King County Combined Sewer Overflow Water Quality Assessment for the Duwamish River and Elliott Bay

Appendix C: ISSUE PAPERS

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List of Acronyms

AIDS	Acquired Immune Deficiency Syndrome
CDC	Center for Disease Control
COPC	Constituent of potential concern
CSO	Combined sewer overflow
DO	Dissolved oxygen
EEC	Estimated exposure concentration
FDA	Food and Drug Administration
IDD	Insulin-dependent diabetes
IgA	Immunoglobulin antibody
NRDC	National Resource Defense Council
PAHs	Polycyclic aromatic hydrocarbons
PCBs	Polychlorinated biphenyls
RNA	Ribo-nucleic acid
RWSP	Regional Wastewater Services Plan
SCUBA	Self-contained underwater breathing apparatus
WERF	Water Environment Research Foundation
WSDOE	Washington State Department of Ecology
WSDOH	Washington State Department of Health
WSDOF	Washington State Department of Wildlife
WQA	Water Quality Assessment

INTRODUCTION

The King County Combined Sewer Overflow Water Quality Assessment is an assessment of the ecological and human health risks from exposure to pollutants in the Duwamish River and Elliott Bay, and what part of these risks are from combined sewer overflows (CSOs). This assessment is described in four volumes of which this report is the third. This report is a compilation of issues papers presented previously to our stakeholders that identify the critical scientific issues that we face in conducting a risk assessment of this nature, such as what types of data we have available to us to conduct the risk assessment. Most importantly, the issue papers identify the strengths and weaknesses inherent in the current state of the science and the approaches and methods used by the scientific and regulatory communities to allow us to assess environmental risks.

Nine issue papers are compiled in this report and are the following:

Issue Paper 1. Study Area Description

This issue paper describes the Duwamish River and Elliott Bay Water Quality Assessment study area. Its purpose is to provide a context for evaluating risks that may be identified by the Water Quality Assessment. Because the study area lies within a highly urbanized watershed, it is affected by a variety of physical, chemical, and microbial stressors, only some of which are associated with combined sewer overflows (CSOs). It is the task of the Water Quality Assessment to describe and quantify the risks to aquatic life, wildlife and humans that use the resources of the study area, and to determine how controlling CSOs will change the risks.

Issue Paper 2. Aquatic Life and Wildlife Site Use

This paper summarizes the use of the study area by aquatic life and wildlife. Its purpose is to understand how important species (receptors) in the study area are exposed to chemical contamination in the Duwamish River and in Elliott Bay. The important receptors that were evaluated include: salmon (e.g. outmigrant juveniles), resident fish (e.g. English sole), the benthos, shore birds (e.g. spotted sandpiper), wading birds (e.g. great blue heron), raptors (e.g. bald eagle), and aquatic mammals (e.g. river otter).

Issue Paper 3. Human Site Use

This paper summarizes the human use of the Duwamish River and Elliott Bay, highlights uncertainties and issues regarding the level and extent of use, and proposes methods for assessing the levels of human exposures in the risk assessment. The risks being assessed are those associated with pathogens and chemicals that may occur in CSO discharges, surface water, sediments, and aquatic organisms.

Issue Paper 4. Pathogens

The purpose of this paper is to describe the issues associated with assessing the risk of infectious disease in people as a result of using the Duwamish Estuary for recreational and commercial activities. These issues apply whenever there is a need to predict the likelihood of illness occurring as a result of environmental exposure. This paper will describe the challenges for this particular project and the approach that will be taken to assess the risks of infectious disease resulting from swimming, wind surfing, scuba diving (and other direct exposure activities), and/ or eating fish or shellfish from the study area.

Issue Paper 5. Physical Stressors

This issue paper discusses what risk physical disturbances, caused by CSO discharges, pose to aquatic life. Physical disturbances include erosion and sedimentation that may occur from increased river flow during runoff and from CSO discharges. Risks to aquatic life from these disturbances have rarely been evaluated in risk assessments, which have traditionally focused solely on chemicals released to the environment by human activities. In the Water Quality Assessment Problem Formulation, the following effects caused by physical disturbances, termed stressors, were identified:

Issue Paper 6. Aquatic and Wildlife Toxicology

To assess ecological risks from chemicals in the Duwamish River/Elliott Bay Water Quality Assessment, we need to identify the minimum amount of a chemical present in water or sediment that would harm invertebrates or fish (the “effects level”). We will identify the effects level for each chemical being evaluated using the basic principles of toxicology, as well as a series of assumptions founded on these principles. This paper presents these elements of toxicology and explores the basis for the assumptions we will make in the risk evaluation.

Issue Paper 7. Human Health Toxicology

To assess risks to human health from chemicals in the Duwamish River and Elliott Bay, the Water Quality Assessment requires two types of information: (1) the amount of chemical that people are exposed to when engaged in different activities (exposure), and (2) the amount of chemical taken into the human body that is considered to be either “safe”, or associated with a defined level (generally small) of health risk. This paper discusses the assumptions and uncertainties of the information used to establish “safe” doses as defined using accepted principles of toxicology.

Issue Paper 8. Interfacing the Model with the Risk Assessment

This paper describes the need for a model to predict risks to aquatic life, wildlife, and humans in the WQA study area. Also described is if the chosen model meets the needs of the WQA, what specific issues and information are needed to determine risks, and what extra steps will be taken to address risks at critical habitat areas.

Issue Paper 9. Risk Predictions and Aquatic Community Responses

This paper discusses three specific issues associated with this assessment of risks to the aquatic community. These issues are:

- How can predictions of risk to the aquatic community be interpreted?
- How quickly can aquatic communities recover after a disturbance?
- What are the consequences of the loss of species from a community?

