lower than those derived from biological effects may be required.

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Application of a Mathematical Model (SEDCAM) to Evaluate the Effects of Source Control on Sediment Contamination in Commencement Bay

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INTRODUCTION

A simple mathematical model was developed to evaluate the relationship between source loading and sediment accumulation of problem chemicals in Commencement Bay, a Superfund site in south-central Puget Sound. The model was developed to assess the potential for natural recovery of sediments in different depositional environments and under differing degrees of source control. Acceptable sediment concentrations of problem chemicals (also designated "cleanup goals") are based on the lowest chemical concentration above which some biological effect is always predicted to occur [Apparent Effects Threshold (AET) described in Tetra Tech (1988)]. The following processes were incorporated into the model formulation:

- Sediment accumulation
- Surface sediment mixing
- Chemical-specific loss due to biodegradation or chemical diffusion.

The model was applied to the nine high-priority problem areas in Commencement Bay to answer the following questions:

- What degree of source control is required for natural sediment recovery in a reasonable time frame?
- What degree of source control is necessary to maintain acceptable sediment concentrations in the long term?
- What portions of the waterways are predicted to recover naturally in an acceptable time frame, given some feasible degree of source control?

This work was conducted by Tetra Tech, Inc. and PTI Environmental Services for the Washington Department of Ecology and U.S. Environmental Protection Agency as a part of the Commencement Bay Feasibility Study.

METHODS

The technical approach developed to establish the relationship between source control and sediment recovery includes four main components: 1) formulate

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the mathematical relationship between source supply and sediment accumulation of contamination, 2) characterize the depositional environment in each of the waterways and along the Ruston-Pt. Defiance Shoreline, 3) select indicator chemicals for major sources and source groups, and 4) evaluate chemical-specific losses due to biodegradation and diffusion across the sediment-water interface. Each of these components is briefly described below.

Model Formulation

The model treats surface sediments as a well-mixed box, the size of which is controlled mainly by the depth of the mixed layer (Figure 1). Mixing in the surface sediments arises from the physical activities of benthic organisms and, in shipping waterways, propeller mixing and ship scour. Material is supplied to the box as freshly-deposited material and removed from the box by burial. Freshly-deposited material is derived from the nearby Puyallup River and localized particulate sources within the waterways. Biodegradation and diffusive losses across the sediment-water interface are formulated as a combined first-order loss term. The variation in contaminant concentration (C) in the surface mixed layer with time (t) is described by the differential equation:

$$\frac{dC}{dt} = \frac{CI \times M}{S} - \frac{C \times M}{S} - k \times C \qquad (1)$$

change in concentration = accumulation - burial - decay over time

where:

CI = Concentration of contaminant in freshly-deposited material after source control (mg/g)
M = Rate of mass accumulation of solid material in the sediments after source control ($g/cm^2/yr$)
S = Total accumulation of sediments in the surface mixed layer (g/cm^2)
k = Combined first-order rate constant for contaminant loss by *in situ* decay and diffusion processes (1/yr).

The solution to this equation is:

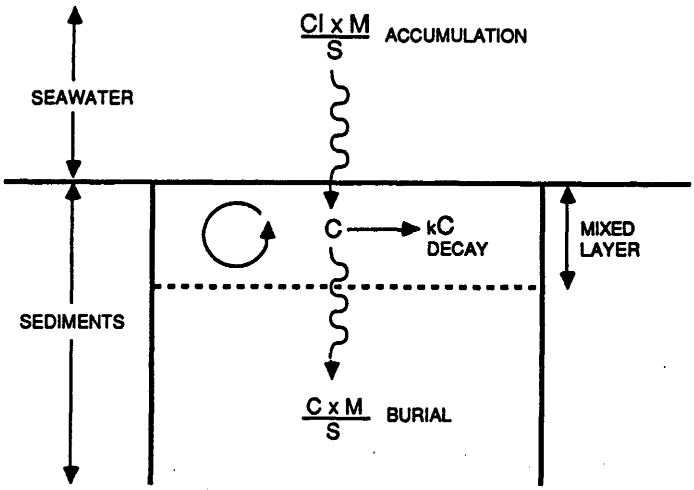
$$C = \frac{M}{(M+kS)} \times CI \times \left[1 - e^{\frac{-(kS+M)t}{S}} \right] + C0 \times e^{\frac{-(kS+M)t}{S}} \qquad (2)$$

where:

C0 = Concentration (ug/g) of contaminant in the surface mixed layer at t=0.

Characterization of the Depositional Environment

The depth of the mixed layer and sediment accumulation rate were determined from high-resolution excess Pb-210 sediment core profiles collected during the feasibility study using the techniques of Carpenter et al. (1981). Sediment mixing acts to attenuate the observed decay in Pb-210 activity with depth. The mixed layer is defined as the surface layer of sediments to the depth at which a break in the slope of excess Pb-210 is observed. Below the mixed layer, the gradient in excess Pb-210 activity can be related directly to the



Cl - CHEMICAL CONCENTRATION IN FRESHLY DEPOSITED SEDIMENT (g/g)
C - CHEMICAL CONCENTRATION IN SURFACE MIXED LAYER (g/g)
M - SEDIMENT ACCUMULATION RATE (g/cm^2/yr)
S - MASS OF THE MIXED LAYER (g/cm^2)
k - FIRST ORDER DECAY CONSTANT (1/yr)

Figure 1. Schematic of processes controlling chemical concentrations in surface sediments.

half-life of Pb-210 (22 yr). The slope in the line relating excess Pb-210 activity to sediment depth can be directly translated to a sediment accumulation rate. Sedimentation rates were also estimated from the depth of the sediment layer overlying dredging horizons of known date. Details of the approach are described in Tetra Tech (1987a).

The regional source of particulate material is the Puyallup River; however, particulate sources within the Commencement Bay waterways may play an important role on a local scale. The importance of localized particulate sources was evaluated by comparing measured particulate loads from point sources, accumulated over a representative area, to the total sediment accumulation determined from excess Pb-210. The total accumulation of sediments in the Commencement Bay waterways was compared to the total particulate load of the Puyallup as a rough test of the reasonableness of excess Pb-210 derived sediment accumulation rates.

Selection of Indicator Chemicals

Commencement Bay is a complex, multi-source site with a large number of identified problem chemicals. To simplify the evaluation of the relationship between sources and sediments, and to focus on the most serious environmental problems, indicator chemicals were identified for major sources or source groups. Source groups were combined when more than one individual source either could not be distinguished or contributed a common chemical or chemical group. Details of the approach are described in Tetra Tech (1987a).

Evaluation of Chemical Losses from Biodegradation and Diffusion

Potential losses due to biodegradation or diffusion across the sediment-water interface were evaluated for selected problem chemicals in Commencement Bay. The relative importance of biodegradation and diffusion would be expected to vary for the different problem chemicals in Commencement Bay because of their varying solubility in pore water and susceptibility to biodegradation. Potential loss due to biodegradation was determined from a thorough review of field and laboratory studies. Evaluation of the importance of diffusive loss was determined by comparing the estimated diffusive flux of problem chemicals to the total mass of the problem chemical in the mixed layer. Details of the approach are described in Tetra Tech (1987b).

RESULTS AND DISCUSSION

Identification of Major Parameters

Sediment accumulation rates determined from the excess Pb-210 data and dredging horizons are summarized in Table 1. The sedimentation rate determined to be representative of a the problem area is designated "assigned value." For cases where data were unavailable or unacceptable, sedimentation rates were estimated. The mixing depth estimated from excess Pb-210 profiles typically ranged from 10 to 15 cm. A 10-cm mixing depth was determined to best represent average conditions, and was selected to represent all problem areas in Commencement Bay. Local particulate sources were typically a negligible component of the total sediment accumulation.

TABLE 1. SUMMARY OF ESTIMATED SEDIMENTATION RATES

Problem Area	Station	Pb-210 Methods (cm/yr)	Pb-210 Methods (mg/cm^2/yr)	Dredging Horizons (mg/cm^2/yr)	Assigned Value (mg/cm^2/yr)
Head of Hylebos Waterway	HY-91	0.38	530	1,470	990
	HY-92	0.77	990	no deep core	--
Mouth of Hylebos Waterway	HY-95	1.77	2,500	no deep core	2,500
Sitcum Waterway	SI-92	unacceptable data		3,150	2,400
St. Paul Waterway		no data		no data	1,000
Middle Waterway	MD-91	0.14	230	no horizons	430
	MD-92	0.39	630	no horizons	--
Head of City Waterway	CI-91	1.26	1,760	no horizons	600
Wheeler-Osgood Waterway	CW-91	no data		no horizons	375
	CW-92	0.31	375[a]	no horizons	
Mouth of City Waterway	CI-92	0.67	950	no horizons	950
Ruston-Pt. Defiance Shoreline		unacceptable data		no records	<200

[a] Value represents the average value of the two extreme linear fits of the excess Pb-210 data to depth.

Reference: Tetra Tech (1987b) from analyses conducted by Battelle Northwest Laboratories.

In general, sedimentation rates are higher at the mouths of the waterways, and higher for waterways adjacent to or north of the mouth of the Puyallup River. This is consistent with the observation that the Puyallup River is the major regional source of particulates to the area, and that prevailing long-shore currents transport particulate material from the Puyallup in a generally northerly direction. The total sediment accumulation in the Commencement Bay waterways was estimated to be, on average, 3.6×10^{10} g/yr, or approximately 7-8 percent of the total yearly particulate load estimated by Downing (1983) and Dexter et al. (1981), respectively.

The identification of first-order decay constants representing biodegradation and diffusion was limited by a lack of degradation studies of comparable environments and the unknown nature of processes controlling pore water profiles. Based on the results of the literature review and flux calculations, a first-order decay constant of 0/yr was selected to represent first-order losses due to biodegradation and diffusion for all major problem chemicals evaluated except trichloroethene, tetrachloroethene, and 4-methylphenol. A value of 0.69/yr (i.e., a half-life of 1 yr) was selected to represent potential diffusive losses for these more mobile chemicals (Tetra Tech, 1987a); however, results presented here do not include this decay term in order to provide a "worst case" evaluation.

Model Application to Problem Areas in Commencement Bay

The generally protective assumption that steady-state between source loading and sediment accumulation existed prior to source control was applied to nearly all sources or source groups because limitations in contaminant loading data precluded a more detailed evaluation. This assumption is accurate for areas in which contaminant loading has been constant over time, is an overestimate of recovery rates for areas in which contaminant loading has decreased over time, and is an underestimate of recovery rates for areas in which contaminant loading has increased over time. In all cases where this assumption was applied, it was determined from source history or sediment profiles that loading had either decreased or remained relatively constant over the last several years.

Results are summarized in Table 2. The major indicator chemicals in each problem area are listed, as well as the types of sources or source groups to which the indicator chemicals are attributed. The model predicts that significant areas of sediment contamination defined by the indicator chemicals in some of the problem areas will recover (i.e., attain acceptable concentrations of problem chemicals) within 10 yr, given the indicated levels of source control. However, no individual problem area is predicted to recover entirely in a 10-yr time frame. Based on the simplifications and assumptions applied here, the sensitivity of different problem areas to recovery is primarily a function of sedimentation rate, the elevation of surface sediment concentrations over cleanup goals, and the degree of source control estimated to be feasible. Details of the approach are provided in Tetra Tech (1987b).

Recommendations

These results provide a useful intercomparison of the sensitivity of the different problem areas to recovery, and a rough estimate of the degree of source control necessary to effect recovery. These results should not, however, be considered as accurate predictions of sediment recovery rates. The model should be fine-tuned, using 1) improved estimates of contaminant loading to the environment, and 2) refined estimates of the spatial distribution and elevation over cleanup goals of problem chemical concentrations in sediments. These data will be collected during the remedial design phase, following finalization of the record of decision (ROD). Recovery rate predictions should be validated during baseline and post-remedial action monitoring.

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TABLE 2. SUMMARY OF MODEL APPLICATION TO COMMENCEMENT BAY PROBLEM AREAS

Problem Area	Indicator Chemical	Source Represented	Source Control Required for: 10-yr Recovery[a] (%)	Source Control Required for: Long Term[a] (%)	Fraction Problem Area Predicted to Recover in 10 yr[b] (%)
Head of Hylebos	PCB	Scrap yard unknown	Not possible	95	50
	Arsenic	Chemical co., log sorting yards, landfills	Not possible	60	70
	HPAH	Aluminum manufacturer	Not possible	65	50
Mouth of Hylebos	PCB	Unknown	Not possible	90	50
	Hexachloro-benzene	Chemical company	Not possible	95	60
Sitcum	Copper	Ore unloading facility, storm drains	Not possible	95	75
	Arsenic	Ore unloading facility, storm drains	100	85	85
St. Paul	4-methyl-phenol	Pulp mill	Not possible	>99	30
Middle	Mercury	Shipyards	Not possible	90	0
	Copper	Shipyards	Not possible	40	0
Head of City	HPAH	Storm drains, marina fires	100	35	60
	Zinc	Storm drains, shipyards, plating facility	Not possible	90	10
	Mercury	Shipyard, unknown	Not possible	75	10
Wheeler-Osgood	HPAH	Storm drains	Not possible	85	0
	Zinc	Storm drains	Not possible	70	0
Mouth of City	HPAH	Fuel storage facilities, marinas	Not possible	65	70
	Mercury	Unknown	Not possible	65	70
Ruston-Pt. Defiance	Arsenic	Copper smelter	Not possible	>99	0
	Mercury	Copper smelter	Not possible	>95	0
	LPAH	Copper smelter	Not possible	75	45

[a] Recovery estimate based on the highest surface sediment problem chemical concentration measured in the problem area.

[b] Assuming that the feasible degree of source control is 70% for Wheeler-Osgood, Middle, and City Waterways; 95% for St. Paul Waterway and the Ruston-Pt. Defiance Shoreline; and 80% for Sitcum and Hylebos Waterways (except for PCB in Hylebos, where 70% source control is assumed).

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TERMINAL 91 SHORT FILL
MONITORING A DREDGE MATERIAL DISPOSAL SITE

Douglas A. Hotchkiss*

INTRODUCTION

__Project Background__. In 1984 when the Port of Seattle proposed the Short Fill Project, there was an increased concern for the health of Puget Sound and for the effects of dredged material in particular. The 4-Mile Rock dredged material disposal site had been shown to be a "hotspot " for bulk chemistry values. The renewal of the shorelines permit for this open water disposal site, though containing new strict conditions, was being contested. More stringent interim criteria were proposed while the agencies called for a Sound wide study of dredged material disposal. Similar situations were occurring around the Sound.

Into this climate the Port launched a proposal for nearshore disposal of contaminated dredged materials behind clean fill berms with no special liners or leachate control systems. The Port had done many nearshore fills in the past relying on the theory that, if the particulates were contained and the general sediment conditions were maintained, there would be minimal loss of the contaminants since they would remain absorbed to the particulates. One of the recent (T-105, January 1982) had been monitored and seemed to confirm this idea (Port of Seattle, 1985). This project would increase the area needed at T-91 for offloading cars, and as a disposal site for the dredged material unsuitable for open water. Figure 1 shows the location of the project.

__Site and material conditions__. The primary material designated for the Short Fill was the dredged material from the expansion of Terminal 30 across an old bulk petroleum product handling and storage facility. This project required the dredging 133,000 cy. of material 87,000 cy. of which was judged unsuitable for open water disposal under the new 4-Mile Rock interim criteria. The contamination levels of the unsuitable material were typical of the Duwamish, and far from classification as dangerous waste. According to King County Solid waste criteria, they qualified for disposal at an inert construction landfill. Other Port sites which provided material for the Short Fill were several small maintenance dredgings at existing port terminals amounting to approximately

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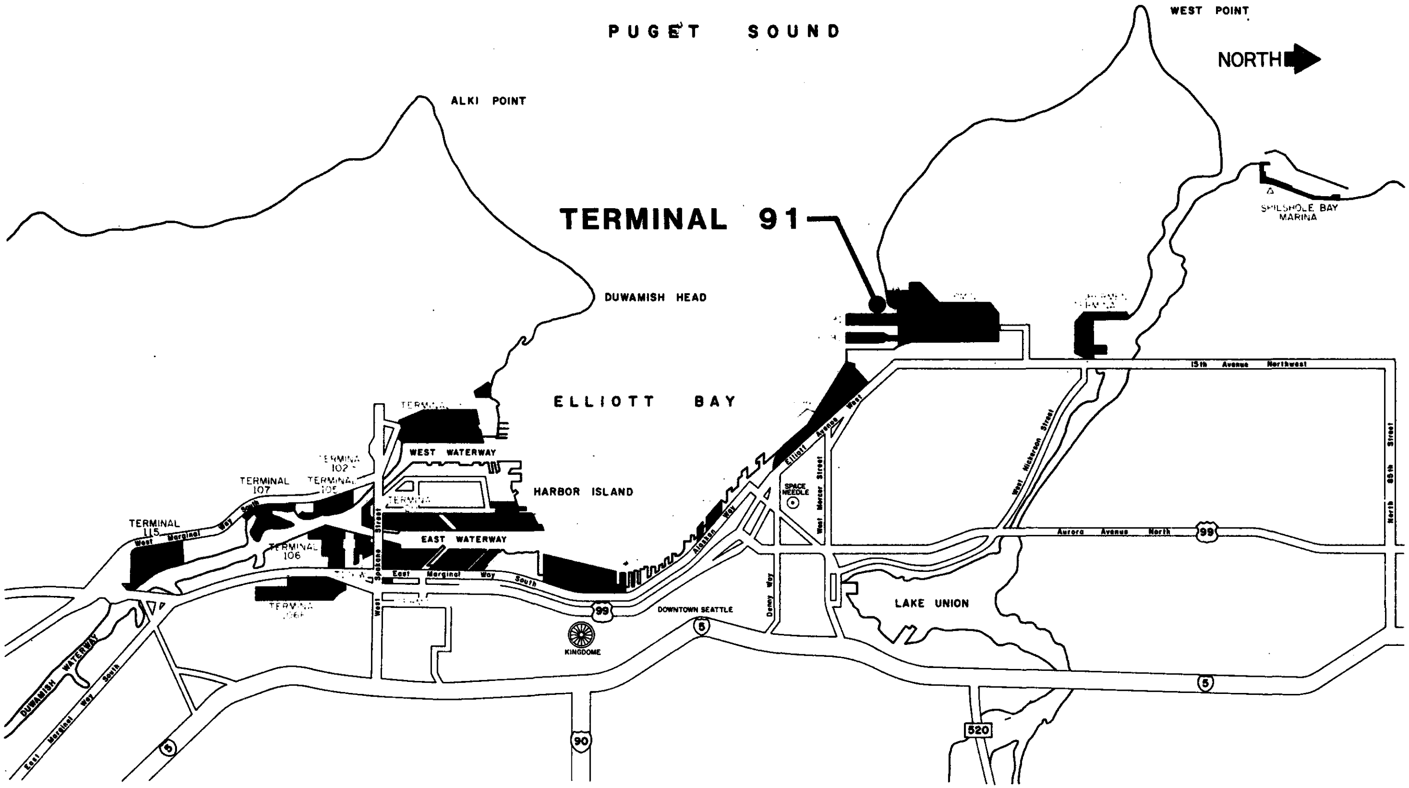
PUGET SOUND
ALKI POINT
WEST POINT
NORTH
SHILSHOLE BAY MARINA
TERMINAL 91
DUWAMISH HEAD
ELLIOTT BAY
HARBOR ISLAND
WEST WATERWAY
EAST WATERWAY
TERMINAL 102
TERMINAL 105
TERMINAL 107
TERMINAL 115
TERMINAL 106
DUWAMISH WATERWAY
East Marginal Way South
West Marginal Way South
East Marginal Way South
West Spokane Street
Alaskan Way
Elliott Avenue West
West Mercer Street
Denny Way
SPACE NEEDLE
DOWNTOWN SEATTLE
KINGDOME
LAKE UNION
West Nickerson Street
Aurora Avenue North
15th Avenue Northwest
North 85th Street
99
5
90
520

FIGURE 1

15,000 cy., that were in the same general composition and chemical quality as Terminal 30 material. There was also about 25,000 cy. of Corps of Engineers Duwamish maintenance dredging unsuitable for open water disposal. This Corps Duwamish maintenance dredging was generally less contaminated and was the last dredged material placed in the site.

The area surrounding Terminal 91 has a relatively high background contamination level from past and present sources. Among these are: its history as a large navel facility; an old garbage dump/landfill adjacent to the site; the major CSO and storm drains in the adjacent slips; and the petroleum product handling facility that has been on terminal area since the 20's. The casual disposal of raw sewage and industrial wastes, as well as the accidental spills and equipment failures have contaminated the surrounding waters, sediments, groundwater and soils. As a result, the sediment in the 90/91 slip is as contaminated as the worst of any of the dredged material that was proposed for contained disposal in the fill.

THE PLAN

Assessment of the impacts of this proposed project on the environment requires a prediction, with some reasonable assurance, of how much of the total contaminant load is mobilized and how fast it moves out of the system into the surrounding waters.

Modeling. These questions were answered by modeling the proposed project (Hart-Crowser and URS, 1985) The site particulars are: the sides are contained by existing solid fill piers; there is tidal pumping into the southern berm; and a northern berm separates the fill from the head end of the slip (to prevent deforming the old concrete bridge structure), thus forming a pond. The site as originally planned is shown in Figure 2.

Mobilization was considered by taking the chemistry data for the worst case of T-30, existing data on Duwamish sediment interstitial water for similar sediments, and combining these with the known adsorbtion constants and equilibrium partitioning coefficients for the compounds of interest. This showed how fast the compounds would leave the particulates and how much of the total concentration of any compound would be retained on site, provided that saturated/anarobic conditions were maintained.

The other main consideration was the hydraulics of the system, and therefore the rate at which the compounds might be moved through the system. Components of the hydraulics model included the existing groundwater head, the varying permeability of the system elements (berms, piers and fill), and the tidal pumping in the south berm.

The results of the modeling were:

- For metals the average release was less than 1% in a 100 years and a total release of 1% to 2%. This is primarily

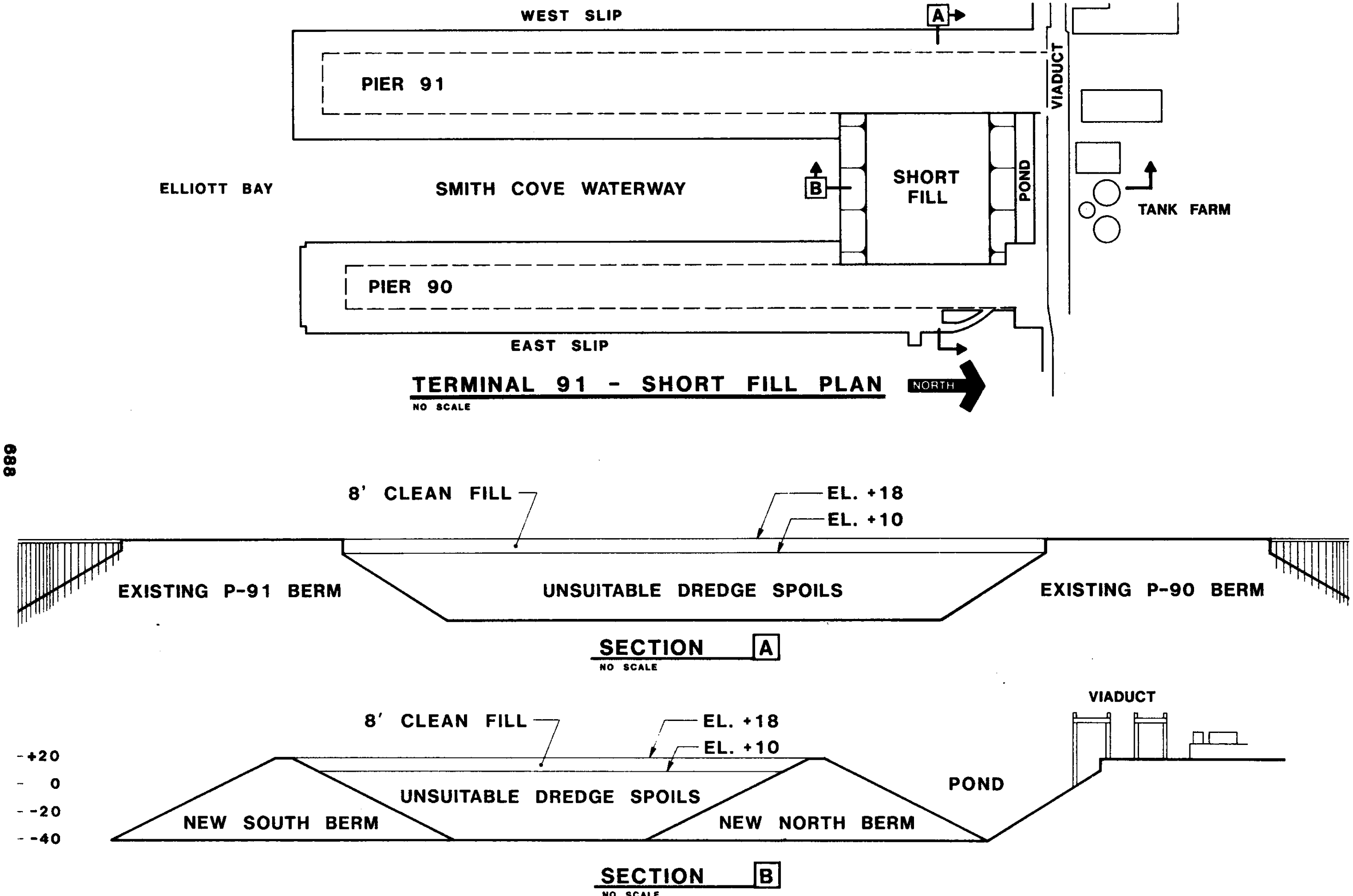
WEST SLIP
A
PIER 91
VIADUCT
ELLIOTT BAY
SMITH COVE WATERWAY
B
SHORT FILL
POND
TANK FARM
PIER 90
EAST SLIP
TERMINAL 91 - SHORT FILL PLAN
NO SCALE
NORTH
8' CLEAN FILL
EL. +18
EL. +10
EXISTING P-91 BERM
UNSUITABLE DREDGE SPOILS
EXISTING P-90 BERM
SECTION A
NO SCALE
VIADUCT
8' CLEAN FILL
EL. +18
EL. +10
+20
0
-20
-40
UNSUITABLE DREDGE SPOILS
POND
NEW SOUTH BERM
NEW NORTH BERM
SECTION B
NO SCALE

FIGURE 2

due to keeping the sediments saturated and anarobic. These results include the more mobile heavy metals.