Issue Paper 1: Study Area Description

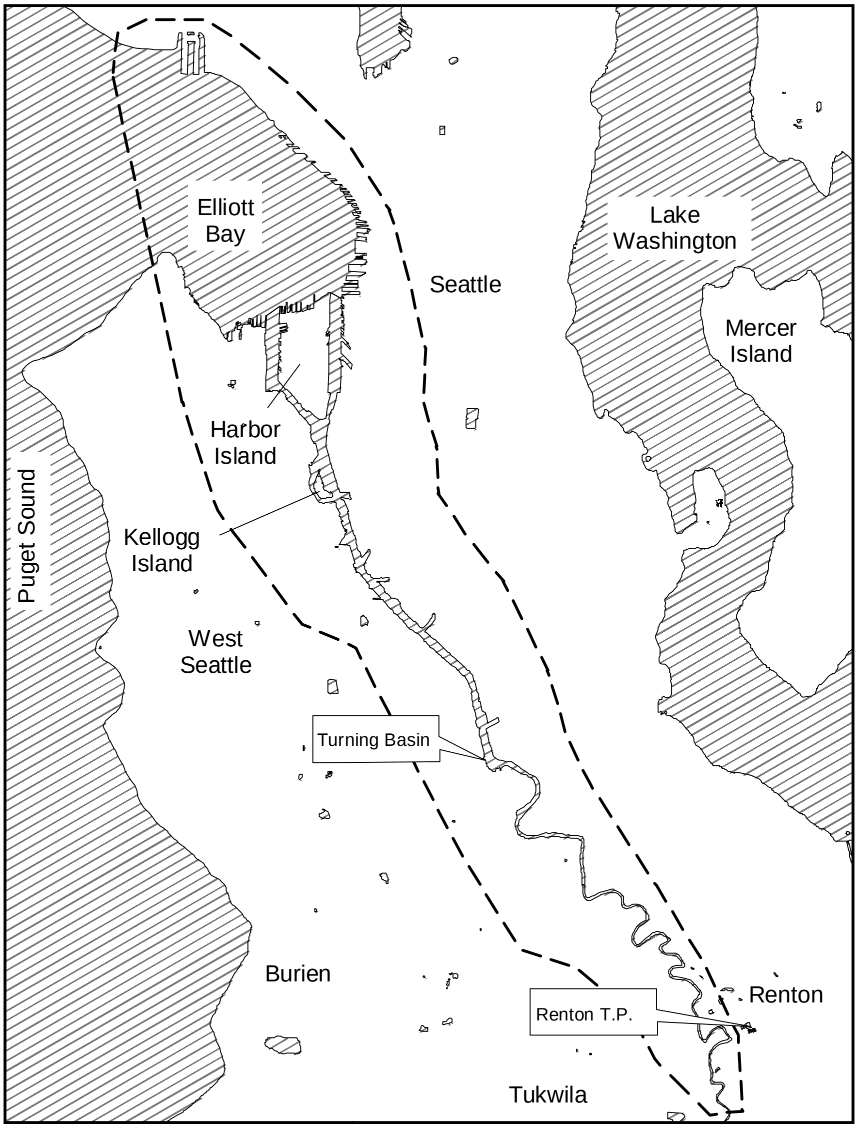
This issue paper describes the Duwamish River and Elliott Bay Water Quality Assessment study area. Its purpose is to provide a context for evaluating risks that may be identified by the Water Quality Assessment. Because the study area lies within a highly urbanized watershed, it is affected by a variety of physical, chemical, and microbial stressors, only some of which are associated with combined sewer overflows (CSOs). It is the task of the Water Quality Assessment to describe and quantify the risks to aquatic life, wildlife and humans that use the resources of the study area, and to determine how controlling CSOs will change the risks.

This paper first describes the Duwamish River/Elliott Bay study area in terms of its physical characteristics (boundaries, bathymetry, flow dynamics, and sediment types). Next, it presents the extent of flow diversion and development that has occurred as this system has become urbanized. Finally, the effects these changes have had on the major aquatic resources of the study area are addressed. This account is not meant to be comprehensive, rather it highlights the information that is most pertinent in understanding the present state of the estuary. Additional information about the study area and its resources can be found in King County's Duwamish River/Elliott Bay Problem Formulation and Planning Document (King County 1997a).

1.1 Duwamish River and Elliott Bay Study Area

The study area includes the Green-Duwamish River from just upriver of the Renton Sewage Treatment Plant downstream to where it flows into Elliott Bay, a distance of about 15 miles (Figure 1). It also includes that portion of Elliott Bay east of a line drawn north from Duwamish Head to Magnolia Bluff. The study area can be considered an estuarine system, that is, an aquatic system that exhibits both marine and freshwater characteristics. The upriver portion of the study area is primarily a freshwater river with tidal influence while the seaward boundary of the study area in Elliot Bay is primarily marine with a variable freshwater layer, especially in the winter months during periods of higher river flow.

The lower Duwamish River is a highly industrialized salt-wedge estuary. This area is influenced by river flow and by tidal effects. As is typical of salt-wedge estuaries, the Duwamish is characterized by a sharp interface between freshwater outflow at the surface and salt-water inflow at depth. The layer of salt water near the river mouth occupies most of the water depth, but tapers towards the head (upriver portion) of the estuary. The location where salt-water intrusion tapers to zero is called the toe of the salt wedge. In the Duwamish River, the toe of the salt wedge is located approximately 7 miles upstream of the river mouth in the vicinity of the upper turning basin but can extend upriver to near Tukwila (another two miles) during low-flow (dry) years. It is because the salt wedge extends so far up river that we have assumed that the ecosystem within our study area



Water Quality Assessment Area - - - - -

King County/Duwamish/55-1521-27(07) 1/99

Figure 1.
Water Quality
Assessment Study Area

consists mainly of marine species. This assumption is supported by the finding of bay mussels, *Mytilus trossulus,* as far up river as the Norfolk CSO (J. Strand, King County Department of Natural Resources, Seattle, Washington, personal observation). The fish community of the Duwamish River up to 7 miles from the river mouth is also known to be marine dominated because of the presence of starry flounder and staghorn sculpin (Matsuda et al., 1968). Only above 10 miles from the river mouth are some freshwater fish species found, e.g. longnose dace, speckled dace (Matsuda et al., 1968). Starry flounder, threespine stickleback, and prickly sculpin, all species found in the lower Duwamish River, are found as far up river as the Renton Sewage Treatment Plant (12 miles from the river mouth).

The lower portion of the river, below the upper turning basin (6 miles from the mouth) has been straightened, dredged and armored with rocks in many areas to facilitate navigation and industrial development. The depth of the river portion varies from approximately 50 feet near the river mouth at Harbor Island to less than 3 feet in the upper river portion of the study area. Bottom sediments in the river range from sands to mud, depending on the sources of sediment and the current speeds. The flow of the river is largely controlled by releases from the Howard Hanson Dam, located in the upper portion of the Green River watershed). Summer flows in the river, gauged at Auburn, are in the range of 250 cubic feet per second (cfs) (L. Fuste, U.S. Geological Survey, Tacoma, Washington, personal communication). Winter flows average about 1500-2000 cfs, with peaks to more than 10,000 cfs during storm events.

Elliott Bay is approximately 8 square miles and is located on the eastern shore of central Puget Sound. The bay opens to the main basin of Puget Sound to the west. Depths on the western Elliott Bay boundary are in the range of 450 to 540 feet, while depths close to the developed Seattle waterfront are 30 to 60 feet. A deeper submarine valley in the center of Elliott Bay leads to the main basin of Puget Sound. A branch of this valley runs south along the east side of Duwamish Head and is about 240 feet deep. Natural shorelines with a gradually sloping intertidal zone are located along the western and northern shores of the bay. Sediments in these areas range from gravel and cobbles in nearshore areas to fine muds in the deeper areas.

The open portion of Elliott Bay is dominated by Puget Sound marine water masses, with the freshwater layer from the Duwamish River limited to the upper five meters, or about five percent of the water column. In winter this layer can be clearly seen in Elliott Bay from its brown sediment color. The river water is mixed with incoming Puget Sound water and enters the greater Puget Sound circulation. Sediment falls from the surface layer to the bottom in both Elliott Bay and in Puget Sound.

1.2 Urbanization of the Duwamish River and Elliott Bay

Prior to settlement in the mid-1800s, the nearshore environment of Elliott Bay once consisted of 2,100 acres of marsh and eelgrass (Stober and Pierson 1984). The Duwamish River flowed freely meandering through wetland swamps and tidal marshes before emptying into Elliott Bay just east of what is now West Seattle. Much of the area from West Seattle east and north to Pioneer Square was mudflats and marshes. The floodplain and river shifted often in the narrow valley plain where flooding was a common natural occurrence. The historic Duwamish River drained 4,254 square kilometers of watershed (Warner and Fritz 1995) and was fed by the Green River, Cedar River, White River, and from Lake Sammamish and Lake Washington through the Black River.

By 1900, about 20 percent of the tidal marshes and over half of the wetland swamps were filled to create 800 acres of "usable land" (Sato 1997). Seattle's early waterfront was created by filling 300 acres of shallows and flats with dredged materials and other soils. The river still meandered through the remaining wetland swamps and tidal marshes and swept around Kellogg Island, then a quarter-mile wide, 200-acre intertidal marsh and wetlands surrounded by shellfish beds (Sato 1997).

The White River was diverted to the Puyallup River in 1906 by a high water event, possibly with human assistance (Warner and Fritz 1995). In 1916, the Black River was drained with the construction of the Hiram H. Chittenden Locks. The locks replaced the Black River as the outlet for Lake Washington by lowering the lake from an average of 29.8 feet above mean lower low water (MLLW) of Puget Sound to the existing lake elevation of 21 feet above MLLW (Chrzastowski 1983). At this time, the Cedar River was diverted permanently to the Lake. Currently the Duwamish River only drains the Green River watershed (475 square miles).

In a project begun in 1910 and completed in 1917, the lower 10 miles of the Duwamish was converted to a waterway four and a half miles long with three turning basins along its length (Sato 1997). It connected with Elliott Bay by two 750-foot wide channels (now the East and West Waterways), 35 feet deep, and each a mile long. A navigational channel is maintained all the way to the uppermost turning basin (river mile 6) resulting in deepwater habitat where none existed before. Most of the shoreline was armored with vertical bulkheading, rock riprap, and piers. The uplands were nearly all converted to industrial use.

In 1912, the Tacoma Headworks Dam (a water diversion dam) was built at river mile 60. This was a 17-foot high dam from which water was diverted to a water pipeline extending to the City of Tacoma. The U.S. Army Corps of Engineer's Howard Hanson Dam was built in 1961 at river mile 64. This dam was built for flood control and also for preservation of fish life by providing higher flows when river flows were naturally low in the summer and fall (Sato 1997). Neither dam, however, was built with fish passage facilities, eliminating access to an estimated 107 miles of historic fish habitat. The diversion of the Renton Sewage Treatment Plant effluent in 1986 further decreased summer flows to the Duwamish River (Warner and Fritz 1995).

1.3 Loss of Tidal Marshes and Mudflats

Over the past 125 years the drainage area of the Duwamish River has been reduced by about 70 percent due to development and flow diversion. Most (98 percent) of approximately 1,270 acres of tidal marsh and 1,450 acres of flats and shallows, and all of about 1,250 acres of tidal wetland, have been eliminated (Blomberg et al 1988). Remnant tidal swamps and mudflats account for only 5 and 55 acres, respectively (Leon 1980). Sections of natural shoreline only occur in the Duwamish River above the head of navigation, located at approximately river mile 6 (Tanner 1991).