- For organics of concern, though they were 100% available for release, the low equilibrium coefficients and the surface adsorbtion to the berm material made release rates so low that measurable levels should not be found in the south berm wells. The estimated loss in 100 yrs. for the PCB's and PAHs was much less than 1%. This was not true for the volatile organics, such as tolulene, which showed a release of 20 % in 100 years. Fortunately only a small amount of this dredged material contained volatile organics and at such low levels it was not a concern.

- The hydraulic analysis showed that the route of water flow and therefore contaminant loss in this system was through the south berm. This was due to the longer travel distance and the lower permeability of the solid fill piers on the sides. The flow through the berms is controlled by the permeability of the fill, the pathway distance through the berm, and the head of upstream ground water versus the meantide.

- The sensitivity analysis showed that, interstitial concentrations and fill permeability were the largest factors in the concentration of released material.

- Predicted concentrations in the waterway at the berm face were controlled by the release rate and dilution in the berm during tidal pumping. The predicted dilution due to tidal mixing while passing through the berm was about 200 to 1. The predicted values in the waterway were comparable to the background water quality.

Monitoring. The Port in cooperation with Department of Ecology and the other regulatory agencies, applied the modeling results to the proposed design, and drafted the "Criteria, Thresholds, Monitoring, and Remedial Actions Plan" (Port of Seattle, 1985). The project criteria were effects in the waterway (EPA. Water Quality Criteria or 10X background at the berm face). A secondary concern was percent retention of the contaminants within the fill, so quantification of contaminant mobility was also considered in the design.

The site monitoring was based on through flow and well concentrations since this provided data to directly address water quality violations and provided a reasonable estimate of total flux through and mass loss of the system.

The plan stipulated installation of 18 wells at 10 sites and 4 water sampling stations (figure 3). The 3 well depths were placed to coincide with: the hydraulically active upper layers; the shallowest layers of the dredged material fill; and finally the shallowest layer of the most contaminated project in the fill (T-30).

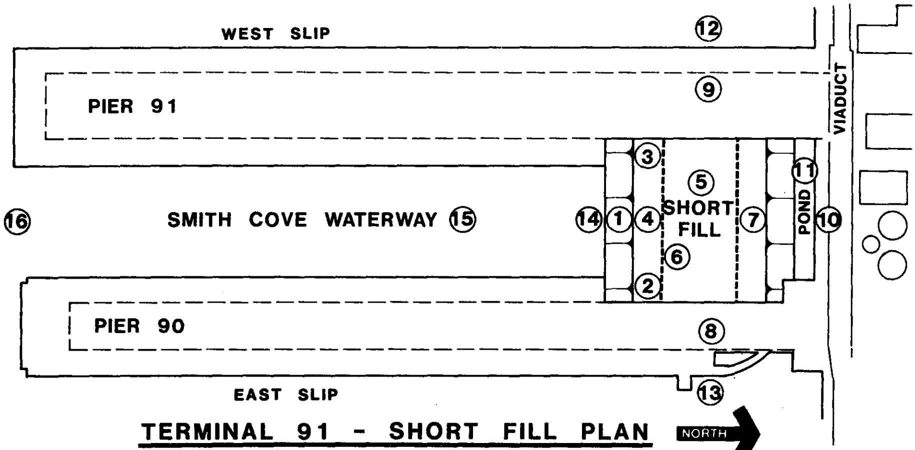

Figure 3.

The original sampling scheme included:

- Presampling (after the material was placed in the site and before it was surcharged with the clean fill on top.

- Monthly sampling after the material was in place and surcharged to follow any dewatering pulse and also to cover special sampling conditions (a tidal fluctuation sampling, and a special organic sampling).

- Quarterly sampling to follow for 2 years.

- Semi-annual sampling to follow for an additional 2-1/2 years.

REALITY

The following is a sequence of events highlighting the major variations that have effected the original plans:

- There was no additional Port expansion project so Corps of Engineers Duwamish maintenance dredging material unsuitable for open water and additional quantities of clean structural fill were used to fill the site. The result was the clean structural fill cap was deeper than planned. The dredged material came to approximately +1 ft. (mllw) with a structural fill cap covering the site to +18 ft. This should reduce the contaminant loss and provide a higher permeability conduit for the contaminated upstream ground water leaving the site.

- There was more oil in the T-30 material than expected and it escaped during the bottom dumping a portion contaminating the berm face.

- There was a 6 month delay between the filling, and the placement of the cap. This period allowed the sediment to begin to consolidate and compact under its own weight before being covered.

- Some cap material from an adjacent beach mitigation cut was loaded onto the edge of the fill causing a displacement of the surface of the dredged material rather that an even surface.

- Cap material was stockpiled on the berm, which prevented installation of the sampling wells until the berms were cleared and leveled.

- Rather than placing the cap by crane, the water was first pumped out of the site (after being sampled, and passing Water Quality criteria) and then cap was leveled across the dredged material by small loaders. This minimized the dewatering surge.

- During the capping there was an oil spill at the terminal which grounded a considerable amount of heavy bunker oil from the the 3,000+ gal. spill on the south berm face. This delayed the well installation while a portion of the berm face was removed, and the weathering process followed to determine how deeply the oil had penetrated the berm. The consequences of continued oil contamination on the monitoring program were reconsidered.

- After the results of the first several samples showed no significant water quality problems it was decided to wait until the center fill wells could be drilled before continuing the sample series. The center wells were placed in March and April.

- The Port and the Department of Ecology decided not to put a well in Pier 90 since soils information indicated lower permeability there. While drilling Pier 91 for soils information, a higher permeability layer was found and therefore a well was installed. Analysis with the hydraulic model showed that even if the high permeability layer did run through the full cross section of the pier, it would carry only a small portion of the flux out of the site. Because of this the monitoring stations in the adjacent slips were to be left out of the routine monitoring.

- The berm face wells planned for installation in the south berm face were not installed at first due to the questions regarding oil spill contamination. Also with the data results as clean as originally expected it was thought that rather than install this series of wells "up front" and have them possibly contaminated before long, it would be more appropriate to wait and install them later only if there appeared to be a specific water quality problem with a compound moving through the berm into the adjoining waterway. In this way the well placement could be optimized. This proposal was recently agreed to by the Department of Ecology.

- Well Site 6, in the shallow southern edge of the dredged material actually ended up being located in a mixture of dredged material and the cap material. Though theoretically all dredged material at this point, the field log showed a mixture of both materials.

- The actual sampling schedule was as follows:

Nov. '86	wells 2,4a, 4b
Dec. '86	wells 2, 3, 4 abc, 7 abc, 9, 11 sites 10, 14, 15, 16.
Jan. '87	wells 2, 3, 4 abc, 7 abc, 9, 11 sites 10, 14, 15, 16.
May. '87	wells 2, 3, 4 abc, 5 abc, 6, 7 abc, 9, 11 sites 10, 14, 15, 16.
Oct. '87	wells 2, 3, 4 abc, 5 abc, 6, 7 abc, 9, 11 sites 10, 14, 15, 16.
Jan. '88	wells 2, 3, 4 abc, 5 abc, 6, 7 abc, 9, 11 sites 10, 14, 15, 16.

RESULTS

Physical system. The permeabilities sampled on site after well installation differed only slightly from those used in pre-construction modeling. The heights of the upstream pond have born this out, as it generally stays in the 9 foot range, as predicted. The permeabilities were very close for the dredged fill, which is the most important as it determines rate of flow thru the system. As previously stated, an area of higher permeabilities was found in pier 91, but this was found not to greatly effect the flux estimates. The south berm permeabilities were found to be the same as those used in the modeling.

CHEMICAL RESULTS

General. The most contaminated organic samples found were from the dredged material fill, the upstream groundwater, and the pond. They were all contaminated by the same general series of PAH's. The metals profile for these areas varied, and most of the high metals values were found in the structural material of the berms and cap. The presence of many of the same organic contaminants in the upstream ground water and the pond water, coupled with the fact that these contaminants are not found in the berm between the dredged material and the pond, indicated that the source of this low level contamination for the pond is most probably the groundwater and definitely not the dredged material. The pond water at a depth of 5 meters was more similar to the ground water. Another anomally of the pond was a high arsenic level (34 ppb ± 16). This may have resulted from unique conditions at the head end of the slip that became evident only with the entrapment of the water by the berm construction, such as bird dung, or runoff carrying rat poison.

The oil spill caused no apparent contamination within the berm wells. There was no indications of organic contamination from either side.

The interstitial water contaminant levels observed in the field sampling were generally lower than those assumed for the dredged material in the model (table 1). This shows the model to be a conservative estimate as was intended.

Table 1
COMPARISON OF MODEL ASSUMPTIONS AND ANALYSIS OF INTERSTITIAL WATER FOR DREDGED MATERIAL FILL (ppb)

contaminant	model value	measured value (5 b,c)
Cd.	16.2	ND 1.
Hg.	3.3	ND 0.05
As.	15.1	8.7 ± 6.9
PCB	0.18	ND 0.1
Naphthalene	22.8	2.8 ± 1.2
Toluene	7.3	ND 1.5

ND is Non-detected

Organics. Most important from a performance and regulatory point of view was that the berm wells have not shown any sign of contamination by organics.

The upstream groundwater was shown to be a source of low level organic contaminants to the pond, both PAHs and volatiles. Most of the organics appear to be lost in the surface water, since they were only found there once, during the winter, when the areators were not functioning well. This loss of organics at the surface was undoubtedly due to multiple causes which may include aeration, oxidation, photodegredation, and biodegradation. These organic constituents have not moved deeply into the berms and the cap as shown by the lack of organic contamination found there. Table 2 shows some of the more consistent of these compounds.

Table 2
COMPARISON OF ORGANICS IN THE POND AND UPSTREAM WELL (ppb)

contaminant	upstream well	pond, surface	pond 5m.
Benzene	18, 29, 19	1.7, 2, 1	1.4J
Toluene	6.1, 8. 7.1B	1.5, 2, 0.8	0.5M
Total Xylenes	3.9, 7B, 3.9B	2.4, 2, 1.8	0.7M
Naphthalene	27, 30, 26	1.6, 2.6J, 1.6	18
2-Methylnaphthalene	18, 31, 35	0.9, 0.5J, 0.9	2.8
Acenapthene	2.4, 3.7J, 3.2	0.5, 2.2J, 0.6	6.2
Dibenzofuran	3.9, 4.1, 4.8	0.8, 1.1J, 0.8	2.6
Fluorene	4.6, 5.2, 5.9	0.6, 2.9, 0.6	2.5
Phenanthrene	3.1, 3.1J, 4.3	0.8, 1.1J, 0.8	2.5

J = Estimated value
M = Estimated value with low spectral match
B = Also found in blank

The TOC is the one area where organics have been found in the berm, the fill, and the surrounding water column. The lack of an obvious gradient and the background values makes it hard to establish the origins of the material.

Metals. There were many more interesting results in the metals since they are more mobile and also more commonly found in the surrounding environment and in the construction materials. The origin of the metals was the most important question and the system did not give any straightforward solutions.

In the early data sets there was a small amount (9.6 ppb $\pm$ 1.7) of Cu. evident in the pond, in the top and middle of the North berms (3.1 ppb $\pm$ 1.5) and in the waterway (1.6 ppb $\pm$ 0.9). This appeared to indicate the pond as a source of Cu. but, Cu. was only detected once (2 ppb) in five samples in the upstream groundwater. Later sampling, including the fill and cap, showed that the highest values (31 ppb $\pm$ 20) for Cu. were found in the "clean" structural fill material of the cap.

The most obvious tracer for the fill was tin (Sn.) as it was the only compound found in the dredged fill (55 ppb $\pm$ 36). Its chemical activity is similar to other metals modeled. Its activity would be different (more mobile) if a large percentage was found in the methylated form. This should be investigated in the future. Sn. may prove to be a good indicator for positive indications of leaching.

Well #3 in the south berm was damaged, but has remained constant before and after the break except for an elevation in Mn. (undetected to 133 ppb). This indicates probable contamination with gravels from the berm material since Mn. is a common crustal element released with the weathering of igneous rock. Since no other contaminants were evident it was decided to continue the use of the well, for which there was previous background data rather than sink a new well and start over in that site.

The highest concentrations of Ni. were in the bottom of the south berm (100.6 ppb ± 12.8). There were also varying concentrations found in the bottom of the north berm (39 ppb ± 31), in the "clean fill" cap (42.7 ppb ± 25.2), in the fill at station 6, which contains some cap material (15 ppb ± 15), in the pond (6 ppb ± 1.8), and in Pier 91 (5 ppb). In the dredge fill (5 b,c), Ni. was only detected once (2 ppb) in 6 measurements. This distribution indicated that the origin of the Ni. in the bottom of the berm was either due to the structural material of the berm, or possibly the surrounding sediments, which were high in Ni.(65 ppm ± 7). These sediments could have mixed with the berm materials during placement by dump barge.

Dewatering surge: Due to the final construction sequence no large dewatering pulse was expected. Even with the alteration of the sampling plan, the early sampling schedule was adequate to have detected a large dewatering surge. The data did not indicate that a surge occurred. Early data for a few metals seemed to show a possible few percent increase attributable to a dewatering pulse, but later data including the fill material shows these metals are associated with the structural material.

The "clean structural fill" used to cap the dredged material has provided some of the most interesting results. It was a material that more easily released crustal elements than the berm structural fill. The well in this material has been the site of the highest values for Cu., Al., Sb., Fe., Si. , Ti., Vn., Zn., and Hg. These values were consistently greater than those for the dredge fill below it. Some of the metals, notably Hg, did not adhere consistently to this pattern, making the pond a possible source for these constituents.

CONCLUSION

The data discussed here are just the early findings and by no means a complete final analysis. Soon with the additional data being developed we plan more sophisticated statistical evaluations of the results. Preliminary results have suggested some additional research evaluations and analysis that will help understand the events taking place. This brief look at the early results, has produced the following conclusions.

- The fill is behaving close to what was predicted in the modeling.
- There is no indication at this time that the dredged material in the fill is causing water quality problems in the adjacent waterway.
- The movement of the upstream contaminated groundwater into the pond has been documented by the difference between the

pond and the new head end of the slip. The majority of the higher metal results are associated with the "clean" structural fill material of the cap or the berm.

- The organics in the system do not readily move thru the berm; neither from the fill into the south berm nor from the pond into the north berm.

The extensive modeling of the system assisted the regulatory agencies in judging the proposed project, and the Port in the designing the monitoring. The field results so far are showing the model the be reasonably accurate and a very appropriate and helpful tool in understanding such a proposed project.

The interesting surprises in the project have come from those areas not focused on by the model, and in documenting the movement of the groundwater into the pond. So far we have a successful containment project and we have found we have a lot to learn about leaching from "clean structural fill".

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Sediment Capping at Denny Way CSO

Pat Romberg[1] and Alex Sumeri[2]

Summary

Source control has been effective at reducing toxicant discharges into Puget Sound but further remedial actions will be required at many sites if contaminated sediments are to improve in a timely manner. Of the 32 toxicant "problem areas" identified in the lower Duwamish and Elliott Bay, the area off Denny Way CSO was prioritized as one of the top sites needing corrective action. Metro recently developed a CSO Control Plan that will greatly reduce the volume of overflow at Denny Way, but it may take years before the required projects are completed. Also, Metro has been implementing source control activities in the collection system to reduce toxicant loading to Elliott Bay and is now proposing to cap offshore sediments with a layer of clean material as a demonstration project of environmental enhancement.

The proposed capping project will be performed jointly with the Army Corps of Engineers who will obtain the cap material during the next Duwamish maintenance dredging now scheduled for February 1989. Only material dredged from the head of navigation will be used for capping since previous tests showed this to be clean sand. The cap will be placed with a bottom dump barge that is opened slowly allowing the sand to "sprinkle" over the bottom. This capping method was successfully used in a project on the lower Duwamish where a cap thickness of one to three feet was obtained. This paper provides further details regarding the Denny Way capping project and a review of other capping projects.

Denny Way Capping Project

Site conditions. Denny Way CSO has a shoreline discharge located in Myrtle Edwards Park at the north end of the Seattle waterfront and overflows

1 Municipality of Metropolitan Seattle
2 U.S. Army Corps of Engineers

about 50 times a year. Stormwater separation projects will reduce the overflow frequency to about 10 events per year yielding nearly a 90 percent reduction in overflow volumes by the mid-1990's. Bottom sediments near the outfall contain substantially elevated levels of heavy metals and organic toxicants and benthic communities are impacted (Malins et. al. 1980, Tomlinson et. al. 1980, Romberg et. al. 1984, Comiskey et. al. 1984). Heavy metals and organic PAH's have distribution patterns similar to that of lead shown in Figure 1, but PCBs levels are patchy with no clear pattern. Concentrations are highest near the outfall and decrease more rapidly with distance offshore (increasing depth) than they decreased with distance along shore (Romberg et. al. 1987).

Based on bulk sediment chemistry values there is no indication that offshore sediments would be classified as a hazardous waste. Sediments taken from a sewer line upstream in the Denny Way drainage basin pass the special elutriate test, used for defining hazardous wastes, and those sewer line samples have much higher bulk sediment concentrations than occur in offshore sediments. Even with intermediate chemistry levels, it is doubtful that sediments from near the outfall would qualify for disposal at the new open water dredge spoils disposal sites.

<u>Capping considered best alternate</u>. There are only two other alternatives to capping and these are to either dredge the contaminated sediments or to do nothing and let natural sedimentation cap over the affected area. Metro has considered these alternatives and finds them both unreasonable. Dredging is ruled out because site conditions are not severe enough to warrant the high costs associated with this method and there are severe limitations in finding a place to dispose of contaminated sediment in Puget Sound. Waiting for natural capping is ruled out because sedimentation rates are so slow in Elliott Bay that it could take 20 to 60 years before 6 inches of new material accumulates.

Ideally, it is preferable if all discharge sources are eliminated before capping but there are several practical benefits to capping at this time. First, sediment quality improves immediately instead of waiting perhaps 20 - 60 years and new sediment concentrations should remain below existing conditions due to the reduced sources. Secondly, the cap provides a new clean environmental background which makes it easier to identify remaining toxicant sources and focus on their elimination upstream. Also, this project will provide information and experience that will expand our understanding of capping as a tool for improving environmental conditions in Puget Sound.

Size and thickness of cap. Proposed boundaries of the capping project are shown in Figure 2 and enclose an area about 200 feet wide and 600 feet long. A cap this size will cover the highest sediment concentrations closest to the outfall and a good portion of the lower sediment concentrations that extend beyond. This cap is not intended to cover all the area above background because concentrations all along the Seattle waterfront are above background. Figure 2, also shows that the cap will cover most stations where sediment values exceed the "Apparent Effects Threshold" (AET) values established for benthic infauna (Tetra Tech 1986). While these AET values are not standards, they do provide an indication of the lowest chemical levels expected to cause a change in benthic community structure. Many stations exceed one or more AET values for heavy metals, but only one station exceeds an AET for organics (PCBs only).

Average cap thickness of 2 feet is proposed based on past experience plus limitations on the amount of clean capping material available. About 9,000 cubic yards of clean material is required to cover the proposed capping area with 2 feet of material. It is estimated that about 10,000 cubic yards of clean sand can be obtained from the head of navigation during a single dredging project. This volume is a limiting factor which means that changing the thickness would require a corresponding change in aerial coverage.

Monitoring considerations. Demonstration projects are an important form of applied research that can further our knowledge on how well capping works. There are four main components to a monitoring program as follows: 1) Bottom depth surveys are needed before and after capping to determine cap thickness and proper placement. 2) Surface grab samples are needed over time to see if there is any accumulation of toxicants in the cap due to current discharge conditions. 3) Sediment cores are needed periodically to see if there is any migration of underlying toxicants up into the cap, and; 4) Benthic community samples are needed over time to document how well the cap is recolonized by benthic infauna.

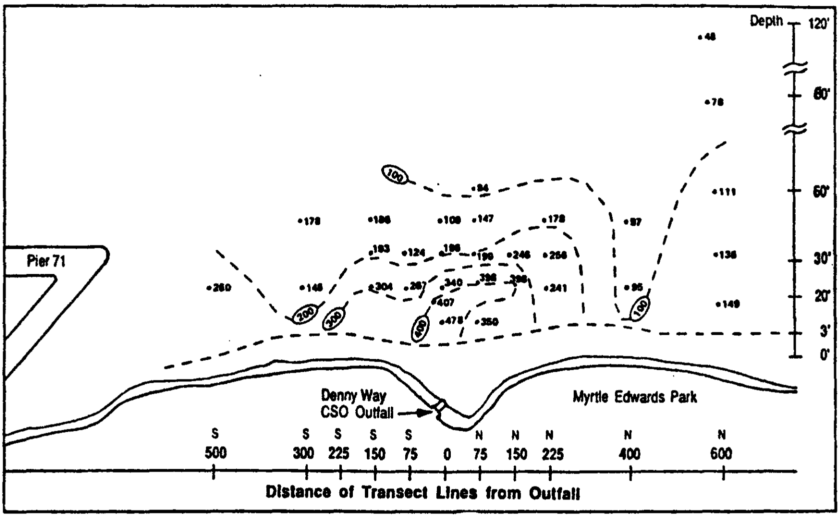

Figure 1. Distribution of lead concentrations in surface sediments off Denny Way CSO.

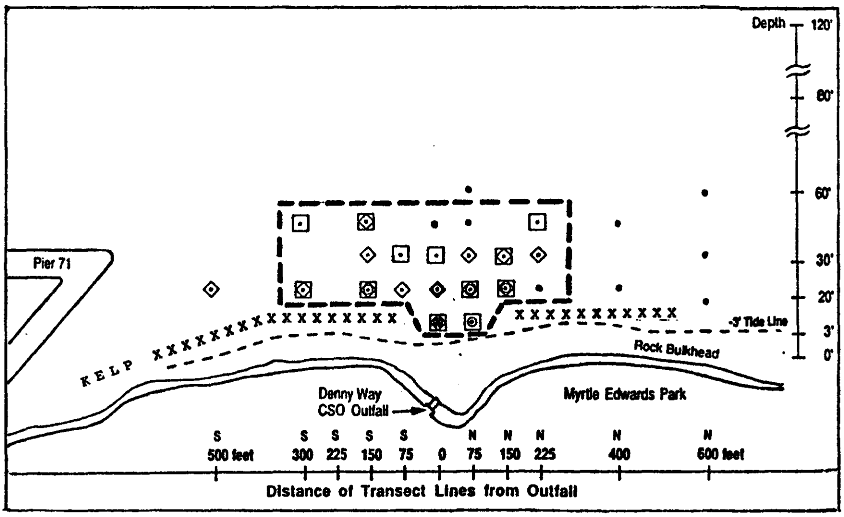

Figure 2. Boundary of capping area is a rectangle about 600 feet long and 200 feet wide. Stations having heavy metal concentrations that exceed AET values for benthic community change are indicated by the following symbols: * = cadmium; O = lead; ◇ = mercury, and □ = zinc.

Capping Contaminated Sediments

Before continuing with details on a proposed method to cap the contaminated bottom sediments in the vicinity of METRO's Denny Way Combined Sewer Overflow, a brief overview of capping will be presented to provide you confidence in the method. This overview will review international capping applications, laboratory capping research findings, monitoring results of the 1984 Duwamish capping project, potential and proposed capping applications in Puget Sound, and the Corps of Engineers' policy on participating with public entities in beneficial uses of material dredged from federal navigation projects.

The technical and operational feasibility of isolating contaminated sediments from the overlying water column with a clean soil cap has been demonstrated by laboratory research and practical field applications since the late 1970's. Upland disposal sites near dredging operations have become virtually extinct. Truck haul to more distant upland disposal sites (if they can be found for "contaminated" material) is usually at least 2-5 times as expensive as open water disposal. Nearshore disposal requires costly monitoring. Also, isolating contaminants is more difficult. Subaqueous capping is often an economically viable alternative.

The London Dumping Convention (LDC) to which the United States is signatory is an international treaty that is implemented thru the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter. In February 1984 the LDC accepted the capping concept, subject to additional monitoring and research, as a means of rapidly rendering harmless the contaminants of concern in sediments.

Design Parameters. Many design parameters must be considered in a capping project, making each design highly site-specific. The main design considerations include but, are not limited to types of contaminants, contaminated and capping sediment properties, disposal site water depths, future use of the disposal site, types of dredging and disposal equipment available, cap thickness, cost, water currents, cap erosion potential, ship prop scour, bottom contours, positioning control, monitoring requirements, etc. Considering the above, the necessity of an engineered design for capping quickly becomes obvious. The many design variables of capping operations also explains the variety of projects and equipment that has been used.

Capping Applications. Brief, details of several capping projects follow:

Central Long Island Sound has been used as a disposal site for dredged and other materials for many years. Since the late 1970's the area has also been used to dispose of and cap contaminated dredged material in 55 to 70 feet of water. At least six of eight mounded level-bottom capping sites have been documented. Contaminated (Cd=25ppm, Cu=520ppm, Ni=50ppm, Pb=450ppm, PCB=5300ppb) material (33,000 to 92,000 cubic yards (c.y.)) has been dredged by clamshell and point-dumped with scows at buoyed disposal sites. Silt and sand caps (40,000 to 1,300,000 c.y.) were placed by hopper dredges and scows. Mounds were 4.5 to 13 feet high and a radius of 400 to 500. Of the original volume 80% was deposited inside a 100-foot radius; 90% within 400 feet (Morton 80). In point dumping one must be careful to also cap the periphery of the contaminated material. The need for accurate positioning can not be overemphasized.

At New York Bight 860,000 c.y. of contaminated (mean values in ppm:Cd=14, Cu=711, Pb=767, Zn=1813, and PCB=1.9) dredged material was point dumped and capped by 1,800,000 c.y. of fine sand with scows and a hopper dredge in 75 feet of water. Cap thickness averaged 3-4 feet with a maximum of 6-9 feet. Two taut-wire buoys, with elastic moorings to reduce tide-induced lateral movement, were used to precisely dump on target (O'Conner 82).

In Hiroshima Bay, Japan in-situ sewage sludge with high levels of COD(42mg/g), T-Phosphorus (0.77 mg/g), and T-Nitrogen (2.4 mg/g) were capped with 40 - 50-centimeter layers of clean sands at two projects (Togashi 1983 and Kiekegawa 1983). These test projects were carried out to determine the feasibility of restoring the benthic environment (including fishing grounds), and reducing nutrient loading of the water column. Marine sands were dredged and transported 50 kilometers for these demonstration projects.

In one case, a 19,200 square meter area was capped using a specially constructed 2000-cubic meter (c.m.) sand spreading barge. Two conveyor belts, each with a capacity of 1000 c.m./ hour, fed the sand to a 2.5-meter diameter telescoping vertical pipe (tremie). The cap was placed at a rate of 6.5 to 13 c.m./min. with a swing speed of 5 m./min. while one end of the barge swung about a spudded end. Considering the time (14 min.) required to back the barge after each swing, each 40 m. cycle required 30 min. and effective placement was 208 c.m./hour. The mean cap thickness was 55 cm. and ranged between 5 cm. and 1.35 m.

In the other case, sand was brought by a hopper dredge and loaded into a scow moored to a pumpout barge. A hydraulic suction pump was used to transport the sand slurry thru a 5-meter wide submerged spreader bar with diffuser ports while the barge was being moved by anchor cables. A 44,800 square meter area adjacent to the above area was capped. To determine the rate of sand being pumped the pump's suction pressure was calibrated by pumping sea water and various sand-water concentrations.

Results from monitoring these projects between 1979 and 1984 (Kakigawa 1986) show lower release rates of nutrient salts (particularly Phosphorous), an increase in the number and species of benthos, but, no conspicuous effects on COD were observed.

In Osaka Bay, Japan 3,000,000 c.m. of eutrophic harbor sludge (COD of >30mg/g) is proposed to be used for beach restoration. The sludge is to be placed behind caissons and capped with 2 meters of sand (Matsubara 86).

Rotterdam Harbor's First Petroleum Harbor could not be dredged between 1976 and 1981 (d'Angremond 1984) as no disposal permit could be obtained due to the high level of contaminants (Dieldrin=1.9ppm, Endrin=0.5ppm, Isodrin=6.0ppm, PCB-1260 (aroclor)=7.0ppm, Oil=10,200ppm). A maintenance dredging backlog of 1.5 million c.m. had developed by mid 1981. An agreement was reached for a one-time cleanup of this harbor, the "Putten Plan". The plan included halting the discharge of contaminants into the harbor and the dredging of pits in the harbor bottom, disposal of the problem material into them, and capping with 60-90 cm. of clean clays excavated from the pits and stockpiled adjacent to them. The caps were placed by dozing the clays into place with a cable-suspended bottom leveler. In all 1.82 million c.m. of contaminated material was disposed of in the pits. Strict controls on dredging and disposal required that no suspended contaminants leave the harbor. A hooded suction head to trap gases, a degassing system, a diffuser for placing hydraulically dredged material into the pits, and automated positioning equipment for the suction head and the diffuser were used to minimize spread of contaminants.

The Lower Duwamish Waterway in Seattle, experienced a minor shoal in 1983, which limited navigation to 25 feet in the 30-foot federal project. Contaminants (PCB 1242=1400ppb, PCB 1260=3100ppb, Aldrin=180ppb, 4-4 DDD=80ppb, 4-4 DDE=30ppb, Acetone=494ppb, Methylene Chloride=805ppb, Cd=1.4ppm, As=22ppm, Cu=130ppm, Pb=190ppm, Zn=359ppm) in the fine-grained material precluded unconfined open-water disposal. The Seattle District, Corps of Engineers in a demonstration project disposed of 1100 c.y. of contaminated material in a subaqueous depression in the West Waterway and capped it with three survey-positioneded bargeloads (4200 c.y.) of clean sand maintenance dredged from the upper Duwamish River waterway (Sumeri 84). A cap with a mean thickness of 2 feet was placed by "sprinkling" sand at a rate of 27 c.y./min. from an incrementally opened split-hull barge. This slow capping procedure allowed the use of conventional equipment and did not displace the contaminated sediments. Hydrographic surveys between barge dumps allowed evaluation of capping progress and adjustments in barge positions during the next dump.

The Corps' Waterways Experiment Station monitored the project for 18 months. Vibracore samples were tested for total concentrations of PCB's, Cu, Pb, and Zn at 4-cm intervals and showed no movement of contaminants into the cap material. The interface between the contaminated and cap sediments continued to be sharp and relatively unmixed.

The Waterways Experiment Station performed laboratory capping efficiency tests using sand and silt from Vicksburg, Miss., and clay from New Halem for capping materials. Contaminated sediments (PCB=17,630ppb, PAH=314,550ppb, Cd=20.5ppm, Cu=2525ppm, Hg=3.34ppm, & Pb=368ppm) from Black Rock Harbor in Bridgeport, Connecticut were capped by different thicknesses of the various materials in reactor tanks (Brannon, et al 85). Polychaetes (Nereis virens - worms) were added to some tanks to test infaunal organisms and to provide a source of bioturbation. Clams (Rangia cuneata) were suspended in the water column above the capping material to determine if contaminants were moving thru the cap into the water column. Test results indicated that a 50-cm cap of any of the three tested, even when penetrated by organisms, is effective in preventing the transfer of chemical constituents and microbial spores to the overlying water and biota during a 40-day experiment. A cap as shallow as 2cm resulted in a 40- to 81-% reduction in overlying water oxygen demand and transfer of NH^4-N from Black Rock sediments into the overlying water. A cap depth of 22cm of any of the tested material was sufficient to stop chemical exchanges between capped Black Rock sediment and the overlying water. Clay was found to be a more effective cap material than silt, and silt more effective than sand. These lab tests accomplished what field monitoring could not provide - controlled conditions w/o external influences.

Dutch Kills contaminated sediment (PCB=21,290ppb, PAH=88,880ppb, Cd=97ppm, Cu=1925ppm, Pb=1430ppm) was tested with Edgewater sediment as capping material. A one-year laboratory study indicated addition of a 10-cm cap of Edgewater sediment along with a suitable depth of material to isolate burrowing benthic organisms from the contaminated material and prevent current and wave erosion from removing the cap should prevent movement of contaminants into the water and biota (Brannon, et. al.86).

Denny Way Combined Sewer Overflow vicinity capping of contaminated sediments would be performed similarly to the above-mentioned Duwamish capping project; except, placement of the sand will need to be controlled better. Clean sands are to be dredged by clamshell, and transported to the Denny Way site by split-hull scow. Placement of the sands differs in that the scow is proposed to be pushed sideways sprinkling a 128-foot-wide sand blanket. Barge displacement changes proposed to be measured by two pressure transducers mounted in stilling wells (to dampen rapid wave-caused pressure changes) at each end of the scow. Telemetry signals from the pressure transducers are to be transmitted to the microprocessor controlling the barge position. A shore-based automatic-tracking laser range-azimuth positioning system will track a prism mounted on the scow and send the positions to the microprocessor on the tug pushing the scow. The tug skipper will see his course on a monitor and where the target prism is at the time, as well as the rate of change in barge displacement - or rate of sprinkling the sand cap. A two-foot thick cap is foreseen at this time. Due to a 12-mile fetch, wind driven waves from the southeast can become sufficient to move sand particles of the size proposed for the cap. Since these conditions are expected to occur only once or twice a year for short periods only, no significant erosion is expected.

Potential and proposed capping projects abound in the Puget Sound area. Elliott Bay "hot spots" have been identified where capping could be economic and effective. Commencement Bay has its Superfund sites. The Eagle Harbor wood treatment contaminants have been added to the Superfund list. Corps of Engineer maintenance of the lower Duwamish River navigation project may be a future candidate. The Everett Navy Homeport includes a 300 to 400-foot-deep capped disposal site; a record depth. Extensive lab testing, model studies, and engineering have yielded a placement method. Initially, a berm will be constructed; then contaminated material will be dumped from barges and be contained by the berm. The contaminated material will be capped in a controlled manner; either with a diffuser and hydraulic dredge or by clamshell and hydraulic pumpout at the site. The capping material will be allowed to settle on the contaminated material. Careful positioning and monitoring of the fate of the capping material will ensure a 1-meter cap. A total of 3.3 million cubic yards will be dredged, of which 928,000 is considered contaminated. Simpson Tacoma Craft Company will cap approximately 17 acres of contaminated nearshore land with 2-8 feet of sand dredged from the Puyallup River creating an intertidal habitat.

Corps of Engineer Policy requires disposal of dredged material at the lowest cost consistent with sound engineering practices and appropriate environmental quality standards. This does not preclude public entities from sponsoring beneficial uses of the material and funding extra costs.

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Summary Statement

Keith E. Phillips *

As the importance of managing contaminated sediments increases, the tools we use in that management are evolving, mostly along natural lines. Key steps in this evolution are 1) management by comparison to background, 2) management by effects testing, 3) management by criteria and standards, and 4) management by technology requirements.

The first step in this evolution is often, because of a lack of better information, the simple assumption that the presence of contaminants means there is risk of adverse effects, and that chemical concentration is proportional to that risk. This results in "background levels" being used as regulatory tools and management objectives. As an example, background levels were used in dredged material management until very recently in Puget Sound; and they remain important in land disposal decisions.

The next step is reliance on effects testing. Sediments have been, and will likely continue to be for some time, managed (at least in part) by direct effects testing (i.e., biological tests) of the sediment. As we begin to learn about the relationship between sediment chemicals and biological effects, we are starting to develop and use numerical criteria and guidelines for predicting the effects of sediment contamination. Along the way, we are learning and demonstrating new technologies for treatment and handling of contaminated sediments. And it was clearly shown today that there is still some gap between the intended design and plan, and what is achieved in the real world. Overall, we lack sufficient experience to rely on technology standards or to define "best technology" at this time. Rather, technology is used to address problems identified by effects testing and chemical guidelines.

We still have a lot of work ahead of us. And there are key social and administrative decisions that must be made along the way. What quality of sediments do we want to have in the Sound? Will source control ever be 100 percent? How long should we plan for this to take? How much are we willing to pay to clean up existing sediment contamination? If we remove it, where do we put it?

Along with this policy development, these questions will also require further technical research and follow-up monitoring to assess the true consequences of the different alternatives we are facing in managing contaminated sediments.

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SURFACE WATER MANAGEMENT

WHAT DO WE KNOW AND WHAT DO WE NEED TO KNOW?

William H. Funk,
State of Washington Water
Research Center
Session Chair

Introductory Statement

William H. Funk*

Considerable progress has been made towards restoring the nation's water quality since the enactment of the Clean Water Act of 1972. However, many bodies of water including lakes, estuaries, and even some streams have not responded adequately to the massive clean-up efforts begun sixteen years ago.

Studies made by the university water researchers and agency personnel have shown that a large portion of bacterial and plant nutrients as well as toxic materials, pesticides, metals, oils and other pollutants reaching water courses have no defined point of origin. The U.S. Environmental Protection Agency reported to Congress in 1984 that nonpoint source pollution was the principal cause of water quality problems in six of the ten EPA regions.

The effect of pollutants reaching the waterways whether from point or nonpoint sources is essentially the same. Nutrients, primarily phosphorus, nitrogen, and essential trace metals, cause increased nuisance algae and weed growth. Large quantities of organic matter result in dissolved oxygen depletion. Excessive sediments smother bottom organisms, cover spawning beds, and destroy valuable habitat. Metals, pesticides, and other toxics poison microorganisms and invertebrates and may injure or kill larger aquatic life forms.

Much of the diffuse source data collected across the United States today have been used mainly to indicate trends and are not directly applicable to specific cases. The purpose of the surface water management session is to define what we know about pollutants entering the Puget Sound area and what we need to know about them. It is readily understood that there are large data gaps and a great amount of research is needed to make prudent management decisions. A very useful aspect of this session is that a number of methods used to mitigate, alleviate, and control sources of surface water pollutants are discussed as well as their state of development, effectiveness and applicability. The session represents an initial effort to better understand diffuse surface pollutants--their effect and possible control.

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Agricultural BMP's: Are They Doing Their Job?

John Philip Andrews*

In four of our northwest counties, Whatcom, Skagit, Snohomish, and King, commercial farming largely coincides with major floodplains. This is significant in two ways;

1. These are areas of occasional flooding with high water tables and short and direct paths to streams and rivers.
2. Their openess makes farming a visible element of the landscape. As an industry, they are subject to public scrutiny and critique.

As such, poor farming practices are easily noticed and the target of enforceable regulation. More than any other industry, commercial farms are being pressured to"clean up their act". In addition, farming is being directly linked to wetland protection. This all means that large-scale farming is becoming a public concern.

Because of these relationships, farm management is correlated with water quality. Pollutants most commonly associated with farming are nitrates, pesticides, bacteria, and phosphates. With approximately 700 commercial farms in the four counties, "cleaning up their act" will have a measurable impact on water quality.(1)

<u>Best Management Practices</u>

Congress passed Public Law 92-500 in 1972 to protect streams and lakes from water-borne pollution. Section 208 of this law dealt with non-point pollution, or that pollution which can't be traced to a single pipe or outlet. In this section, the term "best management practice",or BMP was first used. This can be defined as a proven land management practice or practices that are considered the most effective and practical means of controlling non-point pollution.

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(1) Poll of four Conservation Districts Feb. 29, 1988

Local government, with EPA grant assistance, developed publications describing BMP's. Both King and Snohomish County developed a Farm Water Quality Management Manual describing and illustrating BMP,s in 1977.

The Soil Conservation Service has defined and uses some 150 "Conservation Practices" technically that may be called in laymans' terms, BMP's. Of these practices, around forty apply more directly to our marine coastal environment.(2) Of these forty, around five have the most direct impact on water quality. These are:

1. Roof gutters and downspouts.
2. Animal waste storage.
3. Fertilizer application timing and rates.
4. Pasture soil management.
5. Stream buffers.

These five practices are mentioned because they best isolate or separate nutrients from surface and groundwater and are the most effective "package" of BMP's to combine as a farm planning unit. These practices are at the heart of what we could call farm nutrient management.

<u>Levels of BMP's</u>

We know that the construction of roof gutters, concrete confining pens, manure ponds, and storage tanks will segregate man made waste from free flowing water. These may be called "basic BMP's" that temporarily meet our obligations to store manure. We also know that regular scraping of livestock holding pens, pumping of waste storage tanks, and emptying of manure ponds will satisfy system requirements. These may be labeled "intermediate BMP's" and require that the farmer have a good knowlege of operation and a little of waste management. Few argue that this concrete and steel system does not isolate the waste from uncontaminated water.

Real conservation, however, lies in the knowledge of optimum timing of manure application, the amount, in inches of manure to apply to each acre depending on soils and plant needs, as well as keeping the nutrients within the active root zone to minimize its loss to groundwater. It's these and other management practices that we can still refine. We could call these "advanced BMP's.

(2) U.S. Soil Conservation Service Technical Guide Standards and Specifications, Vol. 1 & 11

Predictable Results

Let us focus on these five practices and how they work as a unit.

1. Roof gutters and downspouts

These are effective and economical in channeling clean water into pipes for transport away from animal waste areas or into a manure pond to dilute manure for later application. In our area, forty five inches of rainfall falling onto an average combined roof area of 30,000 square feet equals 840,000 gallons of clean water that no longer mixes with and carries waste into water bodies. Once installed, these practices need little management.

2. Waste storage ponds

These are earth sided ponds capable of holding one to ten million gallons of combined wash water and cow manure.

Pond size. Recommended storage times and pond volumes have increased from a few months to six to nine months storage time to insure sufficient isolation of pollutants during the wet winter months. Rainfall variability and management safety factors combine in recommending larger ponds to decrease forced early spring and winter application due to overfull ponds.

Pond sealing. In most soil conditions, manure coats or seals the bottom and sides of ponds and prevents leaching of nutrients into groundwater. If manure solids separators are used which take out larger fiberous matter, then unseparated manure must be initially pumped into the pond to act as a seal and then the more liquid manure added. Additionally, groundwater levels must be below or equal in depth to pond water levels to prevent the seal from being broken inward and destroyed by greater external pressure. When constructed in medium to fine textured soils, ponds seal and prevent nutrient loss.(3)

3. Pasture management

Land areas receiving manure must be able to absorb nutrients. Pasture soils become compacted and impervious from animal and machinery use. This is a

(3) Discussion with Craig Cogger, Soil Scientist, WSU Cooperative Service, Puyallup

common problem and liquid simply ponds or runs off. These must be loosened and broken up by chisel plowing (8 to 15 inches) or subsoiling (16 to 36 inches). This should be done every four to five years when reseeding pasture. This practice does greatly increase manure absorption and plant use.

4. Manure application timing and rates

Groundwater levels must be below the crop root zone before fertilizer or manure can be safely applied. This generally occurs from June to September. Manure application rates vary from one-half to two inches per acre several times a year to meet crop nutrient needs. "Surplus" nutrients, those that are not absorbed by the soil or not used by the crop, become candidates for potential pollution. Crop nutrient needs are fairly well known through research and if application amounts equal crop needs, little or no nutrients remain to leach into groundwater.

5. Stream buffers

As an ideal recommendation, the establishment of protective corridors bordering streams would be a final practice maximizing nutrient retention and use. Vegetational strips do detain, filter, and absorb surface runoff and minimize erosion and direct animal access to water bodies. This does often require fencing to keep animals away from newly planted grasses and shrubs until their size discourages trampling or eating.

In summary, the installation of gutters to control direct roof runoff, ponds to isolate and store waste, field soil loosening for nutrient absorption, fertilizer application equaling plant needs, and stream buffers to catch and treat surplus runoff, together provide a minimum nutrient loss package of practices for farm water quality management.

Incomplete Answers and Response

Doesn't this nifty little package seem convincing? When all are installed and used, they compliment each other in storing and using the manure nutrients. What are some of the things we are not sure of that need refinement for even better waste control?

1. Manure ponds

Two questions that need answers are:

A. Do ponds effectively seal to prevent groundwater contamination, and

B. What is the change in nutrients from fresh cow manure until the time it is stored in the pond?

Ponds do seal well in sandy loams and finer soils as opposed to more gravelly soils. This sealing can take from several days to over a month. Because gravelly stream soils are found in floodplain areas, and occasionaly the only place to construct a pond, the Cooperative Extension has a test pond of course soil they are partly lining with a patchwork of different fine and clayey soils with probes in each to detect downward loss of nutrients. This should answer the question of at what soil texture should ponds require a separate sealer. The fiber content of manure does act as a mat even in sandy soils and prevent manure seepage.

The nutrient content of ponds has been higher in nitrogen than earlier calculated by the Cooperative Extension. The nitrogen in fresh manure needs to be analyzed and compared to stored manure to more accurately predict changes in content so that farmers will better know nutrient levels when planning ponds.

2. <u>Fertilizer application timing and rates</u>

Although it is generally recommended to apply manure in the fall to empty the ponds, and apply during the growing season, we should be able to better know groundwater levels and when to apply manure to minimize leaching.

Are we giving the farmer accurate enough information on manure nutrients to apply no more than needed to satisfy plant needs? Application rates require more research. We need to sample the nitrogen, potassium, and phosphorus both in the pond and on the ground during application. Being able to factor in nutrient changes for crop application prior to its application could more accurately determine the gallons and inches required to prevent surplus nitrogen entering ground and surface water.

Groundwater contact with the organic horizon in soils is important for bacterial action to occur. This bacterial action breaks down unused nitrogen and incorporates it into the soil. In well or artificially drained soils, if the groundwater is well below this zone, the lack of bacterial action does not alter nitrogen and it remains in a form that could be transported and pollute. If we install drain pipe in pastures to lower water tables are we increasing chances of groundwater pollution? Again, if our nutrient calculations guide farmers in applying only what crops will use, little nutrient losses will occur.

3. <u>Stream buffers</u>

Fencing is generally required to preserve a vegetative corridor adjacent to water bodies. In the case of Blackberry and Douglas Spirea, their density and hardiness form natural barriers. Soil varieties, rainfall amounts, kinds of riparian vegetation, and degree of adjacent animal use determine widths of these buffers. Many dairies confine their cows most, if not all of the year, resulting in the need for narrower buffer widths than areas of intensive grazing. Well drained deep soils with established shrubs and grasses need to be monitored along with the more shallow soils and common scrub areas found next to streams by measuring for nitrates and other nutrients at depth intervals as well as in the stream reaches.

In many Puget Sound Counties, growing numbers of part time farmers are altering a once forested landscape. An impossible question to answer is "How many small landowners does your county have and how much are they impacting water quality? More than any other single occupation, their mere numbers and land clearing will be a major factor in water quality. As nonprofessionals, they often know little about animal waste management and need assistance more than commercial farms. We need to identify their numbers and stream monitor their impacts on water.

Getting Practical Results

For commercial farms, manure ponds are the first, and easiest step to waste control. In an earlier program on the lower Stilliguamish River, some thirty out of fifty farmers installed ponds. However, learning to efficiently apply two to four million gallons of stored manure is a challenge for anyone. These "advanced BMP's require regular farm visits before and during the growing season to reinforce farm planning recommendations and encourage questions about application rates. Few farmers are presently applying manure to fields at optimum rates. Hiring custom pumpers to apply manure from ponds is an improvement over manually hauling it out in spreaders and results in more uniform application.

It will take several years for farmers to understand and apply their waste at levels just to satisfy crop requirements. This is one of the most important things Districts can help farmers understand. Districts are definitely making progress in the use of BMP's, partially because of the economic advantages of installing ponds. The farmer is starting to understand the value of his stored manure.

The Snohomish Conservation District is working with large farms in the Snohomish watershed in planning and application of BMP's. Additionally, the Tulalip Tribes, under a District contract, are monitoring for nitrates, fecal coliform, BOD, suspended sediment, temperature, and discharge. This is being done at thirty two locations on the mainstem and on six tributaries above and below farmed areas to determine upstream contributions and general farm nutrient loading rates. This is carried out twenty times yearly starting May 1987 and ending November 1991. In several months, a years sampling should provide some trends in pollutants.