As shown in Figure 2, Kellogg Island is the largest remnant of intertidal habitat remaining in the Duwamish River Estuary (Tanner 1991). Habitat associated with the island includes high and low marsh, intertidal flats, and filled uplands (Canning et al. 1979). The intertidal habitat that remains is important for the survival of juvenile salmon, other predator fish, birds, and mammals that feed on invertebrates and small fish found in this reach of the river.

Kellogg Island also provides important nesting and feeding habitat for waterfowl, shore birds, and other birds (song birds). This function was added only recently when in 1974 the Port of Seattle deposited 60,000 cubic feet of dredge spoils on the island (Sato 1997). A mixture of introduced and native plant and tree species rapidly colonized the 17-acre island.

Other small intertidal areas occur in the estuary as marsh and unvegetated marsh habitat (Figure 2). These areas have become the focus of important habitat restoration activities recently undertaken by the Muckelshoot Tribe, the Port of Seattle, King County, and Coastal America, the latter a federal intergovernmental program with participation by the U.S. Army Corps of Engineers, U.S. Fish and Wildlife Service, and the U.S. Environmental Protection Agency (U.S. EPA). The objectives of these projects include the removal of rock riprap and overwater wharf structures, restoration of natural tidal flow, natural colonization of native wetland plants, and use by juvenile salmonids (Cordell et al. 1996).

1.4 CSOs

Combined sewer overflows (CSOs) are discharges of untreated sewage and stormwater released directly into surface waters during periods of heavy rainfall (King County 1995). Combined sewers, those that carry sanitary sewage and storm runoff in a single pipe, are found in much of metropolitan Seattle. Because combining systems was the standard engineering practice, all of Seattle's sewers built from 1892 until the early 1940s were combined sewers. As newer sewers have been installed in Seattle, storm water has been separated from household, commercial, and industrial wastewaters.

CSOs serve as safety valves for the sewage treatment system. In combined systems, the trunk sewers and interceptors have fixed capacities. During periods of heavy rainfall, wastewater volumes may exceed the capacity of the sewer pipes to convey the wastewater to the treatment plant. To prevent damage to the system and to prevent sewers from backing up into homes, combined sewers are designed to overflow. Typically, overflows are designed to discharge to rivers and marine waters where the flushing action of tides and currents can disperse pollutants.

City of Seattle and King County (formerly Metro) overflows occur within the study area in both the Duwamish River and Elliott Bay. Other overflows occur in Lake Washington, Lake Union, and the Ship Canal. The locations of CSOs in the study area are shown in Figure 2. From 1981-1983, nearly 2.4 billion gallons of untreated sewage were discharged from this system each year. As a result of control efforts, this volume was reduced to 1.8 billion gallons per year by 1994 (King County 1995).

1.5 Characteristics of Stressors

Four principal types of stressors are identified as having an effect on the study area. They are physical disturbance, toxic chemical additions, other changes in water quality, and microbial contamination.

1.5.1 Physical Disturbance

Changes in river flow patterns and increases in discharge from CSOs and storm drains occur during the wet season and result in erosion and sedimentation, both of which can adversely affect available biological habitat. Each process is related to event-specific discharge, the depth of the channel into which the runoff or discharge occurs, and the particle size of sediments present in the bed or added to the flow by resuspension or the discharge. Also, periodic dredging of the navigational channel and new construction along the waterway result in the direct destruction of habitat in the study area.

These kinds of physical disturbances result in changes in physical, chemical, and biological conditions that affect the survival, growth, and reproduction of a wide variety of organisms, both plants and animals. Benthic or epibenthic invertebrates (organisms living in or on the bottom) are particularly vulnerable and effects vary in severity from minor and temporary to severe and permanent. For example, dredging and construction effects tend to be more severe and longer-term whereas changes in flow and erosion and sedimentation that are associated with CSO and stormwater discharges result in minor changes that are temporary and often seasonal in nature. Resuspension of chemically contaminated sediments during erosion (scouring) can result in the re-release of potentially toxic chemicals into the water column.

1.5.2 Toxic Chemical Additions

Contamination of estuarine waters is also a consequence of urbanization and industrialization. Potentially toxic chemicals entering the Duwamish River and Elliott Bay mainly include: polynuclear aromatic hydrocarbons (PAHs) from fuel constituents; polychlorinated biphenyls (PCBs) from transformer coolants; organic solvents, phthalates, phenolics, organometalloids (e.g. TBT), and metals (arsenic, cadmium, copper, lead, mercury, zinc) associated with industrial practices. Mercury also enters the Duwamish River from natural (geologic) sources in the Green River. Chemicals enter the study area from both point sources such as permitted industrial discharges, treatment plants, storm water, CSOs, accidental spills, leaks, and illegal dumping; and nonpoint sources such as runoff, atmospheric deposition, and groundwater. Pesticides can be found in the study area in trace quantities and originate from agricultural practices in the upper Green River watershed.

Intensive surveys of sediments conducted by Metro (Romberg et al. 1984), the U.S. EPA (1988), and the State of Washington (WSDOE 1996), have reported that both metals and organic chemicals contaminate sediments throughout out much of the Duwamish River and along the Seattle waterfront. Twenty-five areas have significantly elevated (exceed Sediment Quality Criteria) concentrations of toxic chemicals and are included in the Contaminated Sediment Site List (WSDOE 1996). In the Duwamish River and along the Seattle waterfront, PAHs in the sediments can reach 10 ppm dry weight. PCBs often reach 0.3 ppm dry weight.

CSOs discharge varying amounts of inorganic and organic chemicals during overflows. Stuart and Cardwell (1987) found seven metals (arsenic, cadmium, copper, lead, mercury, silver, and zinc) and 20 organic chemicals or chemical groups (12 PAHs, 5 phthalates, chloroform, trichloroethane, and TE-chloroethane). During King County's Water Quality Assessment, seven metals (arsenic, cadmium, copper, lead, mercury, nickel, zinc), and four organic chemicals or chemical groups (1,4 dichlorobenzene, 4-methylphenol, phthalates, and PAHs), were found routinely in CSOs discharging to the Duwamish River (S. Mickelson, King County Environmental Laboratory, personal observation).