The District is also assisting small landowners in a small watershed emptying into a lowland farmed area. Around 150 horse and livestock owners are contributing pollutants and sediment to a farmed floodplain. Scaled down versions of BMP's are drafted into a mini-plan for individual owners and they are encouraged to apply these by follow up visits and meetings.

Conservation Districts, with SCS assistance, are one of the few public agencies devoted to personal rural landowner contact and on ground improvements resulting in cleaner water. Most of District work has been with large farms. Several Districts have small landowner projects in specific problem watersheds using much of the same practices to improve small horse and livestock farms. Landowner trust combined with low cost practical solutions place Districts in a unique position to help change land management. Our consistent contact with rural landowners and improvement of planning and application of "BMP's"<u>is</u> our research. The result is cleaner water.

Erosion Control-
What is the state of the art?

Rachel Friedman-Thomas and Gary Minton*

Introduction

Numerous studies have documented the tremendous increase in sediment from urban areas undergoing development. Wolman and Schick (1967) showed sediment yields in urbanizing watersheds to increase from only a few hundred metric tons per square kilometer to tens of thousands of tons. Other studies (Clark, 1985) have documented the economic costs of increased erosion. To reduce the adverse effects of accelerated erosion many local jurisdictions have instituted soil erosion control programs which require developers to prepare site-specific erosion control plans. These plans are implemented during the construction phase of a development.

A variety of best management practices (BMPs) are used in an integrated fashion to control soil erosion. These measures are listed in Table 1 and can be divided into two general groups:

- o Those measures which prevent on-site erosion,
- o Those measures that trap eroded sediments before they leave the site.

Ideally, erosion control measures should be the primary methodology, with trapping techniques as the follow-up, defensive strategy.

Experience to date

Few jurisdictions have terminated previously adopted erosion control programs. Thus it could be concluded that local communities consider the control of soil erosion during construction to be effective. However, there has been very little evaluation of the relative effectiveness of particular BMPs and their integration on particular sites.

Research has been performed on some techniques by state highway departments, and by agricultural and forestry research organizations (Bethlahmy and Kidd, Jr., 1966; O.S.U. Extension Service, 1976; Highway Research Board, 1970). However, little research has been done for urban construction.

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Based on the evaluation of several jurisdictions in the State of Maryland, it was concluded that sediment yields were reduced 60 to 80 percent by construction site erosion control programs (Boysen, 1977). In 1983, staff members from the Municipality of Metropolitan Seattle (Metro) and the Conservation District conducted a survey of the use of BMPs. Of 534 sites visited 26% had BMPs in place, and 10% were suffering water quality problems such as turbid water or heavy sediment leaving the construction site (A. Johnson, 1988).

Conservation District research project

Since the inception of the King County erosion control program, the King County Conservation District has provided technical assistance to county staff. Under a 208 grant the District produced a BMP construction manual that is widely used (KCCD, 1981). However, there has never been an evaluation of how well these recommended measures work for western Washington climatic, slope and soil conditions.

Recently the District has received a grant from the Washington Department of Ecology to evaluate the effectiveness of the techniques listed in Table 1. The project has just recently begun. Presented here is a summary of project objectives, the approach and some preliminary findings.

Methodology

Project goal and objectives

The intent of the project is to collect and summarize empirical data relevant to current conditions and technical experience gained since the inception of the King County ordinance for controlling erosion from urban development. The objectives to achieve the end goal include the evaluation of the effectiveness or ineffectiveness of current practices. Syntheses of experiences of staff from agencies with water quality control programs as well as King County staff members directly involved with erosion control review and inspection will occur. Similarly, the perceptions of the effectiveness issue collected from the development and construction industry, environmental groups, and private citizens affected by urban development will be compiled

Study approach

Development site visits and evaluations, by means of a BMP checklist, will take place during two wet seasons.

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1. The Conservation District is not a department of King County, but a separate and independent special purpose district.

The checklist will address the effectiveness issue by attempting to discern why a particular BMP is failing. Failure could be due to inappropriate use, poor design, improper installation, maintenance or materials.

A synthesis of the community's level of understanding will be accomplished through questionnaires and selected personal interviews. A review of ordinances from local jurisdictions will enhance the survey results.

Complimenting these two study methods will be the creation of a technical advisory committee (TAC) to aid the District in the development of research hypotheses. The group, consisting of local practitioners of erosion control, will provide input for the field evaluation and provide review of the research results.

Results and Discussion

Preliminary field evaluation results from one dozen site visits have shown that of those BMPs that were installed, 40% were installed improperly and 29% were improperly maintained. The distribution of slopes on these dozen sites were, 58% with slopes ranging from 7-15%, 17% with less than 6% slopes, 17% with slopes of 16-30%, and 8% with slopes greater than 30%.

The TAC workshop provided a broad array of input. A large percentage of the group felt that, with few exceptions, BMPs can be expected to achieve the intended objective. Most members agreed that failures are due to either inappropriate use and/or improper maintenance. Most BMPs have been considered to be ineffective when used singly, but have a higher effectiveness when integrated with other measures. An example is the use of seeding, mulching and netting to stabilize a slope versus seeding or netting used individually.

The TAC believed that the specifications for many of the BMPs are too vague and do not reflect the necessity for integration. This lack of clarity may be at the root of the installation and maintenance problems, and certainly reflects the missing emphasis on integration of BMPs.

A final issue discussed by the TAC was the differences in the members definition of "effectiveness" of each control measure. The perception varied with the intended goal of a given organization. The group agreed that the development of definitions was necessary for the success of the project as a whole. Work will continue in this vein.

Expected research products

The Conservation District and Washington Department of Ecology are expecting numerous products from this study. The opinions of professionals from both government agencies and the development community about the effectiveness of various erosion control BMPs will be identified and compiled. The causes for success or failure (inappropriate use, improper design, poor installation or improper maintenance) of erosion control measures will be determined. Conclusions will be drawn concerning the ineffectiveness under all circumstances of all BMPs. These conclusions will be specific to western Washington's climatic, topography and soil conditions. Specifications will be improved to clearly identify when each measure is appropriate/inappropriate, how it should be designed, installed, maintained and the best materials will be suggested. Field information and visual aids will be used to develop more effective erosion control training programs. These programs will be intended for professionals from both government and industry working in this field. Lastly, future research needs will be identified.

Appropriate future research

While it is premature to identify future research needs, the results of the research project to date have raised pertinent issues. The effectiveness of an erosion control program is a direct function of the number and ability of local agency inspectors. Existing local programs should be evaluated to determine the level of resources needed to effectively implement soil erosion control programs. Program guidance materials should be developed to assist local governments in the creation of erosion control programs and in the training of plan reviewers and inspectors.

It is anticipated that even the most effective measures, singly or in combination, cannot reduce suspended sediments to a level that meets Washington Department of Ecology water quality standards. In this light, questions should be raised as to the necessity of meeting those standards. Such questions should investigate the level at which fine colloidal sediment causes environmental damage. Conclusions should be developed from the aforementioned investigations to allow erosion control planners and practitioners the flexibility to utilize different levels of erosion control BMPs as site sensitivity requires.

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Table 1: Erosion control best management practices subgrouped as vegetative or structural controls.

Vegetative Controls	Structural Controls
Buffers	Plastic sheeting
Clearing limits	Filter barriers
Staged clearing	Sediment pond
Mulching	Gravel cone and riser
Seeding/sod	CB sediment trap
Netting	Pea gravel bags
Chemical stabilizers	Earth berms
	Check dams
	Straw bales
	Gradient terrace
	Dispersion structure
	Water conveyance
	Rocked outlet
	Rocked entrance
	Tire wash

Urban Stormwater and Puget Trough Wetlands

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What Do We Know about Urban Stormwater and Wetlands?

Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually near, at, or above the surface, and hydric soils and hydrophytic vegetation predominate (Cowardin et al., 1979). They are recognized as potential providers of numerous ecological and societal functions of value. These functions include hydrologic and water quality regulation, primary production, consumer support, and various social amenities.

With the growing recognition of these functions of wetlands has come increasing interest in both exploiting certain functions for environmental management and protecting overall functional integrity. The greatest interest nationwide in employing wetlands in management has concerned providing advanced treatment for sewage effluents, and a number of relatively small systems are doing so. Now getting more emphasis, particularly in the Northwest, is using freshwater wetlands to route and store urban stormwater for discharge quantity control. Also developing rapidly is interest in combining this purpose with capturing pollutants in nonpoint source runoff. While surface water managers and developers have promoted planned use of these functions of wetlands in developing areas, resource managers have expressed

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concern about the potential effects of such actions on the remaining functions.

While substantial losses have occurred, the Puget Trough retains numerous and diverse wetland resources. Many of these wetlands lie within the zones of current and projected development around Puget Sound cities. Particularly at issue are the widely distributed palustrine wetlands in the uplands of Snohomish, King, Pierce, and Thurston Counties. Some of these wetlands are being considered for designation as stormwater retention/detention ponds, and may be modified morphologically and hydrologically for that purpose. A larger number would receive an altered pattern and quality of runoff, in a more incidental fashion, if their catchments are developed.

In 1986 scientists and managers associated with local, state, and federal agencies that have wetland protection and stormwater management responsibilities in the four-county area joined to consider the need for research on the relationships between urban stormwater and wetlands. This group served and still serves as a technical advisory committee for what became known as the Puget Sound Wetlands and Stormwater Management Research Program. The broad questions initially stated by this group were:

1. What impacts on wetland ecosystems and their functions could be associated with their use for storing urban stormwater?

2. What are the potential water quality benefits to downstream receiving waters of draining urban stormwater through wetlands?

Funded by a Coastal Zone Management Act grant, King County staff performed a literature review concerning these questions. Evaluation of the results of this survey led to the following conclusiors (Stockdale, 1986a, b; Stockdale and Horner, 1987), which generally defined what we knew about urban stormwater and wetlands at the outset of the program:

1. A number of physical, chemical, and biological mechanisms can operate in wetlands to remove and hold pollutants entering with influent water.

2. Wetlands have been investigated extensively as sites for polishing municipal wastewater treatment plant effluents, and the resulting experience should be

generally applicable to stormwater applications. However, most of the investigation has occurred in the northern Midwest and Southeast United States and in Europe.

3. Stormwater applications have been developed much less, relative to sewage treatment, but are now receiving more attention.

4. A large amount of data is available on treatment efficiencies afforded by wetlands (almost all in polishing sewage effluents). These data generally indicate relatively efficient removal and retention for most classes of pollutants, but seasonal nutrient export has been documented.

5. Some techniques have been suggested, but not widely tested, for managing natural wetlands to improve removal and retention of pollutants.

6. Design strategies have been developed for artificial wetlands to serve the treatment function, and substantial experience has been gained with such systems in Europe.

What Do We Need to Know about Urban Stormwater and Wetlands?

In preparation for designing a research program, the literature review also took note of major voids in knowledge concerning wetlands and stormwater, as follows (Stockdale, 1986a, b; Stockdale and Horner, 1987):

1. Very little study has been given to the impacts on overall ecosystem functioning of routing point or nonpoint source effluents through wetlands. Especially neglected have been long-term impacts.

2. The respective roles of contaminants and hydrologic change in affecting wetland ecosystems are unknown.

3. Many specific questions of impact were suggested by the inquiry, especially concerning the transport, fate, and effects of toxicants on the wetland system and on adjacent surface water and groundwater.

4. There is very little Pacific Northwest confirmation of results from elsewhere on the water quality benefits of wetland treatment.

With the literature review complete, it was desired to develop a foundation for the research program by establishing the specific needs of resource and stormwater managers. Accordingly, a formal management needs survey was conducted among the advisory committee membership. Starting with a list of 18 needs suggested by members, a process was applied to rank the alternatives according to agreed-upon criteria. The process used was developed by Mar et al. (1985) and Horner et al. (1986) and involves computerized manipulation of matrices of relative pairwise weightings of alternatives for each criterion, and of the perceived relative importance of the criteria. The result was a ranked list of the alternative management needs. Those ranking highest, which became the basis for research program design, were:

1. Means of assessing short- and long-term impacts of urban stormwater on functioning of the palustrine wetlands potentially most affected by projected development in the Puget Sound region;

2. Criteria for evaluating proposed stormwater discharges by palustrine wetland type;

3. Improved understanding of hydrologic features and functioning of wetlands as a basis for developing management guidelines pertaining to hydrologic change;

4. Improved understanding of factors critical to achieving urban stormwater quality improvement in wetlands and how to manage that process; and

5. Determination of allowable flood storage capacities that avoid impairment of overall wetland functioning.

What Have We Done about These Needs?

General research program design

Responding to the identified needs, a research program consisting of four components, as follows, was designed and implemented:

1. A synoptic survey of the characteristics of palustrine wetlands that have and have not been affected by urban stormwater runoff in the past;

2. A long-term investigation of the functional impacts of urban stormwater discharges;

3. A study of water quality benefits to downstream receiving waters of draining urban stormwater through wetlands; and

4. Laboratory-scale or short-term field experiments to answer specific research quaestions.

The synoptic survey was performed during the spring and summer of 1987 and will be reported in this paper. The remaining tasks are being initiated this year and will be described briefly in the concluding section of the paper.

Synoptic survey methods

Approach and site selection. The survey was intended to compare and contrast wetlands that have been affected by urban runoff for an extended period versus similar systems that drain catchments not developed for urban use, agriculture, or other intensive human activities. Its purposes were to perform pilot work for succeeding studies and to identify appropriate preliminary management guidelines that might be tested in that later work. The survey involved observations and sampling on a single day during the growing season, concentrating on ecosystem components that were expected to exhibit any accumulated effects of urban stormwater.

The survey covered 73 palustrine wetlands in King County, 46 that receive urban runoff and 27 that do not. A greater number of urban sites was selected in order to represent some variety in developed land use. The wetlands represented palustrine open water (POW), emergent (PEM), scrub-shrub (PSS), and forested classes (PFO) (Cowardin et al., 1979). Most of the study sites contained more than one class, and a number had all four. Five of the 73 were bogs, a wetland type considered to be potentially different than other palustrine systems in responding to urban stormwater, due to substantial morphological, chemical, and biological differences.

Observations, sampling, and analyses. A program of observations, sampling and on-site analyses was carried out in all survey wetlands. Samples were taken from 31 of the sites to the Municipality of Metropolitan Seattle Water Quality Laboratory for analysis of certain quantities. This allocation provided a very large data base for statistical analyses on field-measured quantities and a smaller, but still substantial, set of data on the quantities that are more expensive to analyze.

Table 1 summarizes the observations, sampling and analyses carried out in the survey program. Qualitative observations were an important component of the program, and a detailed form was developed to guide the recording of these observations. This portion of the program was designed to provide supplementary detail and information on activiies other than urban runoff that could affect the wetlands, which might aid in interpreting results. Analytical effort concentrated on the soils and nonsoil substrates and plants, which were expected to demonstrate any accumulated effects to a greater degree than more transient elements, such as water. Analysis of water column samples was limited to pH and bacterial indicators. Soil microfauna and aquatic invertebrates were sampled systematically, identified, and enumerated; but only incidental sightings were recorded for other animals.

The Microtox test is used to make quantitative measurements of the relative inhibition in light production by living bioluminescent bacteria when exposed to an environmental sample. The result, a relative index of potential sample toxicity, is expressed in terms of the effective sample concentration (EC) causing a specified bioluminescent output reduction after a given length of exposure. Therefore, the lower the EC, the relatively more toxic the sample.

Soil microbial activity was gaged by counting microfaunal organisms and measuring adenosine triphosphate (ATP) in soil samples and by measuring carbon dioxide evolved by respiring organisms and absorbed by soda lime (sodium hydroxide-slaked lime) in enclosures. Being present in all living cells but decaying quickly upon death, ATP is a general measure of living biomass.

Fecal coliforms are the traditional indicators of the presence of pathogens that may cause human disease. However, there is evidence that enterococci correlate better with the actual occurrence of disease than do fecal coliforms (Dufour, 1984). Still, enterococci can occur in the fecal matter of other animals, making the measure an imperfect indicator of human disease potential also.

Data analysis. Initial data analyses were concerned with determining how wetlands affected by urban runoff differ from unaffected wetlands. Therefore, the significance of differences between the two groups was tested statistically for each set of measurements. A two-sample t-test (Zar, 1984) generally was first performed on each

Table 1. Summary of Observations, Sampling, and Analyses

Category	Subcategory	Measurement	Sampling[a]	Analytical Method[b]
Qualitative Observations		Land use, debris intrusion, water level fluctuation, drainage and filling, sediment movements, visible pollution, odor, organism condition	Observation	
Soils and Nonsoil Substrates	Physical	Texture	15 cm core at 10 locations near inlet and outlet and in all zones	Qualitative field characterization
	Chemical	Organic content	15 cm core at 5 locations near inlet and in all zones	Loss on ignition (APHA 209F)
		pH	Same as organic content	pH-mv meter (EPA 9045)
		Oxidation-reduction potential	Same as organic content	pH-mv meter
		Metals (Pb, Cd, Zn, Cu)	15 cm core at 3 locations near inlet or outlet and in PEM and POW zones	Inductively coupled plasma spectroscopy (EPA 6010)

Table 1 Continued.

		Total phosphorus	Same as metals	Ascorbic acid (APHA 424F)
		Total nitrogen	Same as metals	Kjeldahl digestion (APHA 420A)
	Toxicological	Microtox	Same as metals	Beckman Instruments, Inc. (1982)
	Microbial activity	CO_2 evolution	Enclosures with dry soda lime at 5 locations for 24 hours on 1 or 2 occasions	Gravimetric
		Adenosine triphosphate (ATP)	litter and 5 cm scoop at 5 locations in PEM, PSS, and PFO zones on 1 or 2 occasions	TRIS extraction and ATP photometer (Holm-Hansen and Booth, 1966)
		Microfauna	Same as ATP	Identification & enumeration
Water Column	Chemical	pH	Grab sample in POW zone	pH meter (APHA 423)
	Bacteriological	Fecal coliforms	Grab sample in sterile container in POW zone	Membrane filter (APHA 909C)
		Enterococci	Same as fecal coliforms	Membrane filter (Dufour, 1984)

Table 1 Continued.

Plants	Community composition	Cover-abundance	Octave cover scale in all zones (Gauch, 1982)	
	Tissue analysis	Metals (Pb, Cd, Zn, Cu)	Representative collection of _Juncus effusus_ (soft rush), _Phalaris arundinacea_ (reed canary grass), _Spiraea douglasii_ (hardhack), and _Typha latifolia_ (cattail)	Inductively coupled plasma spectroscopy (EPA 6010)
Animals	Aquatic invertebrates	Population sizes	Time-constrained netting at representative locations in all aquatic habitats	Identification & enumeration
	Other	Sighting	Observation	

[a]Wetland zones include palustrine open water (POW), emergent (PEM), scrub-shrub (PSS), and forested (PFO).
[b]APHA--American Public Health Association (1985); EPA--U.S. Environmental Protection Agency (1986). Number is method number in the reference.

data set to test the null hypothesis that the means of the two populations are the same. However, univariate analysis showed that the data often were not normally distributed and did not have equal variances, thus violating the assumptions of the method. The data were right-skewed, and a logarithmic transformation induced normality in some cases. Otherwise, the Mann-Whitney-Wilcoxon nonparametric test (Zar, 1984) could be used for the hypothesis test. These analyses were performed using SAS software (SAS Institute, Inc., 1985).

Some chemistry data, especially plant tissue metals, had observations that were below detection limits, statistically known as left-censored observations. Likewise, Microtox data occasionally exhibited right-censored observations, meaning the test had to be terminated before the actual value could be realized. In both of these cases, the hypothesis test was performed using the SAS LIFETEST procedure and nonparametric Kalbfleisch and Prentice (1980) logrank test, which were developed for right-censored data in biological "lifetime" tests. The left-censored observations were transformed by multiplying by -1 and adding a large constant.

Each hypothesis test was displayed graphically by plotting the interval of nonsignificant difference (IND) (Conquest, 1986). IND relies on the fact that two means are not significantly different when the distance between them does not exceed:

$$t(s_p^2(1/n_a + 1/n_b))^{1/2} \text{ for equal variances, or}$$

$$t(s_a^2/n_a + s_b^2/n_b)^{1/2} \text{ for unequal variances;}$$

where t = Student's t-distribution for a selected alpha level; s_a^2, s_b^2 = variances of populations A, B; s_p^2 = pooled variance estimate; and n_a, n_b = sample sizes of populations A, B. Unlike a confidence interval, IND is a statistically correct way of displaying the significance of differences, because it is based on the standard error of the difference between the two means being compared. If the mean of one population falls within the IND of the other, the two populations are not significantly different for the selected significance level (alpha), and vice versa. The basic significance level used in the tests was 0.05, although 0.10 was also investigated.

Aquatic invertebrate data were analyzed using several different procedures. A stepwise discriminant analysis (STEPDISC in SAS) was employed on a log(x +1) transformation of count data to identify the taxa (adults and larvae) that best discriminate wetlands affected and unaffected by urban runoff (Morrison, 1976). This analysis was taken another step, using the SAS DISCRIM routine, to compute a scoring function for each wetland and an _a posteriori_ probability of membership in either the affected or unaffected group. At the end, the rate of correct classification of wetlands can be computed. This analysis is useful to analyze whether a classification model can be developed, but it must be recognized that there is a bias in favor of correct classification when the same data set is used both to create the scoring functions and to test their ability to classify.

In addition to these discriminant analyses, Shannon-Weaver diversity indices (Shannon and Weaver, 1963) were computed for the adults and larvae collected in each wetland. Mean indices for affected and unaffected wetlands were tested for difference as described previously.

Plant cover-abundance data were analyzed with the use of TWINSPAN, an ordination technique whose acronym is short for two-way indicator species analysis (Gauch, 1982). TWINSPAN is a multivariate method that first classifies the vegetation stands sampled and then obtains a second classification of species within those stands according to ecological preferences. The result is both an ordering of stands along an environmental gradient and an ordering of species in the stands along the same gradient. The identification of trends in the large, complex data set in this way allows determination of whether plant community characteristics distinguish affected and unaffected wetlands. A basic assumption of vegetation ecology is that plant communities reflect underlying environmental conditions; if environmental differences exist between the two groups of wetlands, they should be revealed at some level of the classification by a separation of the stands from the respective groups.

Additional data analyses remain to be completed. They will include multivariate comparisons between the affected and unaffected wetlands, designed to determine how different variables rank in separating the groups, and analyses of correlation between variables. To date only one such analysis has been completed, a bivariate correlation involving fecal coliforms and enterococci (Zar, 1984).

Synoptic survey results

General. Hypothesis test results almost always agreed, whether performed on untransformed or log-transformed data or according to the parametric t-test or the nonparametric Mann-Whitney-Wilcoxon test. This uniformity adds assurance to the conclusiveness of results. It also validated the display of results according to the IND technique, despite the violation of assumptions of the underlying t-test procedure. For these graphs, intervals calculated on the basis of log-transformed data were transformed back to the linear scale for display purposes.

Soils and nonsoil substrates. Among the physical and chemical characteristics tested in wetland soils and nonsoil substrates, the following exhibited no significant differences in any zone at alpha = 0.05 (with only one isolated exception at 0.10):

- Texture
- Organic content
- pH
- Total phosphorus
- Total nitrogen

Figure 1 displays the intervals of nonsignificant difference for oxidation-reduction (redox) potential. While the mean redox potentials in wetlands affected by urban runoff were generally higher than in the unaffected sites, meaning the soils were more highly oxygenated in the affected cases, the difference was significant at alpha = 0.05 only near the inlet. As would be expected, mean redox potentials were lowest (< 100 mv) in the deepest water (POW zones). They were below the level at which oxygen is generally fully depleted (approximately 250 mv) in the inlet, open water, and emergent zones.

Figure 2 presents IND plots for the metals in soils and nonsoil substrates. The inlet and emergent (PEM) zones exhibited significantly higher Pb. If the 0.10 alpha level is used, Cd and Zn differences would also become significant in the PEM zone. If the comparisons are based on soil wet weights instead of dry weights, recognizing that the wetland soils are usually saturated, some additional significant differences appear. Regardless of the basis, copper exhibited no significant differences in any zone, and no means differed significantly in the scrub-shrub zone (PSS).

Figure 3 displays IND for the Microtox test conducted for two lengths of time. Affected wetland open water zone soils had significantly lower EC's at alpha = 0.10 in

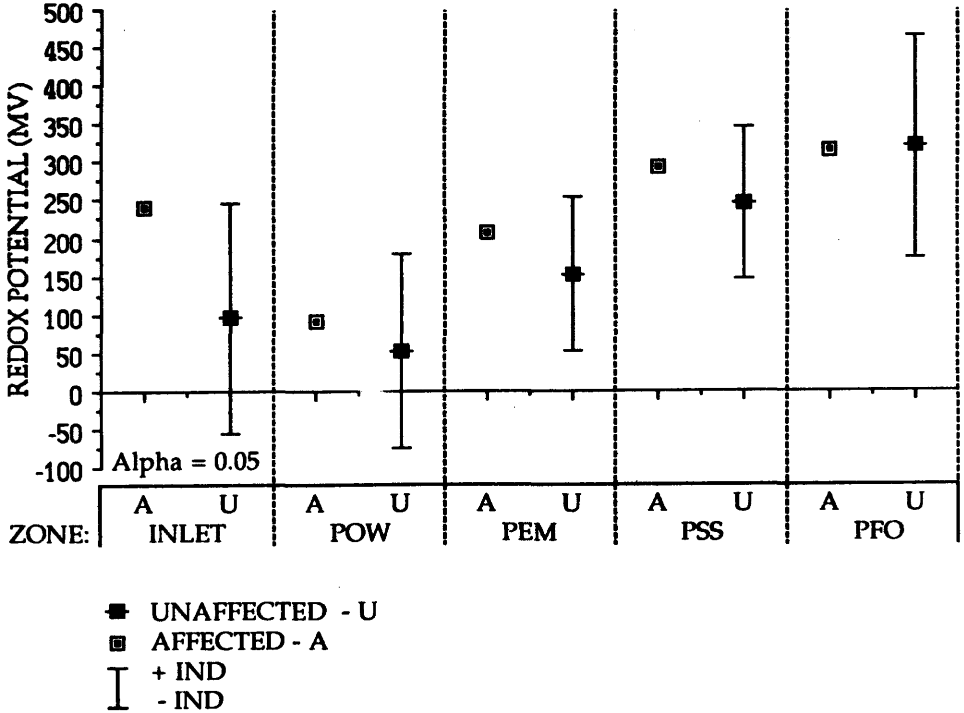

Figure 1. Intervals of Nonsignificant Difference (IND) for Oxidation-Reduction (Redox) Potential

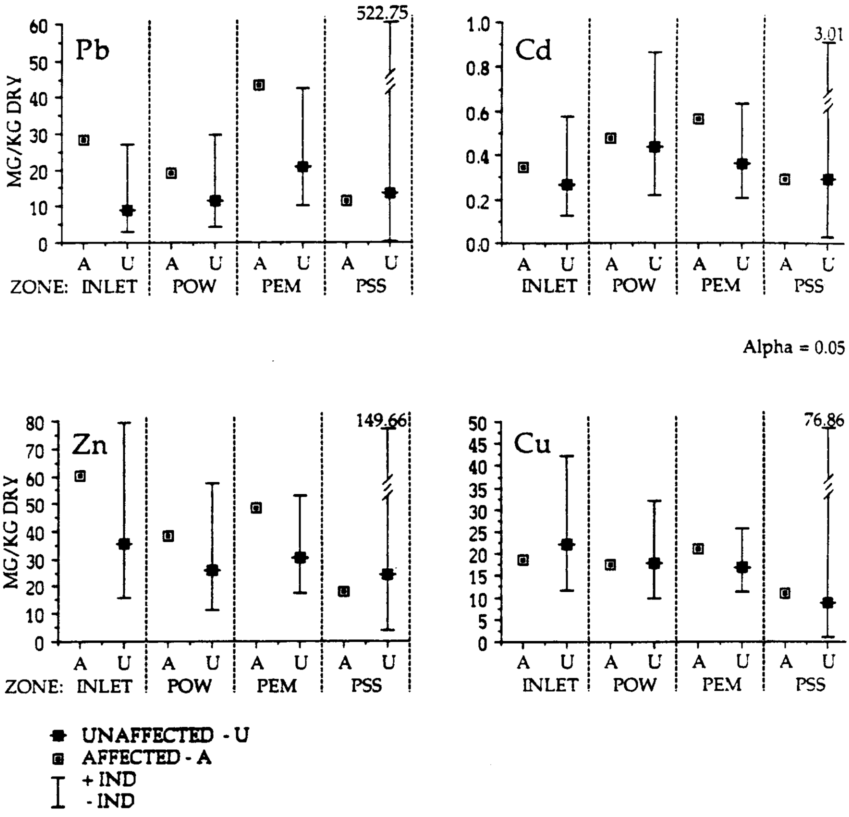

Figure 2. Intervals of Nonsignificant Difference (IND) for Four Metals

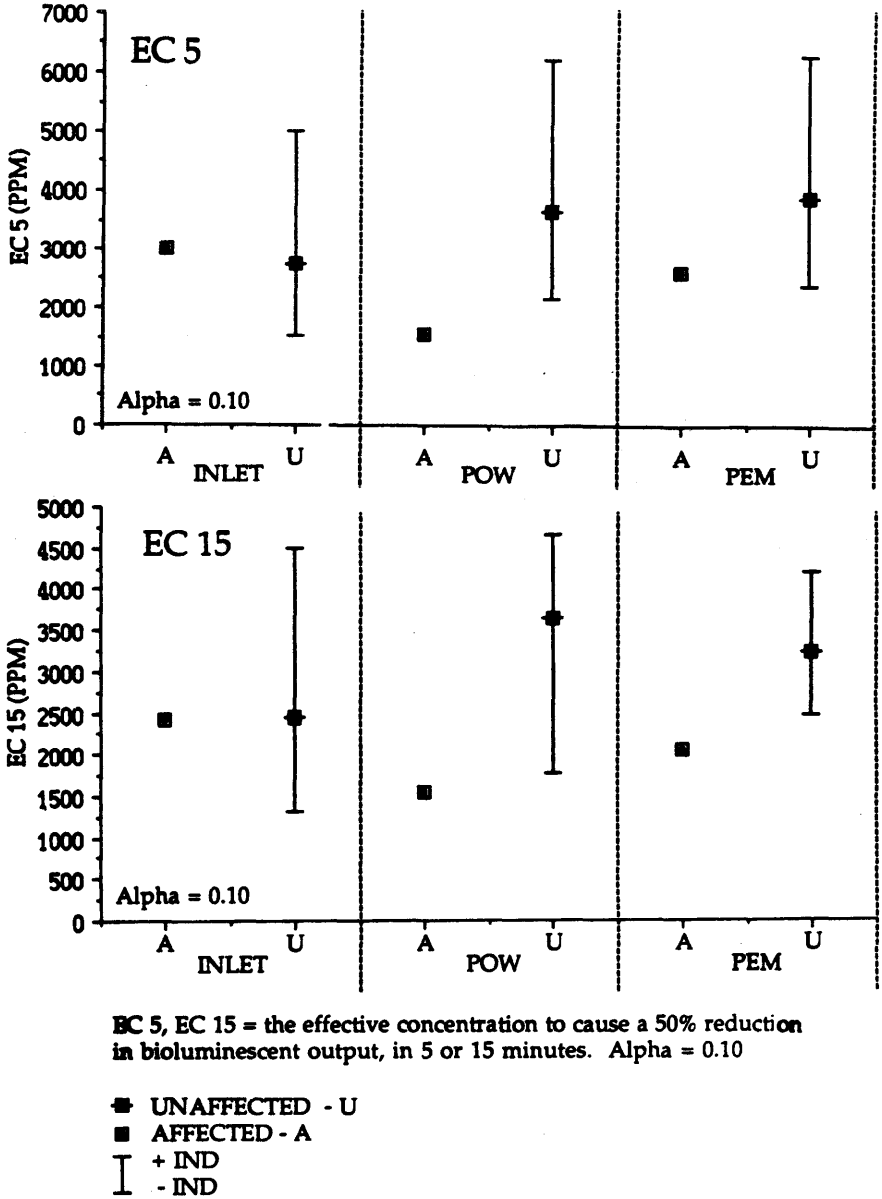

Figure 3. Intervals of Nonsignificant Difference (IND) for the Microtox Test Run for Two Lengths of Time

both tests (therefore, are potentially more toxic). The difference was also significant in the emergent zone in the longer test. Inlet and scrub-shrub (not graphed) zones exhibited no significant differences.

Among the three indicators of soil microbial activity tested (ATP, carbon dioxide evolution, and microfauna counts), only mean carbon dioxide production in emergent zones on one of two sampling occasions differed significantly. There is no apparent biological reason for this singular result.

Therefore, the general physical, chemical, and microbiological nature of the substrates in wetlands affected and unaffected by urban runoff did not appear to differ in many respects. However, a number of instances of significantly elevated metal concentrations were noted. These differences contrast with the results of comparing soil nutrients, where no significant differences occurred. Nutrients cycle relatively rapidly in wetland soils, while metals tend to be more cumulative over time (Horner, 1986), a tendency that seems to be evident in the wetlands surveyed. The presence of elevated metals could explain the potential toxicity sometimes noted in Microtox results. However, the associations are not entirely consistent among the zones; and further data analyses will be needed to document them more fully.

Water column. Water column analyses were limited because of the belief that more permanent components of the systems would better reveal differences in a survey. Mean pH was very significantly ($P = 0.004$) higher (6.72) in the affected compared to the unaffected (6.32) wetlands.

Figure 4 depicts IND for the water column bacteriolgical data. Means of both fecal coliforms and enterococci were significantly higher in the affected cases. However, the fecal coliform mean was below the Washington State standard of 200 organisms/100 ml, and the enterococci mean was below the proposed EPA criterion of 33/100 ml (both geometric means). Log-transformed fecal coliforms and enterococci were significantly correlated (Pearson correlation coefficient = 0.73, nonparametric Spearman correlation coefficient = 0.65).

Review of the data set revealed that seven of the 19 affected wetlands tested had fecal coliform counts greater than 200/100 ml (five also had enterococci above 33/100 ml). Four had coliform counts above 400/100 ml,

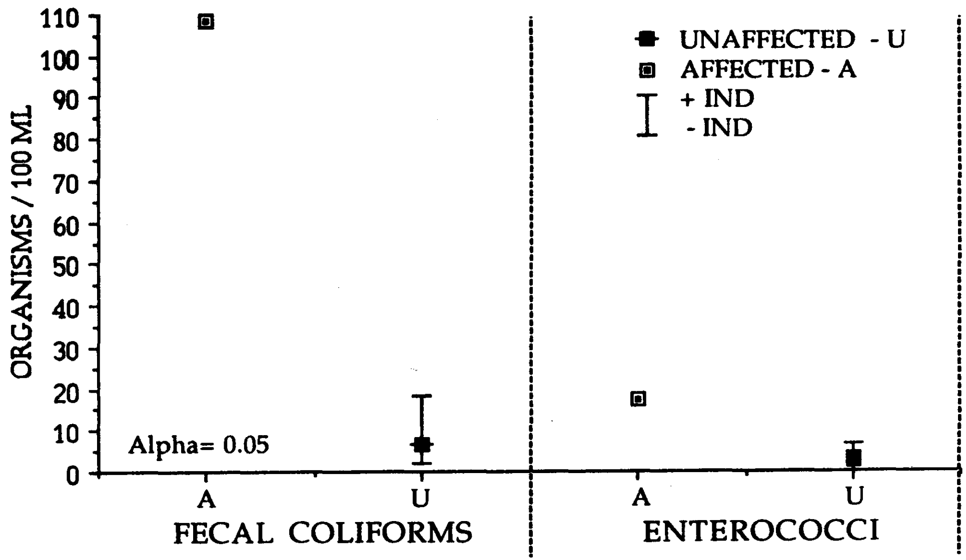

Figure 4. Intervals of Nonsignificant Difference (IND) for Fecal Coliforms and Enterococci

with a high of 4633/100 ml (enterococci ranged 80-230, the highest values measured). The watersheds of those in the latter group all had high density residential or commercial development or both and substantial signs of human intrusion (debris, heavy foot travel). Development in the other three watersheds was primarily low-density residential. The 12 urban-affected wetlands that had fecal coliforms below 200/100 ml also had chiefly low-density residential development, although some high-density residential land use was in evidence. However, this group had no commercial development.

Plants. Plant cover-abundance was analyzed according to the ordination technique by wetland zones. In the open water there was a perceptible separation based on more extensive presence of *Nuphar polysepalum*, a water lily in the wetlands unaffected by urban runoff. This floating-leaf plant grows in relatively deep water, and thus suggests that unaffected wetlands have deeper open water zones. In the emergent zone the clearest stand separation was *Phalaris arundinacea* (reed canary grass), a well known opportunist, in affected wetlands. Vegetation types in the scrub-shrub zone were well separated, but associations with affected/unaffected status were not apparent. In the case of the forested zone, results showed good division between riparian and upland forest, with most of the unaffected stands falling into the upland category. These stands are dominated by *Tsuga heterophylla* (western hemlock), *Thuja placata* (western red cedar), and *Pseudotsuga menziesii* (Douglas fir). To derive further conclusions, additional work needs to be done on this data set to remove "single-occurrence stands" that seem to be obscuring some trends.

Comparison of mean metal concentrations in tissues of four plant species found in affected and unaffected wetlands exhibited no statistically significant differences at $P = 0.10$. However, a number of values were censored, especially for Pb, and a proper hypothesis test could not be conducted in several cases. In future work a more sensitive analytical technique will be used to avoid this problem.

Aquatic invertebrates. Table 2 lists the taxa that explain most of the variation between wetlands affected and unaffected by urban runoff for the adult and larval stages. Too few taxa are represented to draw any general conclusions about the association between taxonomic groups and affected versus unaffected status. Using these lists the discriminant analysis procedure was able to classify wetlands in the proper group 100 percent of

Table 2. Aquatic Invertebrate Taxa Selected by Stepwise Discriminant Analysis (Listed in Order) to Distinguish Wetlands Affected and Unaffected by Urban Stormwater Runoff

Class	Order	Family	Genus	More Prevalent In
Adults				
Crustacea	Isopoda	Asellidae		Affected
Monogononta	Ploima	Bracionidae		Affected
Crustacea	Cladocera	Daphnidae		Unaffected
Insecta	Trichoptera	Limnephilidae	Dicosmoecus	Affected
Crustacea	Amphipoda	Hyalellidae		Unaffected
Insecta	Coleoptera	Hydrophilidae		Affected
Insecta	Odonata	Libellulidae		Unaffected
Oligochaeta	Prosopora	Lumbriculidae		Affected
Pelecypoda		Sphaeriidae		Unaffected
Insecta	Diptera	Tipulidae		Unaffected
Insecta	Hempitera	Nephidae		Unaffected
Turbellaria	Tricladida	Planaridae		Affected
Larva				
Insecta	Trichoptera		Anabolia	Unaffected
Insecta	Diptera	Chironomidae		Unaffected
Insecta	Trichoptera	Limnephilidae	Dicosmoecus	Affected
Insecta	Hemiptera	Gerridae		Affected
Insecta	Ephemeroptera	Heptageniidae		Unaffected
Insecta	Odonata	Lestidae		Unaffected
Insecta	Hemiptera	Miridae		Affected
Insecta	Trichoptera		Psychoglypha	Affected

the time in the case of adults and with 94 percent success for the larvae. Therefore, a discriminating model has been calibrated. However, the chance for bias introduced by using the same data to develop and to test the model requires verification with an independent data set.

Mean Shannon-Weaver diversity indices for adults were 0.74 and 0.62 in affected and unaffected wetlands, respectively. The equivalent indices for larvae were 0.53 and 0.64, respectively. Neither difference was statistically significant.

Synoptic survey conclusions

The wetlands generally exhibited substantial within-group variability. This dispersion detracted from the ability to demonstrate differences statistically. To compensate, at least two possible significance levels were always investigated. When the null hypothesis was accepted, the P value was usually well above 0.10. Therefore, there is a fairly high degree of statistical confidence in the hypothesis test results.

The strongest distinctions between wetlands that have and have not received urban runoff over extended periods lie in the areas of soil metal concentrations, the potential toxicity of the soil material, and bacteriological indicators. Only a few management-related guidelines of a tentative nature can be developed on the basis of these results. The elevated metal concentrations near the inlet of affected wetlands suggest that wetlands can be protected from possible effects of accumulating metals by removing them from urban runoff before it enters. It is known that metals in urban runoff are primarily in the solid state (Stockdale, 1986a). Therefore, effective preliminary solids settling would offer some protection, and settling ponds ahead of wetlands that receive urban stormwater discharge are advisable.

Urban wetlands were found to have much higher bacterial counts than their unaffected counterparts, especially when they drain relatively intensively developed catchments. However, in the majority of cases these counts did not, at least in a single measurement, exceed existing or proposed standards. Moreover, humans rarely use wetlands for contact recreation or as a drinking water source. Therefore, the significance of this result depends mainly on the release of bacteria from the wetland and the use of downstream waters. This survey was unable to document transport, but that appears to be a fruitful subject for succeeding work. If wetlands are effective traps for bacteria in urban runoff, they could serve a significant role in reducing transport of those organisms to waters where they have a considerable impact (e. g., shellfish beds).

Further work is needed on the plant cover data, but documentation of domination of some affected wetlands by an opportunistic species that tends to establish monocultures suggests an area for monitoring attention. It is hypothesized that succession to such patterns is caused more by modified hydrology than by the delivery of pollutants. This is a hypothesis that could not be tested in the one-time surveys, but will receive definite attention in the long-term monitoring program.

What Will We Be Doing in the Future?

In 1988 we are beginning work on all three remaining components of the program: the long-term study of urban runoff impacts, the investigation of water quality benefits that wetland treatment might provide, and two laboratory efforts. The long-term study will be carried out in 16 or more wetlands, half that never become affected by urban runoff and half that become so affected after a period during which baseline monitoring will be done. The study will proceed for five years, although not necessarily five successive years. In one or more later years, annual coverage may be held in abeyance in order to ensure that resources are sufficient to document any slowly developing long-term effects. Monitoring will encompass:

Hydrology--water level fluctuation

Water quality--water column nutrients, metals, bacteria, and other constituents on several occasions throughout the year (emphasizing the wettest and driest periods)

Soils and nonsoil substrates--physical structure, organic content, redox potential, metals, nutrients, and litter surveys for microbial activity once each year

Plants--cover-abundance, productivity, and tissue metals concentrations once each year

Animals--aquatic invertebrate identification and enumeration, bird censusing, mammal and herpetofauna observations

In the water quality benefit study, influent and effluent stations are being placed in two wetlands, one that receives urban runoff and one that does not. Continuous flow gaging will be performed at these stations, and composite water samples will be taken during approximately 10 storms and during dry weather periods as well. Each wetland is being outfitted with a precipitation station to provide both quantity measurements and precipitation samples for analysis. In order to study the participation of groundwater in the hydrologic systems of these wetlands, a series of shallow piezometers will be placed around them in order to measure the position of the groundwater surface and sample for water quality analyses. Analyses of these various samples will be similar to those listed for the long-term program.

The laboratory investigations that are being started in 1988 are: (1) hydrologic manipulation of wetland plant and soil microcosms, and (2) exploration of stress protein response (SPR) techniques for determining the levels and causes of stress in wetland organisms. The first study directly addresses the hypothesis stated above concerning the influence of hydrology on plant community development. SPR techniques are an outgrowth of biomedical research that take advantage of the fact that the cells of organisms under stress produce types and amounts of proteins different than unstressed cells. These proteins can be identified and measured by electrophoresis techniques to determine the extent of stress and what its source may be.

It is expected that this comprehensive program of studies will yield not only the desired guidance to manage wetlands that are under urban influence, but also substantial information to expand the state of knowledge of overall wetland functioning.

Acknowledgments

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SOURCES OF TOXICANTS IN STORM DRAINS, CONTROL MEASURES AND REMEDIAL ACTIONS

Tom Hubbard and Tim Sample*

Nonpoint sources of toxic chemicals are often difficult to track down and even more difficult to control. Metro's experience with source tracing and control in heavily-contaminated storm drains in the lower Duwamish River in Seattle can serve as a model for approaching this complex and challenging issue. Applied research is needed to develop other techniques of source identification and appropriate best management practices for commercial and industrial facilities.

I. What We Know

Receiving environment chemistry, storm drainage maps, land use information, and sampling of sediments in storm drains can identify significant sources of lead, PAHs, and PCBs from past and present practices of commercial and industrial facilities. After sources have been controlled, contaminated sediments can be removed from the drains to prevent further environmental degradation.

The following examples illustrate the steps used in two storm drains in the Duwamish River:

A. Lead in the SW Lander St. Storm Drain. Studies in the late 1970s and early 1980s found elevated concentrations of lead in the water, sediment, and biota of the lower Duwamish River. In the water column, lead exceeded EPA's chronic criterion for marine life during most of the year. In sediments of the river, concentrations of lead were 22 times greater than Puget Sound background conditions. Duwamish mussels had 30 times and seagulls had 9 times more lead in their tissues than animals from rural areas of Puget Sound (1).

Mass balances, comparing existing conditions and known sources, showed that 86% of the lead in the river could not be attributed to permitted

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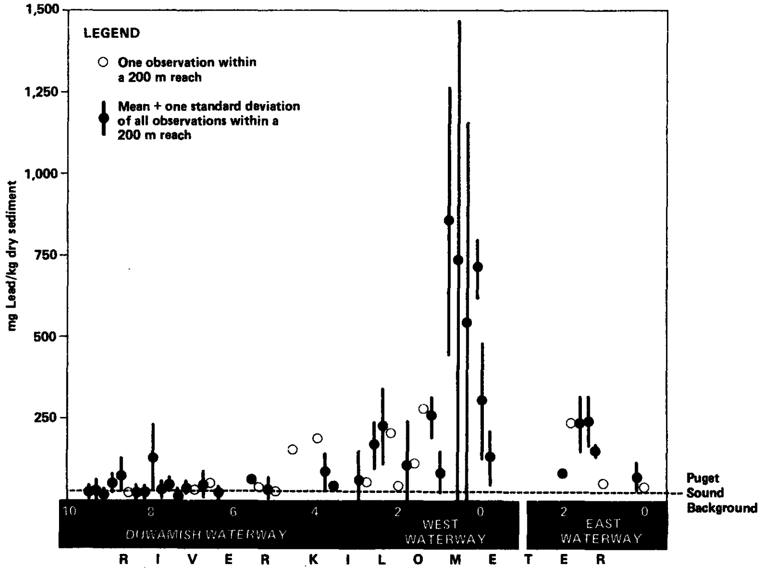
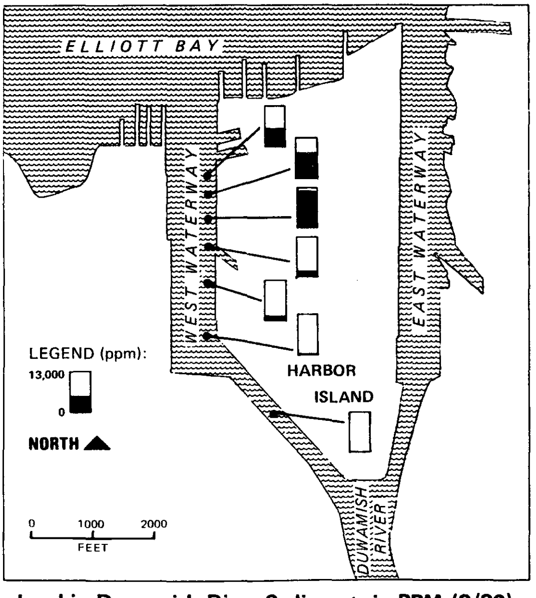


Longitudinal Variation in Sediment Lead Concentration in the Duwamish Estuary, 1972 - 1982 (n=133 observations)

(from Harper-Owes 1983)

FIGURE 1



Lead in Duwamish River Sediments in PPM (3/80)

FIGURE 2

sources. Because high concentrations were found in the water column and the top 2 cm of the sediment, ongoing sources were suspected (1).

Sediment chemistry from several sampling efforts were plotted on a map of the river (Figure 1). Concentrations of most toxicants were not evenly distributed throughout the river. There was a definite peak in the West Waterway 200 - 400m upstream from the mouth of the river where concentrations averaged 750 ppm, the highest in the entire system (1). Analysis of sediment data from the West Waterway (Figure 2) identified a probable source on the east bank of the river. Concentrations showed a definite gradient: those closer to the SW Lander St. storm drain were higher and immediately below the drain's outfall, lead concentrations reached 12,990 ppm (2).

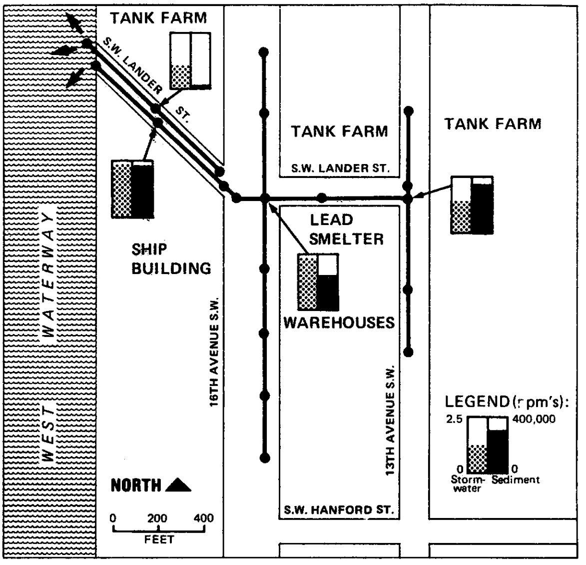
A review of City of Seattle storm drainage maps and land use information of the area identified a smelter which recovered lead from crushed batteries. The smelter was a block from the river in an area drained by the SW Lander St. storm drain. There are two storm drains with outfalls at SW Lander St. One is a small, privately-owned drain, which services an oil storage facility, and the other is a larger system owned by the City of Seattle, which drains approximately two square blocks of the central part of Harbor Island including the lead smelter. Other land uses in the drainage area are petroleum tank farms, small warehouses and ship repair facilities.