Chemical contamination of sediments below CSOs in the study area has been studied by Armstrong et al. 1980-1981; Romberg et al. 1984; 1995, and King County 1996, 1997b. These reports suggest that chemicals discharged from CSOs are adsorbed to sediment particles that settle to the bottom at varying distances from the end of the pipe depending on particle size and hydrodynamics. The area of deposition is known as the footprint and can vary in size from 1000to 5000m^2.

1.5.3 Other Changes in Water Quality

Freshwater runoff and CSO discharges can lower the salinity of receiving waters. CSOs also have the potential to lower dissolved oxygen and pH if their effluents are high in nutrients and organic material. Additionally, CSO discharges may be warmer than the receiving waterbody. Altered salinity, dissolved oxygen, pH and temperature regimes have the potential to affect mostly aquatic species.

1.5.4 Microbial Contamination

Microbial contaminants enter the upper-river portion of the study area in surface runoff from agricultural areas and in groundwater contaminated by failed septic systems. CSOs are the primary source of untreated domestic wastewaters in the lower river and bay. Microbial contaminants of most concern are human pathogens including protozoa,

bacteria, viruses, and possibly helminths (tapeworm, round worms). Munger (1983) determined that pathogenic bacteria including Mycobacteria, Salmonella, and Yersinia regularly occur in King County sewage sludge.

In microbiological surveys conducted by Heyward et al. (1977), fecal coliform levels were found to be 77 times higher in butterclams than in water on King County beaches near King County's wastewater treatment plants. Commercial harvesting of shellfish has not been allowed in Elliott Bay for many years because of high fecal coliform bacteria counts (Stober and Pierson 1984). The King County Health Department and the Washington Department of Health have also recommended against recreational collections of both fish and shellfish from urban shorelines in their jurisdictions (Washington Department of Health 1993). Microbial contaminants persist for varying lengths of time in water, sediments, and shellfish and pose a potential human health hazard.

1.6 Implications for the Water Quality Assessment

Stressors in the study area take on a number of forms, e.g., physical, chemical, and biological, and are associated with a variety of sources, e.g., dredging, construction activities, industrial practices, CSOs, separated storm drains, and agricultural activities. Stressors arise locally, regionally, and even globally. It is within this context that we will describe and quantify the risks to aquatic life, wildlife, and people, and determine how risks will be changed by CSO control.

A particular challenge to interpreting risk predictions is that they are based on exposures to single stressors, while in reality, exposures and ensuing effects involve many stressors acting together. Unfortunately, interactions among multiple stressors are complex and are not well understood. They can act additively, synergistically (multiplicative) or antagonistically (less than additive). They do not generally allow us to combine a set of single stressor risk predictions into a single multiple stressor prediction of risk. This makes it difficult to determine which stressors may be limiting for aquatic life, wildlife, and people that use the resources of the study area. The quantitative assessment of risks from multiple stressors is a topic currently being studied and debated but no one yet has developed a rigorously quantitative procedure to follow. The move to developing basin, watershed, and ecosystem approaches holds some promise but at present, the best we can do is develop single-stressor based risks, then rely on professional judgment to qualitatively describe the overall risk

Another challenge is to be able to estimate incremental risks from different sources of stress; e.g., estimate the benefits of CSO control, which is a hypothetical scenario where no CSOs discharge to the study area, and one that can not be ascertained from field sampling. We have addressed this need by developing a computer model that provides a detailed, mathematical description of the movement of water and sediments in the study area over time. The model also describes the physical, chemical, and biological fate of chemicals both in the water column and in the sediments, which is used to estimate exposure to important receptors (aquatic life, wildlife, and people).

The model is calibrated with actual chemical loadings from various sources, e.g., CSOs, storm drains, sediments, and boundary conditions from the Green River and Elliott Bay. Future data could be collected and used to verify the models' predictions, refine its calibration, and test the validity of underlying principles. The model can be run in different modes and over different time frames depending on which questions are being answered. For example, if interest is in predicting existing or baseline conditions of exposure then the model will be run for a year with all sources of chemicals contributing to the simulation. If interest is in predicting exposure associated with just CSOs, the model is rerun without CSOs contributing to the simulation. The difference in the two model runs represents the influence that CSOs have on exposure, or in the context of our study, the effect of controlling CSOs.

The model is assumed to be the best available tool for predicting the transport and fate of chemicals in the Duwamish River and Elliott Bay ecosystem where field data are unavailable. It is the only way to predict conditions that have not occurred (determine the benefits of CSO control). It is also the only way follow the legacy of historical contamination in the sediments below CSOs as natural recovery will undoubtedly take years, perhaps decades, after elimination of the CSO discharge (overflow).

1.7 Summary

- The Duwamish River and Elliott Bay have changed dramatically since the area was first settled.
- Urbanization (development, flow diversion, industrialization has resulted in a significant reduction of available habitat for aquatic life and wildlife. The natural habitat that remains, e.g., Kellogg Island and other remnants, may be critical to the survival of lower Duwamish River ecosystem, particularly salmonid juveniles and their prey base.
- Stressors in the study area take a number of forms, e.g., physical, chemical, microbial, and are associated with a variety of sources, e.g., dredging, construction activities, industrial practices, sewage treatment plants, CSOs, storm drains, and agricultural activities.
- It is within a highly disturbed watershed that we will describe and quantify the risks to aquatic life, wildlife, and humans that use the resources of the study area, and determine how the risks will be changed by CSO control.

- A particular challenge to risk predictions is that they are generally based on exposures to single stressors, while in nature, exposures and ensuing effects involve interactions of many stressors. At present, the best we can do with single stressor risks is to apply considered professional judgement to qualitatively describe the overall risk.