Data collected by the Puget Sound Air Pollution Control Agency (PSAPCA) had found 182,000 ppm (18%) lead in soil samples collected on unpaved parking lots around the smelter (4). Fugitive dust from the smelter was suspected as a major source of lead contamination in the SW Lander St. storm drain and the Duwamish River.

Sediment and stormwater from both storm drainage systems were collected at four manholes and analyzed for heavy metals. Samples were analyzed according to EPA methods for analyzing wastewater and solid wastes (5,6).

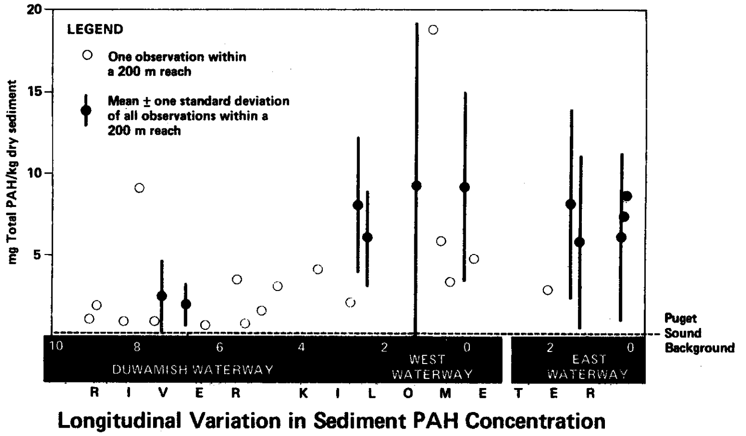
Water and sediment lead concentrations were much higher in the municipal system (Figure 3). Composite stormwater samples had 1.2 - 2.3 ppm lead, ten times more than stormwater in Bellevue (4), and much greater than the EPA acute criterion for marine life (0.14 ppm.). Sediment concentrations of lead were 247,000 - 358,000 ppm (25 - 36% lead dry weight). Typical street dust in the Duwamish basin had 460 ppm or 0.04% lead (4).

Samples of sediment from catch basins on the smelter site were not collected, but a Metro site visit team inspected the smelter. Dark red sediment, similar in color to the smelter's bag house dust, was observed all over the site and in catch basins on the facility. The smelter's



Lead in S.W. Lander Street: Stormwater and Sediment (3/84)

FIGURE 3



Longitudinal Variation in Sediment PAH Concentration in the Duwamish Estuary, 1978 - 1982 (n=40 observations)

(from Harper-Owes 1983)

FIGURE 4

management and the Metro team agreed that additional storm drain samples were unnecessary.

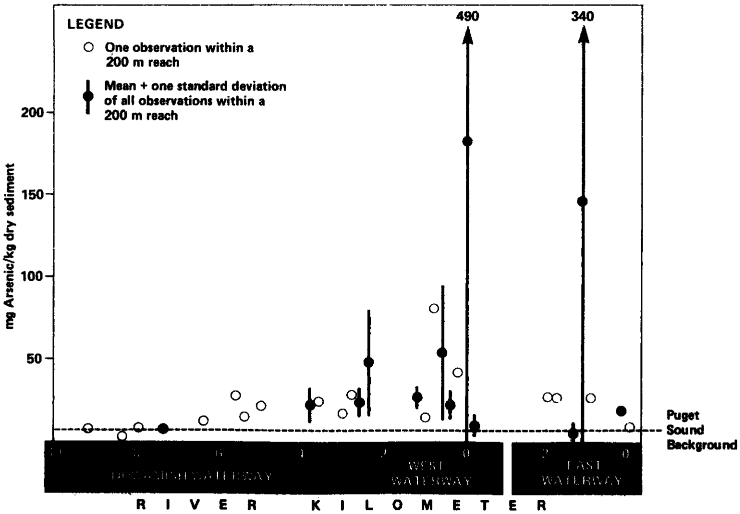
Fugitive dust controls, mandated by PSAPCA, reduced lead emissions from 4 ug/m^3 in 1974 to less than 1 ug/m^3 in 1981. In 1983, the City of Seattle paved parking lots in the area around the smelter to reduce concentrations of lead in the air. In the same year, the smelter was closed permanently.

With the closure of the smelter and paving the parking lots, the major source of lead in this area had been eliminated. A cooperative cleanup effort was initiated to remove the contaminated sediment from the drain before it could be carried to the river during winter storms.

City of Seattle Sewer Utility crews constructed a temporary sandbag weir in the drain at the last manhole before the river. Using high pressure hoses, the drain was flushed. Most of the sediment was trapped behind the weir, and the wash water plus suspended sediment was discharged to Metro's sanitary sewers via the smelter's pretreatment system. Twenty cubic yards of sediment were collected from the drain. The sediment was shipped to a smelter in Oregon and it was smelted to recover the lead.

The SW Lander St. monitoring had shown that sediment chemistry was effective in identifying sources of pollution, and sediment grabs were easier than composite stormwater samples to collect -- especially in drains which are tidally-influenced. In this system, there was a single source which had recently ceased operations, and the sediment could be reused, and therefore not designated as a hazardous waste.

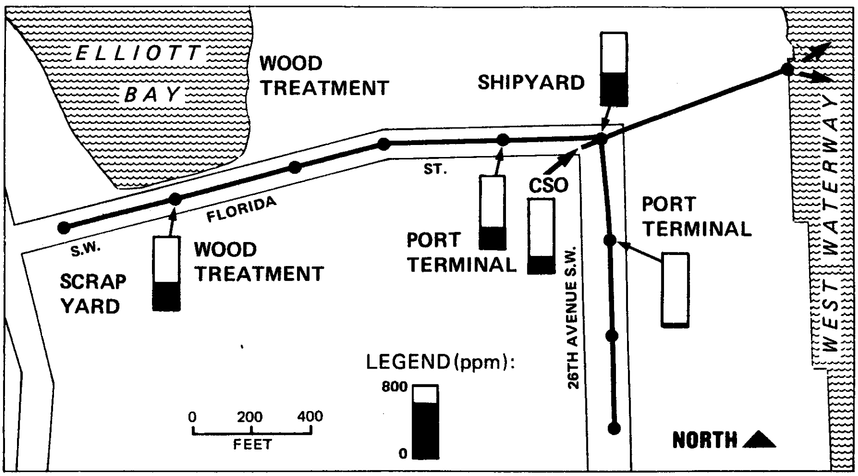
B. Metals and Toxic Organics in the SW Florida St. Drain. Across the river at SW Florida St., there were multiple sources, continuous operations, hazardous waste concerns and an EPA criminal investigation underway -- a situation very different from the relatively straightforward SW Lander St. investigation. Similar to lead, elevated concentrations of other heavy metals and organic toxicants were found in Duwamish water, sediment and biota. Plotting of polycyclic aromatic hydrocarbons (PAHs) in bottom sediments (Figure 4) showed that the West Waterway had some of the highest concentrations of PAHs and heavy metals (copper, lead, arsenic and zinc) in the entire river (Figure 5). It was also one of the few places where polychlorinated biphenyls (PCBs) were found in detectable concentrations. Most of the PCBs were Aroclor 1260. A review of the data revealed definite gradients of concentrations indicating probable sources on the west bank of the West Waterway. Again there were no known point sources in the immediate vicinity of the largest concentrations, but a city-owned storm drain discharges into the river at SW Florida St.



Longitudinal Variation in Sediment Arsenic Concentration in the Duwamish Estuary, 1972 - 1982 (n=54 observations)

(from Harper-Owes 1983)

FIGURE 5



Arsenic in S.W. Florida Street: Storm Drain Sediment (4/84)

FIGURE 6

By consulting city drainage maps and land use information, possible sources of toxicants were identified. A wood treatment facility on the west bank of the West Waterway was suspected as a source of PAHs and heavy metals because creosote and copper arsenate are used in wood treatment. The drainage area also had a major shipyard, port facilities and a scrap metal recycler. The SW Florida St. system has two major trunks and also receives effluent from a small combined sewer overflow (CSO), which overflows less than once a year. Based on the SW Lander St. experience, stormwater samples were not collected. Grab samples of sediment were collected from four storm drain manholes and the CSO. One station was placed upstream of the wood treatment facility and the other was downstream. The samples were analyzed for heavy metals and priority pollutant organics.

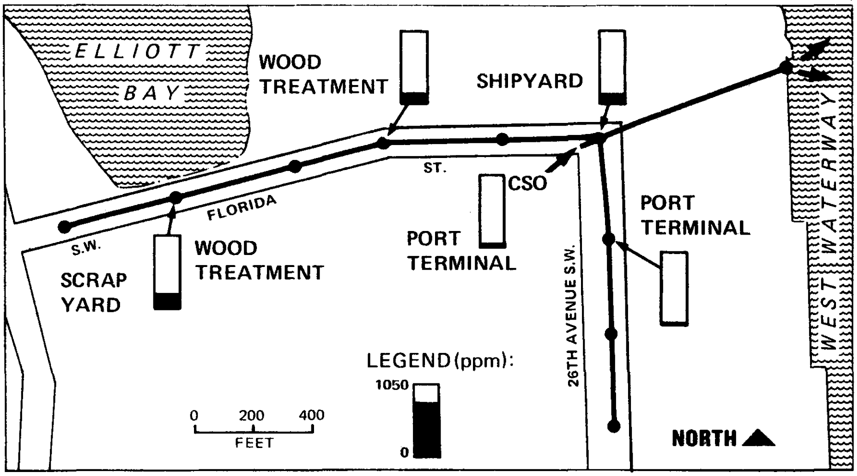
The sediment in the CSO and trunk under 26th St. SW had very low concentrations of metals and organics, but the SW Florida St. line had 57 - 161 ppm total PAHs and high concentrations of copper and arsenic, especially around the wood treatment facility (Figure 6). High concentrations of PCBs, exceeding EPA's hazardous waste designation (50 ppm), were found, and the gradient of concentrations indicated a source upstream of the wood treatment facility. Similar to river sediments near the mouth of the drain, most of the PCBs were Aroclor 1260, which was primarily used in electrical transformers.

To confirm the high PCB concentrations and to collect sediment from the wood treatment facility's catch basins, the SW Florida St. line was resampled. Based on the data from the previous sampling, no sediment was collected from the CSO and the 26th SW line. Sediment was collected at every manhole of the SW Florida St. portion of the storm drain system (Figure 6). High concentrations of PAHs, copper, and arsenic were found in the wood treatment facility, and the PCB data pointed to a source at the head of the line where the scrap yard's stormwater discharged to the system.

Because of EPA criminal investigations of the wood treatment facility, that site was not visited, but discussions with the scrap yard confirmed that they handled electrical transformers from several utilities and waste oil had been spilled on their site.

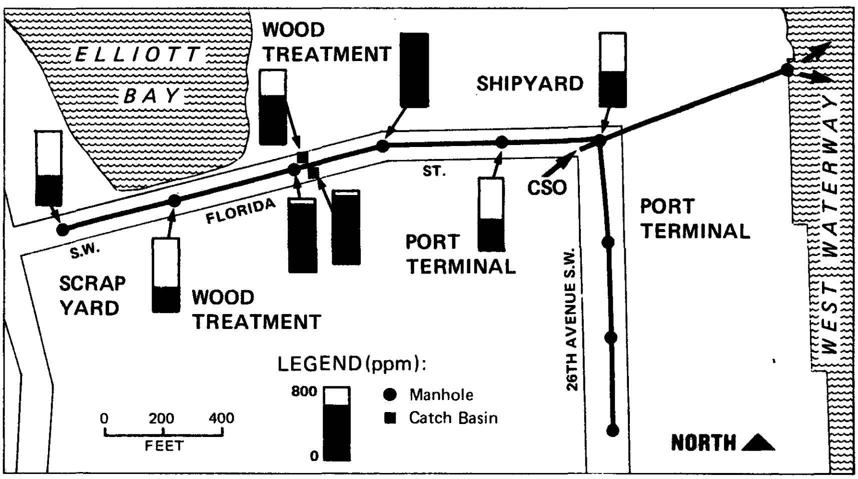
The wood treater pleaded guilty to criminal charges under the federal Resource Conservation and Recovery Act (RCRA) and Clean Water Act and agreed to cease illegal discharges of wastewater to the storm drain. The scrap yard had ceased handling PCB transformers.

Cleanup of this drain was much more complicated because two industries were involved. The sediment was probably hazardous waste and not reusable, and city engineering department crews were not experienced in handling hazardous materials. Furthermore, there was no convenient pretreatment system for disposal of wastewater generated by the cleanup.



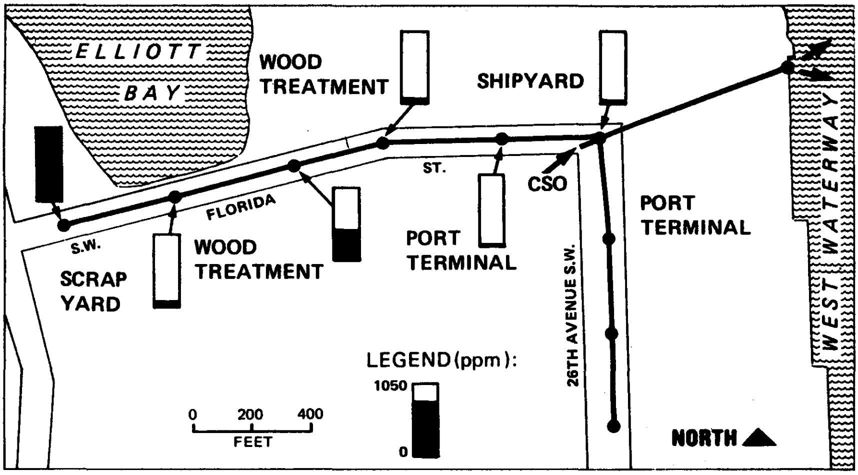
PCBs (total) in S.W. Florida Street: Storm Drain Sediment (4/84)

FIGURE 7



Arsenic in S.W. Florida Street: Storm Drain Sediment (10/84)

FIGURE 8



PCBs (total) in S.W. Florida Street: Storm Drain Sediment (10/84)

FIGURE 9

City crews installed a drag machine (buckets pulled by cables), which scraped the oily sediment out of the line, and vactor trucks were used to clean the catch basins. High pressure hoses were used to dislodge the remainder. An improvised settling pond was constructed to pretreat the water before it was discharged to the sanitary sewer. The sediment was held in lagoons and later disposed in a sanitary landfill. The responsible industries agreed to split the costs of disposal.

II. What We Don't Know / What We Need To Know

The methods described above were successful in a few drainage systems, but other methods are needed to trace and control the diverse sources of toxicants that can enter storm drains. Applied research is needed to develop additional, best management practices and source controls for industrial and commercial facilities.

1. What do you do when there is no sediment in the pipe?

In high-volume, high-energy systems (e.g. steep gradients), sediment may not accumulate in the pipes. Sediments accumulating at the mouth of the drain may be the best sampling site in such situations.

2. Are sediments effective in identifying intermittent sources?

Some sources of toxicants may not be as obvious as the lead smelter, the wood treatment company or the scrap yard; or their discharges may be periodic. Some toxicants, such as volatile organic solvents and dissolved toxicants, may not bind to particulates and accumulate in the sediments. Other techniques are needed to detect those sources. Responding to complaints and reports of oil and chemical spills identified sources of toxicants to the Duwamish River, which were not detected in the storm drain sediment monitoring program.

3. What do you do in large systems with multiple sources?

Separating the effects of multiple dischargers may be difficult without extensive monitoring of drains, catch basins and industrial processes.

4. What are the source controls/best management practices/housekeeping measures that can be recommended to businesses and industries?

   How can mangers and workers be educated on the fate of pollutants dumped into storm drains?

Some recommendations such as paving, covering and berming oil storage areas and installing closed cooling systems have been developed, but many times the business/industry is faced with difficult source control problems. Many businesses do not know what to tell their employees about stormwater and storage, handling, and disposal of materials which could pollute the air, land or water.

5. How often should oil-water separators and catch basins be cleaned?

Without regular maintenance, separators and catch basins cease to collect oil and sediment. The frequency and timing of routine cleaning, necessary to ensure proper operation is unknown.

6. What can be done about "generic" runoff from large facilities or commercial/industrial areas?

Many commercial/industrial facilities are surrounded by large parking lots and impervious surfaces. Similar to areas used for industrial processes, stormwater from parking lots may be a significant source of pollutants to the receiving environment. More research is needed on the design and operation of detention/retention basins and grassy swales.

7. What can be done about contaminated sediments once discharged to the receiving environment? Capping? Dredging?

Sources can be controlled and contaminated sediments can be removed from drains, but techniques for removal or containment of contaminated sediments in the receiving environment are poorly understood.

<u>Summary</u>

1. The distribution of toxicant concentrations in receiving environment sediments can indicate likely areas for possible source tracing.

2. Land use information and storm drainage maps can assist in designing monitoring programs to identify sources of toxicants.

3. Storm drain sediments can be effective integrators for tracing sources. A few samples of sediments in major tributaries can focus intensive monitoring and site visits and are much more efficient than stormwater samples collected during storm events.

4. Stormwater can transport toxicants from commercial and industrial areas into publicly-owned storm drains, which will eventually lead

into the environment. This can also cause environmental degradation.

5. After implementation of source controls, the drains can be cleaned to prevent further environmental degradation.

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AQUATIC ECOLOGICAL RISK ASSESSMENT OF LEAD, COPPER, PHTHALATES AND POLYNUCLEAR AROMATIC HYDROCARBONS DISCHARGED VIA STORMWATERS AND COMBINED SEWER OVERFLOWS INTO LAKE UNION

Robert E. Stuart*

Combined sewer overflows (CSOs) and storm drain (SD) runoffs currently are of special concern because they represent significant nonpoint sources of pollution. Discharges from CSOs and SDs enter metropolitan lakes, streams, and Puget Sound, contributing pollutants which may adversely impact these ecosystems. This issue is particularly important for smaller embayments where the impacts may be greater due to lesser dilution. For this assessment, water column - and bottom - dwelling fish and invertebrates are evaluated to examine their probable responses to contaminants carried by CSOs and SDs.

The objective of this risk assessment was to define the most probable contaminant concentrations to which aquatic life would be exposed, and then relate these to the probabilities of adverse effects on aquatic organisms. The intent was to provide decision-makers with a less subjective appraisal of risk than possible with qualitative risk assessments. This was accomplished through the use of the uncertainty analysis model MOSES (Model Sensitivity Evaluation System), which uses Latin Hypercube Sampling techniques to perform sensitivity and uncertainty analyses on a variety of environmental data.

Four chemicals--lead, copper, phthalates, and polynuclear aromatic hydrocarbons--were selected for evaluation based upon their concentrations in the discharges, environmental fate characteristics, and toxicological properties. Estimated environmental concentrations were predicted for these chemicals in the water column, bound to particulates and sediments, and dissolved within the sediment's interstitial water. The estimated concentrations of the dissolved chemical were compared to the statistical probabilities of that chemical's acute and chronic toxicity to aquatic life. The potential impact to freshwater biota was evaluated using EPA water quality criteria.

Chemicals entering Lake Union from SD and CSO discharges occur in the water column bound to particulates and dissolved in interstitial water. The results indicate that the four chemicals tend to be particulate-bound, in a state where they are not readily bioavailable. Chemicals are found in the highest concentrations in the sediments and sediment interstitial water. Estimated concentrations of the four chemicals dissolved in the water column are negligible and pose negligible risk of either acute or chronic toxicity to organisms inhabiting the water column. Over 95% of the estimated concentrations of dissolved lead in the interstitial water derived from CSO

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and SD discharges exceeded EPA's acute toxicity criterion. Over 95% and approximately 35% of the estimated concentrations of dissolved copper derived from CSOs and SDs, respectively, exceeded EPA's chronic toxicity criterion in the interstitial water. Only 5% of the estimated concentrations of phthalates in the interstitial water derived from SD particulates exceeded EPA's chronic toxicity criterion. Polynuclear aromatic hydrocarbons were found in the interstitial water derived from both CSO and SD particulates; however, none exceeded published data concerning acute toxicity. There are no EPA water quality criteria for chronic toxicity of PAHs.

URBAN STORMWATER RUNOFF AND COMBINED SEWER OVERFLOWS: AN ECOLOGICAL AND HUMAN RISK ASSESSMENT*

SYDNEY F. MUNGER

THE RISK ASSESSMENT PROCESS

The risk assessment process is a useful tool to logically determine the significance of environmental data. It can be used by the scientist to respond to the questions, "What do these data mean?", "What does it mean if very small concentrations of potentially toxic chemicals are added to the environment?", "Is it a problem?". The risk assessment answers these questions by defining the probability of harm occurring to the ecosystem or to human health as a result of a given action. As was discussed by Lowrance in his book, Of Acceptable Risk, nothing we do or have imposed on us is absolutely free of risk. The concept of "safety" is best viewed as a judgement concerning the acceptability of the risk involved.

The risk assessment defines the level of risk using scientific data, combined with scientific judgement. This information can then be used by decision makers to define an acceptable level of risk, and to develop a strategy for risk management.

Our goal in this project was to use the risk assessment process to define the risks to the freshwater ecosystem and to human health, resulting from the discharge of urban stormwater runoff and combined sewer overflows. The approach taken was to select representative chemicals found in both storm drain discharges (SDs) and combined sewer overflows (CSOs) in the Seattle area and to predict their concentrations along paths of environmental exposure. These predicted concentrations were then compared to levels of the chemical known to cause acute or chronic effects in the aquatic ecosystem or in humans. In risk assessment terminology, these two operations--prediction of concentrations and determination of toxic effect--are known as Exposure Assessment and Hazard Assessment. The probability of risk is determined by comparing the exposure levels to the concentrations causing toxic effects. This last operation is called Risk Assessment.

THE METHODOLOGY

Exposure Assessment

The exposure assessment required that the concentration of toxic chemicals in both stormwater runoff and CSO discharges be defined. This was accomplished by compilation of existing local data collected on the quality of stormwater and CSOs during the last 17 years in the Seattle-King County area. These data have been tabulated and are included in Appendix A.

Using this data base, four chemicals or groups of chemicals were selected as representative of the entire data base and most likely to be of concern. The selection was based on the prevalence of the chemical in the stormwater or CSOs, and the toxicity to aquatic species and/or human health, the bioconcentration factor, and persistence of the chemical in the environment. The four chemicals selected were lead, copper, polycyclic aromatic hydrocarbons (PAHs) and phthalic acid esters (PAEs).

The fate of these compounds along exposure pathways was determined based on the quantity of chemical discharged in a typical storm event, on a series of assumptions described below and on application of a statistical modelling procedure —uncertainty analysis—to define the probabilities of different chemical concentrations occurring in the water, sediments, interstitial water, and fish of Lake Union, a highly urbanized lake in Seattle.

The risk assessment process requires that assumptions be made where data gaps exist or where resource limitations require that modelling efforts be simplified. Key assumptions of this assessment are listed here, and are considered critical to using the results of this assessment for decision making.

Key Assumptions

- The data base used to describe the concentrations of chemicals present in stormwater and CSOs is representative of all stormwater and CSOs in the Metro service area.
- The 1986 lead data from storm drains and CSOs more closely represents the current lead concentrations in urban runoff, than does the entire data base (See Appendix A). This assumption is based on more stringent federal regulations concerning leaded gasoline and increased use of unleaded gasoline in recent years, and is supported by local as well as national observations of reduced lead levels in runoff and street dust in recent years
- The 1986 PAH and PAE data more accurately represent the concentrations of these toxic organic groups in stormwater and CSOs than does the entire database (See Appendix A). This assumption is based on improved analytical techniques for environmental organic samples in recent years.
- The entire copper data base best represents the concentration of copper in stormwater and CSOs (See Appendix A).
- Lead, copper, PAHs and PAEs represent the most prevalent, the most toxic and most persistent chemicals present in stormwater and CSOs, and therefore will describe the limiting risk factors to human health and the aquatic ecosystem as a result of storm drain and CSO discharges to a fresh water receiving body.
- Chemicals must be dissolved to be bioavailable.
- The partition coefficient or K_d for lead in Lake Union can be represented by K_d values measured in fresh and marine waters throughout the United States (EPA, 1980).
- Dilution of chemicals in the water column is limited to a defined volume.
- The predicted concentrations of lead, copper, PAHs and PAEs in the sediment assume that 50 percent of the particulates are

transported from the area and the remaining 50 percent are diluted with clean sediment resulting in a sediment concentration that is 50 percent less than pure storm drain or CSO particulates.

Hazard Assessment

The Hazard Assessment for the ecological portion of this document was performed in accordance with the guidelines established by EPA for the derivation of water quality criteria. The probability of acute toxicity occurring in aquatic species, given exposure to lead, copper or phthalic acid esters was evaluated by plotting the distribution of concentrations causing 50 percent mortality in different species, against the population of species that had been tested for acute toxicity. A similar procedure was followed to determine the probability of occurrence of chronic toxicity. Acute toxicity measures the concentration of a chemical that will cause mortality in a short period of exposure (i.e., 48 to 96 hours). Chronic toxicity measures the more subtle effects of a chemical over a longer period of time which can include the entire life cycle of an organism. EPA criteria are based on both acute and chronic toxicity values for a representative sample of species and are designed to provide protection for at least 95 percent of the aquatic species.

The hazard assessment for the human health portion of this report relies on criteria developed by EPA for protection of human health for PAHs, PAEs, and copper. The acceptable daily dose for lead was derived in reference to the work by Elias (1985) from the Environmental Criteria and Assessment Office of the US EPA, and to the World Health Organization (1977), the Center for Disease Control and the American Academy of Pediatrics. The hazard assessment uses existing toxicology data to determine the quantity of a chemical that must be ingested to cause an effect.

In the case of this risk assessment, the relatively low concentrations of chemicals being considered limits the concern to the chronic effects of these chemicals. For lead, this means effects on the blood system and on the mental development of children. The chronic effect of PAH ingestion is an increased risk of cancer. The effects of PAE ingestion used to determine risk in this assessment were liver, kidney and testes impairment.

The Conclusions

The conclusions of this assessment of risks to human health and to the aquatic environment from discharges of CSOs and stormwater is intended to be applicable for all receiving waters in the Seattle-King County region. However, Lake Union has been referred to throughout as a reference point or example. That is, the conclusions refer to expected effects of outfalls in and of themselves. The predicted concentrations of chemicals resulting from CSOs and SDs are also compared to existing conditions in Lake Union to see whether any additional outfalls there would increase, decrease or maintain current levels of risk. Existing conditions in Lake Union arise not only from many CSO and storm drain outfalls around its shoreline, but from current and historical industrial and municipal discharges, many of which no longer occur. The use of Lake Union as an example is related to the availability of data regarding its current condition as compared to other local receiving waters and also to the planning for CSO abatement in the Lake Union drainage areas.

The Assessment of Risk to the Aquatic Ecosystem

A major conclusion of the assessment of risk to the aquatic environment is that lead is the only chemical, from either storm drains or CSOs, where a potential risk is indicated. While lead in the environment is everywhere, the sources are known and controls are feasible and are being implemented. The Puget Sound Air Pollution Control Agency (PSAPCA) has estimated local lead sources to be 96 percent vehicle-related. Vehicle-related lead enters the atmosphere and settles on surfaces near roadways from which it can be washed into sewer systems. In 1980, PSAPCA documented declining levels of local atmospheric lead and projected a continuing decline until about 1988, because of reduced use of leaded gasoline and automobile emission controls.

Lead levels on the ground from road dust are declining more slowly so that reductions in stormwater and CSO levels can be expected throughout the 1980s and probably beyond.

These continuing reductions would be realized via control at the source. Beyond the decreases due to leaded gasoline reduction, additional source control can be obtained through erosion and sediment control since the lead will largely be associated with particulates.

WATER COLUMN

1. Estimated concentrations of the four chemicals of interest (lead, copper, PAHs and PAEs) dissolved in the water column, as a result of CSO and SD discharges, are negligible and pose no increased risk to pelagic fish.

SEDIMENT and INTERSTITIAL WATER

Lead

2. Estimated concentrations of dissolved lead in the interstitial water in sediments deposited by storm drains and CSOs, as well as existing sediment conditions in Lake Union, pose a potential risk (see following text for relative magnitude) to freshwater aquatic organisms.

Storm drains

EPA used ten aquatic organisms to represent the typical receiving water environment to establish chronic toxicity levels. Of these ten organisms, four are representative of bottom (benthic) dwelling organisms that would be dependent upon interstitial waters. More than forty percent of the expected concentrations of lead in interstitial waters derived from storm drains would not exceed the chronic toxicity value for any of the test organisms. Approximately the upper 50 percent of the projected lead concentrations would affect the most sensitive species tested (freshwater amphipod). One additional benthic test organism (a freshwater snail) would be affected by approximately the upper 20 percent of the predicted interstitial water concentrations.

The lowest, 30 percent of the projected concentrations of lead would be below EPA's overall criterion for chronic toxicity.

CSOs

The highest estimated dissolved lead concentration in the CSO derived interstitial water equals the chronic toxicity of the most sensitive species tested by EPA. The lower 80 percent of the estimated concentrations of dissolved lead in the interstitial water derived from CSO particulates are below the EPA criterion for chronic toxicity.

Existing Lake Union Conditions

The highest estimated concentration of dissolved lead in the pore water of existing Lake Union sediments falls between those of CSO and SD sediment deposition. Approximately 30 percent of the estimated concentrations of dissolved lead in the interstitial water of Lake Union sediments fell below the EPA criterion for chronic lead toxicity. One of the benthic species tested by EPA would be affected by 50 percent of the estimated lead concentrations resulting from existing Lake Union sediments.

Copper

3. Copper is not a metal of concern. The highest estimated concentrations of dissolved copper in the interstitial waters derived from CSO particulates, SD particulates and actual Lake Union sediments are 1.65 μg/l, 2.52 μg/l and 0.78 μg/l respectively. These values are less than the chronic toxicity criterion for copper (4.42 μg/l).

Polycyclic Aromatic Hydrocarbons

4. Environmental effects information on PAHs is not sufficient to allow estimation of risk to aquatic organisms. Polycyclic aromatic hydrocarbons are found in the sediment interstitial water from both CSO and SD particulates. The highest concentrations of PAHs predicted in storm drain and CSO sediments are 0.86 μg/kg and 0.04 μg/kg, which are 4 to 5 orders of magnitude lower than the existing levels of PAHs in Lake Union sediment (31,623 μg/kg to 85,266 μg/kg).

Phthalic Acid Esters

5. The estimated concentrations (upper 95 percent confidence level) of dissolved PAEs in the interstitial waters of CSO sediments, storm drain sediments and Lake Union sediments are $2.37X\ 10^{-3}$ μg/l, $8.52\ X\ 10^{-5}$ μg/l, and 430 μg/l, respectively. These estimated phthalic acid ester concentrations in the pore waters of CSOs and SD sediments are well below the chronic toxicity criterion of 34.42 μg/l. However, the estimated concentration in existing Lake Union sediment interstitial water exceeds the chronic criterion.

Apparently either CSO and SD sediment quality has improved and should improve Lake Union sediments over time, or Lake Union sediment concentrations of PAEs is more influenced by other inputs than CSOs and SDs, and therefore changes (decreases) in PAEs will be governed by things other than management of CSOs and storm drains.

The Assessment of Risk to Human Health

When discussing and evaluating the level of risk to human health through exposure to environmental chemicals as a result of a specific event, it is important to place that level of risk in perspective to background levels of exposure to these same chemicals.

In the case of each of the chemicals considered in this risk assessment, background environmental exposure is ubiquitous. Living in an urban environment affects the level of exposure simply through normal daily activity of eating food and water and breathing air that have been subjected to the toxic waste products of our society.

For example, the levels of lead in the urban environment are of significant concern to environmental health specialists. A child who lives in the urban environment and eats vegetables from an urban garden can exceed the acceptable daily dose in their normal life. As noted above, lead in the urban environment is a pervasive problem. Each person can address it by supporting more stringent leaded gasoline laws and by personally choosing to use unleaded gasoline.

A major conclusion of this assessment is that the only restrictions on normal consumption levels necessary to protect human health because of CSO or stormwater effects would be in the consumption of crayfish by urban children. This restriction would be due to predicted levels of lead in the crayfish. It should be noted, however, that in the case of Lake Union, a greater restriction on crayfish consumption results from levels of PAHs currently in lake sediments.

As the reader evaluates the conclusions of this risk assessment, it is important to place any potential areas of increased risk into perspective with what one considers to be acceptable exposures.

Lead (See Table 1E)

1. There should be no additional risk from lead ingestion to children or adults who swim in Lake Union under current conditions or after a stormwater discharge or CSO event.

2. Adults and children should be at no increased risk from the daily consumption of pelagic (water column) fish from Lake Union under current conditions, or as a result of a storm drain or CSO discharge. Fish, such as rainbow trout, living in the water column should not accumulate any detectable increased levels of lead in body tissues as a result of storm drain discharges or CSO events in Lake Union.

3. Fish that eat off the bottom and are exposed to the interstitial water can be expected to accumulate detectable levels of lead. Bottom fish living in Lake Union are currently exposed to

relatively high levels of lead in the interstitial water. These levels would be increased slightly in the area of a storm drain discharge and decreased in the area of a combined sewer overflow discharge. The quantities of bottom fish that can be eaten daily by the most sensitive population classes are summarized in Table 1E.

Bottom fish exposed to storm drain sediments could accumulate increased levels of lead. The consumption of these fish by urban children should be limited to 30 grams or one ounce per day. This means that if Seattle area children consume fish at the national average rate of 6.5 grams per day, then they will not be at increased risk from consumption of bottom fish exposed to storm drain sediments.

The urban adult who also eats fruit and vegetables from an urban garden, smokes cigarettes and drinks wine can consume up to 166 grams of bottom fish exposed to storm drain sediments

TABLE 1E. THE QUANTITIES OF FISH, CRAYFISH AND WATER THAT CAN BE CONSUMED WITHOUT EXCEEDING THE ACCEPTABLE DAILY DOSE FOR LEAD.

Environmental Compartment/ Pathway	Urban Children	Urban Adults+smoker+wine
Swimming	No limit*	No limit
Eating Pelagic Fish	320 grams/day	No limit
Eating Bottom Fish		
Existing	80 grams/day	No limit
SD	30 grams/day	166 grams/day
CSO	320 grams/day	No limit
Eating Crayfish		
Existing	15 grams/day	90 grams/day
SD	5.4 grams/day	33 grams/day
CSO	61 grams/day	373 grams/day
Other sources of Ingestible Lead		
Leafy Vegetables (Urban Garden)	33 grams/day	203 grams/day
Market Milk	750 ml/day	No limit

*No limit indicates that the quantity of water that can be ingested is in excess of 14 liters per day; and that the quantity of fish or crayfish that can be consumed per day is greater than 454 grams (1 pound).

per day before reaching potential effect level and is thus not at increased risk even under 10 times the normal consumption rates.

The urban child, who already ingests close to the maximum advisable lead per day through normal daily exposures, should limit the consumption of bottom dwelling fish exposed to current Lake Union conditions to 80 grams or approximately 2.5 ounces per day. If Seattle area children consume fish at the national average rate of 6.5 grams per day then they will not be at increased risk from consumption of bottom fish exposed to stormdrain sediments.

4. Crayfish live on the bottom of Lake Union and are a popular recreational as well as commercial harvest from the lake. The quantities of these crayfish that can be eaten by the most sensitive exposure groups without exceeding the acceptable dose for lead are summarized in Table 1E. The average crayfish muscle tissue is 2.5 grams and the average number of crayfish in a single restaurant serving is approximately 12. Therefore an average serving is 30 grams of meat.

   The average urban adult can eat 77 grams daily of crayfish exposed to storm drain sediments.

   The average urban child needs to limit daily consumption of crayfish exposed to storm drain sediments to 5.4 grams (about two crayfish).

   Urban adults from the most sensitive exposure class can eat crayfish daily up to a limit of 33 grams for the storm drain exposed crayfish without increased risk from lead exposure. This is equivalent to one average restaurant serving per day.

   The average urban adult can eat 211 grams per day of crayfish exposed to existing Lake Union sediments.

   The average urban child can eat 15 grams per day of crayfish exposed to existing Lake Union sediments.

   Another way to look at these numbers is to say that if the urban child, who is already exposed to other lead sources, eats an average serving of crayfish (12 tails) once every two days for those crayfish exposed to current Lake Union conditions or once every five days if crayfish are harvested near a storm drain outfall, increased risk due to lead exposure is not likely to occur.

<u>Polycyclic Aromatic Hydrocarbons (PAHs)</u> (See Table 2E)

1. The risk attributed to the consumption of polycyclic aromatic hydrocarbons via Lake Union water, fish and crayfish should be viewed as an incremental risk above that experienced by daily

exposure to PAHs in the normal diet. EPA estimates that the average person is exposed to .16 to 1.6 µg PAH per day and the best current estimate of risk is that .061 µg per day is equivalent to a 1/100,000 risk of developing cancer during a lifetime.

2. The concentrations of PAHs estimated to be added to the water column as a result of a CSO or stormwater discharge are insignificant, and would not bring about changes detectable by current techniques.

   Swimming in Lake Union and ingesting the estimated 50 ml of water would not significantly increase exposure to PAHs.

3. Fish living in the water column should not accumulate any detectable increased levels of PAHs as a result of storm drain discharges or CSO events. Consumption of pelagic fish from Lake Union at the national average rate of 6.5 grams per day should not exceed the 1/100,000 risk of developing cancer.

4. Fish that feed from the lake bottom, and are exposed to the interstitial water, can be expected to accumulate increased levels of PAHs. However, no limitations are indicated for consumption of bottom fish from storm drain or CSO outfall areas. Under current Lake Union conditions, however, bottom fish consumption should be limited to 1.3 grams per day. <u>The contributions of PAHs from both storm drains and CSOs would provide sediments that have lower PAH values than are currently present in Lake Union.</u>

5. Crayfishing in Lake Union is a popular recreational and commercial harvest. No limitations are indicated for consumption of crayfish from storm drain or CSO outfall areas. However, consumption of crayfish from Lake Union, based on predicted values for <u>current conditions</u>, should be limited to 0.07 grams per day. If instead, estimates of risk for current conditions are based on the existing measured values for PAHs in crayfish tail muscles, then a person can consume 0.85 grams per day. This quantity is still low, but means that a person could eat an average serving (12 tails) once every 35 days without increasing his/her risk level beyond 1/100,000.

<u>Copper</u>

There are no expected increased risks to human health due to the exposure to copper resulting from CSO or storm drain discharges or from existing Lake Union conditions.

<u>Phthalic Acid Esters</u>

There are no expected increased risks to human health due to the exposure to phthalic acid esters resulting from CSO or storm drain discharges or from existing conditions in Lake Union.

TABLE 2E. LIMITATIONS FOR CONSUMPTION OF FISH AND CRAYFISH BASED ON PAH CONCENTRATIONS PROTECTS AGAINST ANY INCREASED RISK OF CANCER ABOVE 1/100,000.

Environmental Compartment/ Level.	Quantity that can be consumed daily without exceeding a 1/100,000 risk		
Source	Existing	CSO	SD
Swimming	884 ml/day	884 ml	884 g
Eating Pelagic Fish	29 g/day	29 g	29 g
Eating Bottom Fish	1 g/day	No Limit*	No Limit
Eating Crayfish			
Predicted**	.07 g/day	No Limit	No Limit
Measured ***	.85 g/day	ND****	ND
Other Food Sources			
Charcoal Broiled Steak	.42 g	NA*****	NA
Barbecued Ribs	.50 g	NA	NA

*No limit indicates that the quantity of fish, or crayfish that can be safely consumed is in excess of 454 grams (1 pound) of tissue per day.

**Based on the highest predicted concentration in crayfish.

***Based on measured concentrations in crayfish.

**** ND = No Data

***** NA = Not Applicable

TREATMENT OF STORMWATER RUNOFF IN URBANIZING AREAS

by Gary R. Minton, PhD, PE[1]

The PSWQA Plan emphasizes treatment of urban stormwater runoff. Implementation modes are privately funded treatment systems in new developments, and publicly funded facilities controlling large areas. Presented here is the perspective of the engineer called upon to design, maintain, and possibly garantee the performance of the treatment systems. To fullfill this role what do we know; what do we need to know?

WHAT WE NEED TO KNOW IS ...

Characteristics of stormwater

1. What are the specific pollutants of concern?
2. As we first remove the settleable solids, what is:

   - % of each pollutant associated with the solids;
   - settling velocities of the solids?

How big do we build the facility

3. How much of the pollutant are we to remove?
4. How large a storm must the facility treat?

What are the treatment methods

5. What does each do in the way of removal?
6. The experience of other communities - which design features work well, which do not work well?
7. The possible "side-effects" - what opposition may come from professionals or citizens whose interests are not soley with nonpoint pollution?

What are the costs

8. What are the construction costs?
9. The proper operation and maintenance standards?
10. The long-term O&M and replacement costs?

CHARACTERISTICS OF URBAN STORMWATER RUNOFF

We have a good understanding of what is in urban runoff, what are the specific or general pollutants of concern, and the concentrations of each constituent[2]. The concentration of some metals can exceed acute toxicity criteria. However, dilution by the general stream flow reduces these concentrations below the criteria.

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[2]Contact the author for references supporting this paper

We have a reasonably good understanding as to the association of each pollutant with the suspended solids. Generally, the majority of the phosphorus, most trace metals and organics are bound with the suspended particles in stormwater. However, the greater proportion of these pollutants are associated with the finer particulates. Consequently, removing, say, 70% of the suspended (non-filterable) solids will not generally remove 70% of the pollutants in the particulate form.

An important characteristic of stormwater is the settling velocity distribution for the suspended solids. Knowing this distribution allows a rational sizing of the surface area of the pond. National data are limited to a few laboratory settling tests; the USEPA NURP study provides data for 46 storms at five sites across the country.

There are no settling velocity data for western Washington, although there are some data indicating the size distribution for suspended solids. Size distributions can be used to generate velocity distributions. Proponents of this approach assume that the particles have a specific gravity of 2.65 (sand) and behave according to Stokes Law. This approach was followed in generating Figure 1 in which data collected by Bellevue is used. However, this approach probably overestimates the rates of settling.

A final consideration is bioavailability; that is the fraction of the pollutant that is actually influencing the surrounding environment. Very little of the pollutant is in the free ionic form which determines the environmental effect. However, there is relatively information on this question.

HOW BIG DO WE MAKE THE FACILITY

This question comes in two parts: how much of the pollutant will we be asked to remove, and how large a storm must the facility handle.

The removal goal

No one has provided the engineer with the answer as yet; perhaps it will evolve from an analysis of "incremental increase in pollutant removal rate vs incremental increase in cost". This relation-ship is illustrated in Figure 1 where for a given runoff coefficient, doubling the size of a pond will not double the removal rate of suspended solids.

What can be said at this time is that if the removal goal for any of the pollutants is in excess of 50%, methods that only remove the settleable solids will not be sufficient. Exceptions are suspended sediment and lead.

How large a storm

The appropriate frame of reference depends on the objective. With most pollutants, we should be concerned about the aggregate accumulation of the pollutant over time in the water body, not the immediate acute effect. Conse- quently, we should size the facility to capture and treat a reasonable amount of runoff over time.

Figure 1 is based on the mean values of key storm statistics (e.g. mean storm intensity, and interevent time), rather than the more common concept of the infrequent single event storm (e.g. 10 year storm). If the traditional design storm concept is used, sizing the facility for a 1 or 2 year storm should be sufficient.

One pollutant that may require facilities to be sized for the more extreme events is fecal coliform bacteria: a single event can cause the temporary shut down of a commercial shellfish operation. Here we should be conducting a "incremental removal/incremental cost" analysis of sizing the facility to reduce the number of "untreated or reduced treated" events to, say, 1 per year, 1 per two years, 1 per five years, etc.

WHAT ARE THE TREATMENT METHODS

Table 1 lists the most commonly used or proposed methods, some jurisdictions where they are currently being used with regularity, and the findings of research to date. We can group the methods into two general types: those that likely remove only settleable solids and pollutants; and those that remove settleable and colloidal solids, and soluble pollutants.

These points are stressed. With the possible exception of wet ponds, research data are very limited. The table shows that wet ponds and wetlands can be effective at removing both suspended and soluble pollutants, although the data for natural wetlands indicates very inconsistent results with soluble phosphorus.

The experience of eastern states may not be applicable for western Washington. Eastern researchers usually sampled few storms (less than 10 events) and sampled most frequently during the spring and summer months when biological activity and runoff is highest. However, in western Washington the majority of the runoff occurs during the dormant season.

The experience with grass swales, a method receiving widespread attention in western Washington, has been inconsistent. Some studies have found them to be effective at removing suspended solids; others have not.

The problem for infiltration basins in western Washington is that most soils contain a layer of hardpan at a depth of 3 to 5 feet. It may be feasible to place underdrains above the hardpan layer. However, there is no data on the treatment effectiveness of only 2 to 3 feet of soil under western Washington conditions. It is likely that 2 to 3 feet of soil will effectively remove metals, but there is considerable uncertainty with the other pollutants.

What are the possible side effects: possible groundwater contamination by infilration basins although experience has shown that metals, phosphorus, bacteria, viruses and most organics are removed within a few inches to a few feet of soil; deleterious effect of chemicals on fish, either through normal use or abnormal facility breakdowns; inadequate maintanance may during an extreme storm event result in a "slug" discharge of previously removed pollutants.

WHAT ARE THE COSTS

Given clear design criteria, engineers can provide reasonably accurate estimates for the initial construction costs. However, because the experience of the jurisdictions noted in Table 1 is on the order of 3 to 5 years, we have little understanding of the proper maintanance procedures and facility life. Estimates for O&M and replacement costs are therefore very uncertain.

WHAT DO WE NEED TO KNOW THAT WE DO NOT KNOW NOW

Based upon the above discussion, the following suggestions are made:

1. Better understanding of stormwater including:

    - settling characteristics;

    - recalculation of rainfall statistics using the complete local rainfall data base.

2. Removal mechanisms of soluble pollutants in wet ponds, wetlands, grass swales. Can we obtain significant removals in western Washington?

3. Need to evaluate real facilities under real life conditions; facilities designed with specific design criteria. Items of particular interest

   - retrofit strategies for exising ponds and tanks

   - design criteria of rock filter berms in two cell ponds;

   - proper techniques for reestablishing grass in dry ponds and grass lined channels

   - effectiveness of infiltration basins with underdrains

   - proper grass height and density in swales

   - effective methods for fecal coliform reduction.

4. What are the appropriate maintenance practices to insure continued effectiveness, particularly:

   - harvesting in artificial wetlands

   - removal of accumulated solids in dry ponds

   - keeping infiltration surfaces rejuvenated and porous.

5. Community reaction, particularly:

   - wet ponds/wet lands.

6. Open interagency discussion on these issues:

   - infiltration basins and groundwater pollution

   - use of large public facilities in salmonid streams

   - use of chemicals for treatment

   - bioavailability and the benefit of removing the particulate pollutant fraction

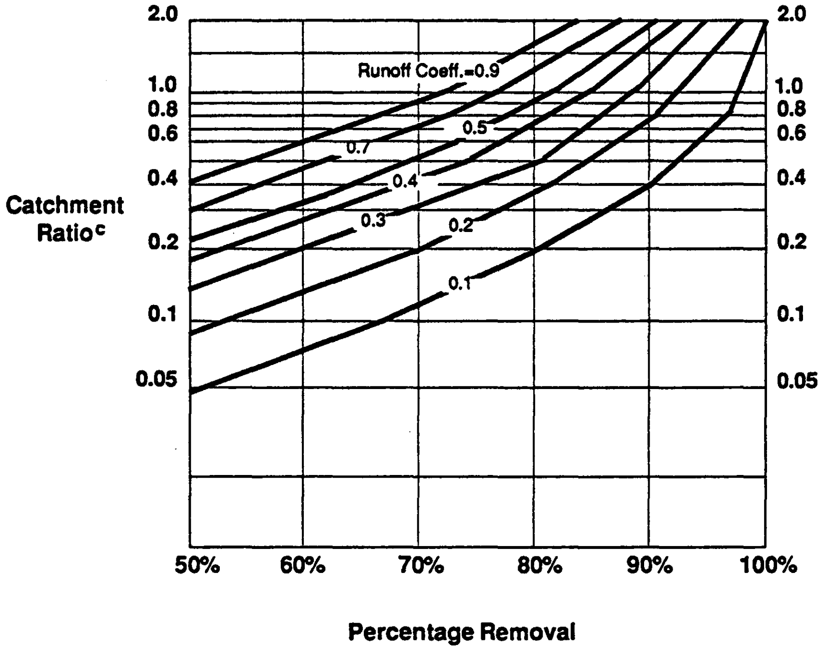
   - incremental cost versus the incremental benefit of facility sizing.