- Another challenge is to be able to estimate incremental risks from different sources of risk such as chemicals from CSOs. We have addressed this need by developing a computer model that provides a mathematical description of the transport and fate of chemicals and microbial contaminants in the study area. This is the best approach to use where: 1) field data are unavailable, 2) there is need to predict conditions that have not occurred, e.g., determine benefits of CSO control, and 3) there is need to follow the legacy of historical contamination, as natural recovery will take years or decades, after elimination of CSO discharge.

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Issue Paper 2: Aquatic Life and Wildlife Site Use

This paper summarizes the use of the study area by aquatic life and wildlife. Its purpose is to understand how important species (receptors) in the study area are exposed to chemical contamination in the Duwamish River and in Elliott Bay. The important receptors that were evaluated include: salmon (e.g. outmigrant juveniles), resident fish (e.g. English sole), the benthos, shore birds (e.g. spotted sandpiper), wading birds (e.g. great blue heron), raptors (e.g. bald eagle), and aquatic mammals (e.g. river otter).

2.1 Salmon Outmigrants

Salmon were selected as a receptor because of their importance to commercial interests, recreational pursuits, cultural values, and their significance to the aquatic community.

Four species of salmon (Chinook, chum, coho, pink) use the study area but generally only for passage. They pass through the study area as adults in summer or in fall when ascending the river to spawn. They also pass through the study area in spring or early summer as fry or juvenile outmigrants en route to Puget Sound and eventually to sea.

Chinook and chum salmon appear to have a greater estuarine reliance than the other species. The juvenile, or outmigrant smolt, is recognized as the most sensitive lifestage found within the study area.

Migration times of juvenile and adult salmon found in the study area are presented in Table 2-1.

Table 2-1. Approximate Dates of Migrations of Salmon in the Green/Duwamish River

Species	Extent of Juvenile Outmigration	Peak of Juvenile Outmigration	Adult Return Migration
Chinook Salmon	Mid-February to Early September	Mid- to Late May	Mid-June to Early November
Chum Salmon	Late February to Mid-July	Mid-March to Mid-April	November to Mid-January
Coho Salmon	Mid-February to Mid-July	Early to Late May	August to Late January
Pink Salmon	Early February to Mid-July	Mid-March to Mid-April	August and September

Sources: Wydoski and Whitney 1979; Grette and Salo 1986; Heard 1991; Warner and Fritz 1995

2.1.1 Chinook Salmon (*Oncorhynchus tshawytscha*)

Most of the wild and hatchery Chinook of the Green River are the fall variety (fish that enter freshwater July to September). A few spring Chinook (fish that enter freshwater May and June) may still use the Green River Gorge area for spawning. At present, most adults enter the drainage from mid July to mid November. Spawning occurs late August to early December with hatching in mid to late February. Wild Chinook spend from several days to several months rearing in freshwater before moving into the estuary (Grette and Salo 1986). Juveniles do not immediately go to sea upon entering the estuary, but appear to linger over a period of two to three months and sometimes longer (Warner and Fritz 1995). Juvenile Chinook outmigrants are present in the Duwamish Estuary from mid February to early September. Peak abundances occur in mid to late May, which is concurrent with releases from the Washington Department of Fish and Wildlife Green River (Soos Creek) Hatchery.

Although natural production of Chinook salmon in the Green River watershed is enhanced by hatchery production from both the Washington Department of Fish and Wildlife Green River Hatchery at Soos Creek and the Muckelshoot Tribe Hatchery at Keta Creek, the numbers of wild Chinook returning to the Green River watershed in recent years have continued to decline.

In 1997, Chinook returns to the Green River were too low to sustain a Muckelshoot Tribal fishery (J. Rector, Muckelshoot Indian Tribe, personal communication). At the present time there are no listings under the Endangered Species Act for salmonids in the Puget Sound Basin but this could soon change. Because of continuing declines in naturally spawning (wild) stocks, Chinook in the Green River and in other Puget Sound watersheds have a high potential for listing by the National Marine Fisheries Service (WDFW, 1996).

2.1.2 Chum Salmon (*Oncorhynchus keta*)

Both wild and hatchery derived chum salmon migrate through and spawn in the Green River. Most are hatchery fish reared by the Muckelshoot Indian Tribe at Keta Creek. Returning adult salmon can enter freshwater anytime from mid September to mid January. Spawning occurs from mid November to late January. After hatching, juveniles move directly to the estuary where they can be found in significant numbers from late February to mid July. Abundances in the estuary tend to peak in response to high water events and do not necessarily respond to releases from the Muckelshoot Keta Creek Hatchery (Warner and Fritz 1995). Both Chinook and chum salmon juveniles occur in the lower estuary during the time when most combined sewer overflows (CSOs) occur, October through April.

2.1.3 Coho Salmon (*Oncorhynchus kisutch*)

Both wild and hatchery adult coho enter the Green River between late October through December (Grette and Salo 1986). Spawning of wild fish occurs soon after but may extend into January. The young hatch in six to eight weeks. Juveniles rear in freshwater for up to eighteen months before outmigrating. Williams et al. (1975) reported that coho outmigration occurred from mid July to mid August, which suggests that coho move more directly to sea than do Chinook and chum. In 1994, Warner and Fritz (1995) found outmigrating coho in low numbers from mid February through mid July. The outmigration peak occurred in May and was correlated with releases from both the Green River and Keta Creek Hatcheries in April. Although based on outmigration of hatchery fish, these data tend to confirm that findings of Williams et al. (1975) indicating that coho spend little time in the estuary and migrate directly to saltwater. The few coho captured by Warner and Fritz (1995) in February were from a hatchery release ten months earlier above Howard Hanson Reservoir.

2.1.4 Pink Salmon (*Oncorhynchus gorbuscha*)

Pink salmon were all but eliminated from the Green River with separation of the White River in 1906. A few fish return (possibly strays) every other year (odd years only) but large numbers of pinks have not returned since the 1930s. As a consequence, a reliable production estimate is not available. Warner and Fritz (1995) found outmigrant pink salmon in their 1994 surveys and reported that pink salmon in the Green River appeared to follow the same periodicity as chum salmon. Based on what is known in general about pink salmon life history, adults are thought to enter the Green River in August and September and spawn shortly thereafter. Upon hatching in six to eight weeks, fry should outmigrate over a period extending from mid-February to mid-July with peak outmigration occurring between mid-March to mid-April (Heard 1991).

2.1.5 Food Habits of Salmon in the Lower Duwamish River

Parametrix (1990) found that *Cumella vulgaris* was the most important prey item for both chum and Chinook salmon juveniles at Kellogg Island in the lower Duwamish River. Also important in the diet of Chinook juveniles were gammarid amphipods, mysids, insects, and larval fish. Leon (1980) suggested that the Kellogg Island intertidal habitat, in particular, was critical to survival of chum salmon. This species as well as Chinook pass through the lower Duwamish when their survival is related to their growth, which is dependent upon an abundant food supply. The larger the outmigrant smolt when entering marine waters, the greater the probability of survival to maturity and return to their natal stream (Warner and Fritz 1995). Elsewhere (Snohomish estuary), Conley (1977) found that juvenile salmon and staghorn sculpin feed on the benthic amphipod, *Corophium*, which also occurs at Kellogg Island. Simenstad and Kinney (1978) determined that harpacticoid copepods and amphipods made up the largest part of juvenile chum salmon diet in Hood Canal.

Aspects of salmon biology influencing their exposure to CSO stressors are:

- Juveniles of all salmon species outmigrate though the study area over a period from late winter to early summer. Most sewer overflows occur between October and April.

- Juvenile Chinook and chum salmon demonstrate greater estuarine reliance than other salmon species and may spend two to three months in the study area before going to sea. More time spent in the estuary increases their potential for chemical exposure.

- Juvenile Chinook and chum salmon when in the estuary feed largely on invertebrates. These invertebrates are largely associated with sediments that can be chemically contaminated.

- Adults of all salmon pass through the study area on their way to Green River spawning grounds.

2.2 Resident Fish

English sole (*Parophrys vetulus*) was selected as a receptor because of its commercial and recreational importance and its significance to the aquatic community. The juvenile flatfish is probably the most sensitive lifestage found within the study area. While other species of flatfish (flathead sole, rock sole, and starry flounder) use the study area, English sole is numerically dominant in Elliott Bay and the Duwamish River (Miller et al. 1977). Therefore, English sole will be used as a representative flatfish in this study.

In general, juvenile English sole are found in shallow intertidal areas (to depths of approximately 18 meters) while adults inhabit deeper nearshore waters (Forrester 1969; Hart 1973; W. Paulson, Washington Department of Fish and Wildlife, personal communication). Early larval stages have not been observed in estuarine larval surveys; sometime after late-stage transformation, juveniles enter estuaries and settle (Rogers et al. 1988). Shi (1987) has speculated that juvenile English sole move into intertidal areas to feed during high tides. Day (1976) found that English sole were distributed in the Whidbey Island area at depths ranging from about 27 to 220 meters. In contrast, few fish were caught near Shilshole Bay or Carr Inlet at depths greater than about 100 meters (Holland 1969). In Puget Sound, English sole are typically encountered at depths of 25 to 50 meters (Smith 1936).

English sole frequently occur on soft sand or mud bottoms (Smith 1936). In these habitats, juvenile English sole (those less than 110 mm) eat annelids (Smith 1936), copepods, amphipods, and mollusks (Holland 1954). Adult English sole studied in British Columbia were found to eat clams, clam siphons, small mollusks, marine worms, small crabs, small shrimps, and brittle stars.

In south Puget Sound, adult populations of English sole concentrate in Elliott Bay and Port Gardner, but disperse after spawning, which usually is in winter (W. Paulson, Washington Department of Fish and Wildlife, personal communication). English sole migrate seasonally to their spawning grounds in Puget Sound in winter (Forrester 1969) with spawning typically occurring in Puget Sound during February and March (Smith 1936). Angell et al. (1975) reported off-season emigration in winter and spring of all age groups of central Puget Sound fish from Meadow Point to Carkeek Park (northwest side of Seattle) at depths of 3 to 30 meters. Early demersal juveniles (10 to 25 mm standard length), not all completely metamorphosed, migrated from spawning areas to nursery grounds to begin settling in December or May and June.

Aspects of English sole biology influencing their exposure to CSO stressors are:

- Older juveniles and adults are generally present in the study area during early winter, spring, summer, and fall; they may be absent January through March when many CSOs occur.
- They occur primarily on soft sand or muddy sediments, which can be chemically contaminated.
- They eat invertebrates that are largely associated with these sediments.
- Eggs and young juveniles are not likely found in the study area.
- Spawning and nursery grounds for English sole in Elliott Bay have not been documented.

2.3 Benthos

Bottom dwelling invertebrates (benthos) are recognized as sensitive indicators of chemical and physical impacts. Benthos that can be found attached to or residing near the sediment surface are known as epibenthos. Benthos that live in the sediments are referred to as infauna. Benthos can be sedentary or motile and can exist both intertidally and/or subtidally. The benthos can be further divided into macrofauna and meiofauna on the basis of size. Macrofauna are large enough to be retained on a 0.5-mm screen while meiofauna pass through the 0.5-mm screen but are retained on a 0.153-mm screen. Most benthos in the Duwamish River are small, very numerous, and are found in fine-grained sediments. Also, most are important food resources for commercially and recreationally important fish, and wildlife.