TABLE 1 TREATMENT OF STORMWATER

METHOD	REMOVAL RATES (a) SUSPENDED SOLIDS	SOLUBLE POLLUTANT	COMMUNITIES
Dry ponds	0% (b)	0%	Everyone
Extended detention dry ponds	40 - 65%	0%	Virginia, Maryland
Sand filtration	50 - 75%	0%	Austin, Texas; Florida
Concentrators (c)	40 - 90%	0%	Field tests only
Grass swales	50 - 90%	0 - 15%	Virginia, Maryland, Florida Mountlake Terrace, Redmond
Wet ponds	50 - 90%	0 - 50%	Maryland, Florida, many other jurisdictions
Wetlands	50 - 95%	0 - 50%	Many jurisdictions
Natural soil infiltration basins	100%	75 - 90%	New York, Florida, Maryland, California
Natural soil infiltration basins with underdrains	90 - 100%	50 - 75%	None
Peat/sand filtration system (constructed soil system)	100%	50 - 90%	Minnesota
Chemical treatment (Alum)	50-95%	50 - 75%	Tallahasse, Florida

(a) For most methods, values based on very little field research
(b) Two cell pond with intrapond gravel berm could remove some suspended solids
(c) Swirl and teacup concentrators

Figure 1
Predicted Removal of Suspended Solids in a Wet Pond as a Function of Runoff Cofficient[a] and Catchment Ratio[b]



a. Runoff coefficient - the fraction of rainfall that becomes runoff.
b. From Minton, 1987.
c. Catchment ratio is the ratio of the surface area of the pond to the surface area of the development expressed as a percentage.

Summary Statement

William H. Funk*

Agriculture contributes over half of the sediment load to the nation's waterways from croplands, grasslands, and livestock operations. Significant amounts of key nutrients, phosphorus, nitrogen, as well as bacteria and pesticides are also contributed. In many instances, pollutant loading from diffuse agriculture loadings can be reduced by relatively inexpensive best management practices (BMP's) such as rain collection devices, sediment traps, and vegetative buffers. Farmers and ranchers need incentives and assistance to control and reduce diffuse pollution sources. Implementation of BMP's is most effective when achieved by cooperative efforts rather than regulatory measures.

Urban runoff also plays a major role in contributing to water quality problems, especially in those areas undergoing development. Sediment addition has increased by as much as a hundred times in some instances. Urban runoff can also contribute heavy metals, pesticides and toxic chemicals in addition to nutrients. Best management practices developed by those individuals involved with development have been evolving due to a recognition and acceptance of the need to control pollutants. Mechanisms are being evaluated by professionals working in both the development and pollution control fields.

Wetlands are increasing being used for flood storage either directly or indirectly to improve runoff water quality. Over seventy wetlands around King County are being studied to determine the long-term consequences of runoff quantity and quality. It is hoped to gain sufficient knowledge to improve present management techniques and protect the many ecological functions of the wetland.

Storm drains may act as residue collection systems for heavy metals and toxic substances. Industrial areas and abandoned factories in the Puget Sound area may be significant reservoirs for toxicants such as lead, polychlorinated biphenyls (PCB's) and polynuclear aeromatic hydrocarbons (PAH's). Strategies are being developed to contain and isolate toxicant-laden sediments until removed. Sampling of sediment in drains at key junctures of the storm catchment system and review of land uses in the drainage basin, in many instances, has indicated potential sources of toxicant pollutants. Similar approaches in large systems have not been as effective because of the flushing action of high-volume systems, forcing the sediment through the

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system into receiving waters. Development of appropriate source control, best management practices and housekeeping measures for industrial facilities is needed. Additional research in developing techniques for prevention and control of toxicants in stormwater is essential to achieve reduction of hazardous materials in the sediments of the Puget Sound area.

Aquatic risk assessment of four chemicals, lead, copper, phtalates, and PAH's, were selected for evaluation based on their concentrations in stormwater discharge (SD) and combined sewer overflow (CSO), their toxicological properties and possible environmental fate. The objectives was to define probable contaminant concentration to aquatic life and relate possible adverse effects on aquatic organisms in Lake Union. The potential impact to freshwater organisms was evaluated by EPA water quality criteria. Highest concentrations of the chemicals were found in the sediment and interstitial waters. The metals exceeded EPA's acute toxicology criterion. Only 5% of the phtalates exceeded EPA's chronic toxicity criterion. Measurements of the PAH's did not exceed published data concerning acute toxicity. There are no EPA water quality criteria for chronic toxicity of PAH's.

It was recommended that a greater research effort is needed to verify these results in other bodies of water and to test for other contaminants that may be added to other bodies of water in the Puget Sound area.

Risk assessment was performed on the effects of stormwater runoff and combined sewer overflow to determine the significance of four groups of chemical contaminants to human health and the freshwater ecosystem.

Lake Union was used as an example, but results were intended to be applicable for all receiving waters in the Seattle-King County region. For risk to the aquatic ecosystem, lead is the only contaminant from either storm drains or CSO where potential risk is indicated.

Human risk assessment includes consideration of background environmental exposure. Tables of data were presented to indicate the quantities of water, fish and crayfish that can be consumed without exceeding acceptable intake levels. The treatment and removal of stormwater pollutants is not achieved by any uniform design. Each region or watershed may have characteristics for which design and treatment criteria have to be developed. Research efforts are essential to evaluate those methods most applicable in Western Washington.

CONCLUSIONS

We are indeed fortunate in the United States in most instances because of our technological capabilities and understanding as well as a comparatively good education background of our population. Public understanding and appreciation of water problems are critically important in dealing with threats to our public health and welfare. We have stopped and then begun the reversal of the trend of environmental contaminants entering our surface waters from point sources. We must take decisive action to control pollutants from diffuse sources.

Damage to the aquatic environment by these pollutants is many faceted. Primary damage can occur from rapid runoff carrying large amounts of sediment from urban areas, construction sites, agricultural and sivicultural areas. The sediment abrades micro- and macroscopic organisms from their attachment sites, destroys their habitat and deprives larger organisms of their food supply. A secondary poisoning effect may occur when metals, pesticides and other toxins are swept in from the urban, agriculture, and silviculture areas and directly released or released by life processes of organisms living in the sediment. The biochemical mechanisms are not fully understood and additional research is essential.

Thirdly, top soils flushed from construction, development, agriculture, and silviculture sites contain the inventory of prime nutrients such as phosphorus, nitrogen, potassium and essential growth factors. This process leaves the original site impoverished and the nutrients upon arrival at quiescent sites stimulate undesirable aquatic plan growth in the slow-moving streams, rivers, and estuaries.

The authors of this session did a good job in document the issues and complexities of nonpoint or diffuse source problems. Many took us several steps forward in showing solutions, partial solutions and mechanisms that can work to alleviate many water problems. There were common threads of concern among several speakers such as the observation that in many instances, there appears to be a lack of perception of the interconnection of water problems and in too many instances no overall responsibility or interest in resolving the problem. In fact, in some cases there was an attempt to push decisionmaking back to the researcher rather than handling it at the elected or management level.

Several authors and attendees expressed disappointment that no groundwater session was included in the conference. They pointed out the need for managers to have a more holistic view of water issues. There is a need to better understand the importance of clean usable water for all species as well as habitat requirements. The role of wetlands in trapping nutrients and contaminants is poorly understood at present. Quantitative studies are necessary to determine this role as well as that of long-term effects of toxic substances upon the aquatic community. It is also essential that managers appreciate the close relationship between surface and groundwater while research continues to elucidate the mechanism by which toxic compounds move from either.

Apologies are expressed in advance to the authors for any misinterpretations, omissions or other unintentional slighting of their work. A sentence or two cannot express the many hours of research or though that they have expended in the quest for solutions to water problems.

MEETING HIGHLIGHTS

WHERE DO WE GO FROM HERE?

Katherine Fletcher
Moderator

Panel Discussion

MEETING HIGHLIGHTS: Where Do We Go From Here?

Katherine Fletcher[1], Usha Varanasi[2], Dennis Flannigan[3], Sheri Tonn[4], Michael Thorp[5]

Kathy Fletcher

I am pleased that so many of you made it all the way through both days of technical sessions and plenary sessions and hope that you have found it worthwhile.

I am joined by four people who will offer some observations about the highlights of the meeting, reactions that each of us has had and some thoughts about where we do go from here. [Other panelists were introduced].

Usha Varanasi

Speaking as a researcher, I would like to discuss some of the comments that were made by the panel that reviewed the committee's report.

The first comment that caught my attention was the comment made by Dr. Kai Lee. His phrase "cautious creativity" was a wonderful phrase because there is a considerable amount of frustration and a certain degree of disdain about directed research. It is true that a number of agencies do have very narrow objectives and the research conducted within those narrow confines is quite often heavily directed. However, in the last two days you have seen the results of a number of those efforts. It is evident that one can use imagination and flexibility within those narrow confines and can be cautiously creative in answering some of the questions about important environmental issues. So cautious or directed creativity, especially in face of budget constraints and the need for short-term solutions, is not a bad thing at all; it could be a very good watchword for us to follow.

The second comment that caught my attention was made by Dr. Ross Heath about management, especially commending the Authority for obtaining

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information first before making management decisions either on research priorities or on the mechanisms for the research foundation. He seemed to think that management based on ignorance is usually the rule of the day and that management based on information was a rare and delightful occurrence. I believe as scientists it is our responsibility to provide credible, clear information to management. What often happens is that there is pressure to come up with answers, and we are tempted to give simple, sleek solutions to managers. They believe in us and therefore sometimes, actually quite often, such simple solutions or concepts get incorporated into multi-million dollar actions. It is up to us to be critical of our concepts and have them reviewed and use them carefully, because without careful information from the scientist it is very difficult to make sound management decisions.

I believe very strongly that no information is better than misleading information. If we have no information we will be frustrated enough to look for answers. If we have misleading information, in addition to making a lot of costly errors, we might also fall into a certain degree of complacency and not continue to search for the right answers.

The third comment that delighted me the most was the comment by Dr. Heath and Mr. Waldo about the importance of peer reviewed publications. It is not just a question of scientists putting their data into peer reviewed journals because it is good for their careers and the prestige of their institutions. It is a necessity today. All the attorneys that are working on environmental issues know very well the importance of peer reviewed journals and information published in peer reviewed articles because that is the information that gets weighted the most. Even though sometimes it is difficult and the process is long, to publish in peer reviewed journals is the only way that we will produce the right kind of information. We should resist the temptation of gray and soft literature that really abounds in the environmental area.

The fourth and most practical of all the comments that were made was made by Senator Talmadge about the considerable concern on the part of the public about the quality of water and the environment; however, when it comes to a long-term investment and long-term commitment somehow there is benign neglect. We need grass-roots support. Instead of just continuously saying that there is no money for research, scientists need to spend some time educating the public, especially young people, in language they can understand, about the fact that having good environmental quality is a luxury in a technological society; a luxury for which we have to pay just as well as we pay for many other necessities of life. Scientists and managers should make a commitment to educating people on this so that it is not only when a crisis, an environmental disaster, strikes that we do something; rather, we will realize that environmental issues are everyday problems that all of us have to work on together.

I would like to thank Kathy Fletcher, who gave me a fine opportunity for the entire year to work on a different process for setting research priorities, on the mechanisms for the foundation, and also on this research conference. I want to especially thank Marcia Lagerloef, whose dedication and even temper were an example for all of us. Sheri Tonn, Jim Abernathy, and Kathy Callison all helped us put this conference together. It was a pleasure.
Thank you.

Dennis Flannigan

I come here to reflect on the meeting from the perspective of the politician because I think that public policy actually makes possible the implementation of all of the dreams that research is headed for. Almost all the research that I have heard about here is directed toward something that we all want, a liveable environment in the Puget Sound region.

A friend was with me yesterday who did not come back today. His comment after about 2 hours was, "All the boring conferences are put on by government and education; all the stimulating ones by business or industry." He has worked in both fields. His comment really was saying that business presenters realize that they are selling something; they are trying to provoke. I want to comment about that because when we began, the inspirational message of the keynote speaker helped sleep appear in several people, including myself. I would expect that in this business (and I think it is the business of trying to save the world we live in), we would want a moment of inspiration to pass between the lips of those people at the front and those ears near the rear. I was really appreciative of the panel, of Jim Waldo, Phil Talmadge, and the others, because they offered the challenge. We were asked to do something with your talents and to make those talents somehow the possibility for the future.

As I wandered around yesterday I was stumbling with the beginning of the conference and feeling that I was in the wrong room at the right time or vice versa. I realized that the conference was aimed more at you than at me. One of things I came to see was how much information and how much research is going on. As a politician, I almost decided we should stop all of the research money, because there is so damn much research that no implementation occurs. I listened to people say that the English sole is dying because the larvae are being killed at the surface; other people said that it can be killed later too. Knowing that the English sole probably can be killed all the way from the bottom to the top of the water is really helpful! The point is that we had better do something about the killing.

We entered the conference feeling pretty proud and smug about ourselves and we wind up the conference having heard that perhaps we have not turned this place around yet. Puget Sound isn't as clean as we hoped or getting cleaner, but perhaps instead continues to deteriorate.

The last thing I heard was about wetlands and how we are mitigating them by ignoring them. Every time we plow one under we say that we should certainly not do that next time. That is a good rule! I serve on the Pierce County Council and developers come to us to point out that any half-acre or smaller wetland area really isn't something to worry about, it is just kind of there, awaiting development. We hear from the "Eco-Freaks," but not too many scientists unless there is someone somewhere with enough money to bring you into the room; and the developers are able <u>not</u> to bring you into the room.

It is not against development I speak, but against the uneven balance. We on the Council will make decisions that there will be another form of pavement, complete with run-off capabilities and all the harm that that may or may not cause, just to the left of your home. The point is that 600 people came here, paid $15-35 for two days, a remarkable number of you continue to be here despite the good advice from old Sol and others. How many of you will make that same two day commitment to the politics of the environment rather than to the research of the environment? If next year you would devote the same 48 hours to marching on the legislature, or holding a brick upside my head and hitting me twice when I vote to the left or the right of a good decision, you will do far more service, believe it or not, even than coming here, because we do not hear your message.

The other end of it is that there is so much research we hear all of the messages. As someone said, a scientist is someone that a lawyer can bring in and the other lawyer can bring in another scientist on the other side of the issue and those of us who sit in the middle are equally confused. Both sides may be sanctioned by the peer review process. People who care but have differing views should get together with their research and try to march at least to a parallel drummer if not the same drummer.

Sheri Tonn

Nearly 700 people registered for this event. Not a whole lot of scientists by the way; academic scientists or consulting scientists yes, but only 100 or so. Absolutely excellent registrations on the part of people from state, local, and federal agencies. I am delighted that everyone came. The people who were absent were the public, the developers, attorneys. I hope that we can remedy that next year. We are going to do this again in 1989. There were 700 of us here this time, maybe we can come up with a few more hundred next time, including representatives of the groups that were absent.

Most of us sat here as sponges during this event. Seven hundred sponges out there taking in what the speakers up front had to say, but other than at the delightful reception last night many of us didn't have a chance for the kind of interchange that would have been nice. I hope that next year we can help make those of us who were passive participants in this event more active participants. I welcome your ideas. Maybe we can come up with some good ways of making it a more active event when there are 1,200 of us here.

Michael Thorp

As a member of the Authority I want to thank everybody who participated on the committee on research for all their hard work and the really fine report that they have turned in. Members of the Authority are trying to deal with 12 or 13 elements in the 1987 Puget Sound Plan. Those elements are very comprehensive and technical, and we couldn't do it without our committees and our task forces. These people have volunteered to help, they have spent a lot of long hard hours on these projects and we on the Authority really appreciate all the hard work that they have put in. We are also indebted to and very thankful to all the people who participated as speakers at this conference the last two days. Again, these people have put in some long hard hours in preparation of their papers and delivering those papers and we really appreciate that.

I have a couple of thoughts to share. I hope by now that you have had a chance to look at the committee's report on research recommendations. This report will be considered by the Authority members in the next month and then we will decide what to do with it. We welcome your thoughts on this report and where we ought to go on research as part of our Puget Sound plan. Please get your comments to us because we all really need to hear from you.

Personally, I am inclined to think that the creation of the Puget Sound Research Foundation is a good idea, but I have some concerns. As you are probably aware, like it or not, we are in an era of fewer and fewer dollars being available for environmental programs. It is very hard to get a hold of those dollars and once you get them it is extremely difficult for those of us who actually have to make a decision about where that money is going to target it for research. It is that simple. We have some extremely good programs here in Puget Sound; we are not funding them all right now. So, when someone wants $100,000 for a research project, that is $100,000 that is not going to go someplace else. Maybe it is not going to go into the watershed planning process; maybe it is not going to go into the program to tighten up discharge permits on industry. It is a very, very hard decision to make.

It is going to be hard for this foundation to get money. It will not be able to do that without broad grass-roots support. Support from folks like you. I don't think you can let it go right now and just say, "That sounds like a good idea." You have to ask yourself, "Would I be willing to participate in that foundation's work? Would I be willing to ask people for money? Would I be willing to go to the legislature to ask for money? Would I be willing to contribute money myself?" If the answer to those questions is no, then we are spinning our wheels here and we might as well forget this project and just try to budget research as part of the Authority.

Also, I wonder how easy it will be to get the foundation off the ground as an independent organization. We need to give that some thought. Maybe for the first few years it will have to be more closely tied to an existing organization such as the Authority in order to get going, get some projects, and show that it can do what we hope it will do. In the long term, it is not going to get $3-5 million a year without showing some successes.

Finally, one thing struck me in the presentations I heard: there is a lot more research going on in Puget Sound than a person might be aware of. One of the big problems that we still face is coordinating that research, everybody knowing what other people are doing and sharing that information. We just don't seem to be able to get a good handle on that. One thing we could do is to try to get together a little more often like this. It doesn't have to be a big two-day get-together. Existing organizations like Ecology, EPA, NOAA, Fish and Wildlife, the counties and Metro could sponsor get-togethers on one given topic, invite people in to present differing views, get some debate going, entertain questions and share information. On behalf of the Authority, I would like to challenge each of those organizations in the next several months to set up one session like this so that we can get an exchange of information going. Maybe by this time next year we can all say, "It has been a great year. We have had a lot of great debate and discussions." Then we can meet together and summarize and go on from there.

Katherine Fletcher

Throughout the meeting we have identified some optimists and pessimists, critics and fans among us and probably mixtures of that within each of us. I would like to make a few comments and perhaps pick up on a few of the highlights of the last two days.

There was one thing that struck me very strongly when hearing about the results of some recent research relating to Puget Sound and some recent bringing together of research and data collected some time ago. We have felt comfortable by describing Puget Sound in general as a relatively healthy body of water with some serious danger signs and some serious problems but overall in pretty good shape. I now personally feel uncomfortable with that characterization as a result of some of the things I have heard. Perhaps it may be more accurate to describe Puget Sound in general as a system showing signs of deterioration with some indications of hope. It is difficult for each of us to get beyond our particular emphasis or interest, but it is important for us to step back and try to grasp what is happening overall in Puget Sound. We did indeed hear some bad news: precipitous drops in numerous fish species in Puget Sound; dramatic evidence relating to toxic effects of the sea-surface microlayer, new evidence of fish reproductive failures; and an admonition to pay much closer attention to some of the nutrient issues. In hearing about all those various things we are invited to make some linkages of our own, but we are also a little frustrated because perhaps we don't know enough to make all the connections that may be staring us in the face.

We did hear some good news. We heard increasing evidence that some specific chemicals that have been regulated or banned are decreasing in the environment and their effects are decreasing. Perhaps related to that, we are seeing seal populations rising. (That may also be a bad news item because we are now pointing our fingers at those critters as a source of possible pollution!) Also perhaps on the good news side, some people turned their creative minds to coming up with a new term for nonpoint source pollution, possibly a more accurate term, "diffuse source pollution."

It is clear that we know a lot more than we did, and that just seems to raise a whole lot of new questions. But we are getting better at framing those questions and probably better at answering them and making some linkages. Numerous speakers pointed out the importance of the political reality and feasibility and the practical implementation aspects of everything we are talking about. I am encouraged to hear that awareness growing among all of us with all of our different backgrounds. We have also heard from different speakers that we need to make better use of the data that we already have. There have been a lot of expensive and extensive studies. We need to take advantage of what we have, put it together, and step back and see the connections that we may be able to make based on this work. We also need to take advantage of the fact that every time we make a public policy decision we are setting an experiment in motion; we usually don't set it up to take advantage of the data that might result. Maybe we should be thinking explicitly in terms of what we can learn from our mistakes and assume that we will make mistakes.

There have been lots of indications in many presentations that a well-designed monitoring program is important. Also, repeatedly in different ways, we heard many of you say we are dealing with a complex system where both natural and unnatural things are happening over the long term. We are struggling with our own short-term efforts to understand those effects. In addition, we are trying to understand some short-term effects that are occurring. We need to be cognizant of that.

Where to from here? The premise of both the meeting and the committee's work seems valid after thinking about it through this conference. We clearly do need some kind of ongoing process or processes to set priorities, to coordinate, communicate, and to find the funding to do the things that need to be done. We have all also agreed that money doesn't grow on beaches. Both Jim Waldo and Dean Heath in their own ways talked about how we should be thinking about how to get scientists working on things together--about collaborative processes as well as peer review as ways of reducing the battle of the experts when it comes to controversial subjects. We need to figure out ways of developing common understandings when it comes to making some very tough decisions, regulatory and otherwise. One thing Jim Waldo said about the research foundation idea is something I would appreciate all of you thinking a little bit more about: maybe we need to make some incremental steps toward the goal. What might those incremental steps be; how can we structure the process? Those are some practical thoughts that we should take to heart because we are unlikely to get the instant endowment that would make it all possible in the next month or year.

Where to, specifically? I hope you will take the time to evaluate this conference and provide us your comments either today or in the mail. I hope that you will take the time to comment on the research committee report and keep your eye on the Authority because we will be revisiting these issues and coming to some closure. Over the course of this summer we will be issuing a draft revised management plan which will address this issue, and we will make some final decisions toward the end of the summer.

Before we close, I would like to acknowledge the hard work of the Authority staff in putting this meeting together. As you can imagine, virtually the entire staff was involved at one point or another, but I would specifically like to commend Kathy Callison and Marcia Lagerloef for the work that they have done in organizing this meeting.

Thank you all for coming.

POSTER SESSION

Sampling Strategies for a Reconnaissance of Petroleum Contamination in the Port Angeles-Sequim Bay Region, Washington: *W.H. Pearson, J.Q. Word, J.R. Skalski, R.B. Lucke, J.M. Gurtisen, P. Wilkison, Battelle Marine Research Laboratory, Sequim, WA.*

PAH Analysis in Marine Sediment: Interlaboratory Precision: *W.D. MacLeod, D.W. Brown, S-L Chan, and U. Varanasi, Environmental Conservation Division, Northwest & Alaska Fisheries Center - NOAA, Seattle, WA.*

Toxic Chemicals: Biological Effects: *Donald Norman, Institute of Wildlife Toxicology, Huxley College, Bellingham, WA.*

The Water Quality of Padilla Bay: *Paul Cassidy, Shannon Point Marine Center, Anacortes, WA.*

Tidal Residual Eddies in the Strait of Juan de Fuca and Approaches: Comparison of a Numerical Model with Field Observations: *C.A. Coomes, C.C. Ebbesmeyer, P.B. Crean, Evans-Hamilton, Inc., Seattle, WA.*

Potential Transport of Near Surface Contaminants Between Canada and the U.S. Within the Strait of Juan de Fuca: *J. Cox, Evans-Hamilton, Inc., Seattle, WA.*

The Puget Sound Environmental Atlas: Summary and Future Direction: *G. Rosenthal, Evans-Hamilton, Inc., Seattle, WA.*

Water Quality Within and Near the John Wayne Marina, Sequim Bay, Washington: *C.W. Apts, Battelle Marine Research Laboratory, Sequim, WA.*

Post-Glacial Sedimentation in Puget Sound (or the Container: History and Hazards): M.L. Holmes, U.S. Geological Survey, University of Washington; R.E. Sylvester, Williamson and Associates, Inc., Seattle, WA; and R.E. BURNS, Seattle, WA.

Statistical and Experimental Design Issues for Water Quality Monitoring Studies in Puget Sound: *L.L. Conquest, K.M. Lohman, Center for Quantative Science, University of Washington, Seattle, WA.*

Timelapse Photography in the World of the Geoduck: *D. Jamison, R. Heggen, J. Lukes, Marine Research & Development Center, Department of Natural Resources, Olympia, WA.*

Public Understanding of Local Urban Wetlands: *S. Schauman, M. Young, Department of Landscape Architecture, University of Washington, Seattle, WA.*

Remote Sensing Inventory of the Seagrass Meadow of the Padilla Bay National Estuarine Research Reserve: *H.H. Webber, Huxley College, Bellingham, WA.*

Trace Organic Chemical Phase Associations in Puget Sound Sediments: *R. Carpenter, M. Strom, S. Socha, School of Oceanography, University of Washington, Seattle, WA.*

Ocean Data Evaluation System; Puget Sound Activities: *Tetra Tech, Inc., Bellevue, WA.*