2.3.1 Intertidal Community

While there are several remnant intertidal habitats in the study area, the Kellogg Island intertidal habitat is by far the largest and the most studied (Leon 1980; Parametrix 1990, Cordell et al. 1996). It is a mudflat of about 17 acres that is dominated by detrital material and fine-grained sediment. Leon (1980) found 43 different kinds of benthos in sediment cores from the intertidal mudflats at Kellogg Island. Most organisms occurred infrequently and only nine kinds accounted for 97 percent of all individuals as shown in Table 2-2. These data indicate that the benthic community was dominated by *Manayunkia* (small marine worm), harpacticoid copepods (small crustacea), and oligochaetes (smaller worms related to earthworms).

Table 2-2. Abundances of Common Intertidal Organisms from Kellogg Island, Totals for All Samples, August and April. (Leon 1980)

	Total Number from All Cores	Percent of Total
Oligochaeta	887	21.6
Capitella	222	5.4
Manayunkia	1,768	43.1
Polydora	72	1.7
Pygospio	190	4.6
Harpacticoid	653	15.9
Cumacea	81	2.0
Anisogammarus	76	1.9
Corophium	49	1.2
All Other Organisms	107	2.6

While the data in Table 2-2 represent the abundances of the nine dominant organisms in both April and August samplings, Leon (1980) found that greater abundances occurred in August. By the following April, most species decreased by 50 percent reflecting a winter die-back. Parametrix (1990) used a different sampling method (plankton pump) at Kellogg Island and found 80 different kinds of invertebrates (epibenthos) inhabiting the intertidal community. Nematodes (small worm), oligochaetes, small harpacticoids, ostracods (small crustacea), and sabellid polychaetes (probably *Manayunkia*) were the dominant forms.

Cordell et al. (1996) found 27 kinds of macrofauna and 32 kinds of meiofauna in sediment cores from Kellogg Island collected in May 1995. The macrofauna community was numerically dominated by four groups: nematodes, oligochaetes, polychaete worms (*Manayunkia*), and the gammarid amphipod, *Corophium*. The meiofauna was dominated by nematodes and harpacticoids. These data suggest that the benthic community had changed very little since the studies by Leon (1980) and Parametrix (1990).

Many of the animals that live in the intertidal zone, particularly polychaete worms, such as *Capitella*, feed on particles of decaying plant and animal matter deposited on the sediments. The polychaete, *Manayunkia,* however, filters particles from the water column. Others organisms, such as copepods, feed on diatoms (single-celled plants), detritus, or their own larvae. Oligochaetes feed on bacteria, diatoms, and other microorganisms.

2.3.2 Subtidal Community

In 1980, the subtidal community was represented by more than 60 different kinds of organisms (Leon 1980). The distribution of individuals among species in van Veen grab samples was more even in the subtidal habitat than in the intertidal habitat as the 13 most abundant groups accounted for 90 percent of the total individual collected (Table 2-3).

Table 2-3. Abundances of Common Subtidal Organisms from Kellogg Island, Totals in All Grabs, August and April (Leon 1980) [a]

	Number	Percent of Total
Oligochaeta	676	9.3
Capitella	260	3.6
Heteromastus spp.	274	3.8
Cirratulus	1,841	25.3
Tharyx	1,112	15.3
Lumbrineridae	879	12.1
Pygospio	760	10.4
Axinopsida serricata	202	2.8
Macoma balthica	137	1.9
Macoma incongrua	69	1.0
Anisogammarus	75	1.0
Ostracoda	83	1.1
Psephida lordi	198	2.7
All Other Organisms	726	10.0

[a] These figures exclude *Manayunkia* found in an anomalous mass at a single station in August

While some of these animals were found intertidally (oligochaetes, *Capitella, Pygospio*, ostracods), most subtidal species were deposit (detrital) feeding polychaete worms which are characteristic of the deeper, turbid, waters of the Duwamish River. Small deposit-feeding clams (*Macoma, Axinopsida, and Psephidia*) and an amphipod (*Anisogammarus*) also appeared here. *Anisogammarus* feeds on diatoms and green algae.

In the Parametrix studies (1990), nematodes, oligochaetes, small harpacticoids, and cumaceans dominated the subtidal epibenthos. Differences in the results of the two studies may be due to the different sampling method employed in each case. The pumping method employed by Parametrix tended to collect more of the weakly swimming (planktonic) forms and also more of the organisms loosely attached to sediments.

While some of the stations studied by Leon (1980) were in the dredged navigational channel, the fauna at these sites did not show signs of any impact due to dredging. This was attributed to the infrequent schedule of dredging of about once every 10 years. Leon (1980) indicated that while dredging would be expected to eliminate all benthic life, recolonization would be rapid, less than a year for most species.

Benthic communities inhabiting sediments in the vicinity of CSOs can be subjected to both chemical and physical stress following discharge events. Chemicals tend to accumulate and persist in depositional areas downstream from CSOs and sedimentation could also smother shellfish and other benthos. Discharge events occurring in late winter or early spring could also expose the eggs and larvae of benthic species to chemical and mechanical stress. Most benthic species have a floating egg and weakly swimming larvae, which are found in the water column over a period of days to a few weeks in late winter and early spring. It is generally assumed that egg and larval lifestages are more sensitive to chemicals and other stimuli than the adult lifestage.

Aspects of benthos biology influencing their exposure to CSO stressors are:

- An abundant and diverse benthic community inhabits the study area.
- The benthos occurs on and in sediments that can be chemically contaminated or physically disturbed by CSO discharges.
- Eggs and larvae of benthic species occur in the water column in late winter and early spring at a time of increased CSO activity.
- The benthos are an important food resource for fish and wildlife inhabiting the lower Duwamish River.

2.4 Shore Birds, Wading Birds, Raptors, and Aquatic Mammals

Spotted sandpiper (*Actitis macularia*), great blue heron (*Ardea Herodias*), bald eagle (*Haliaeetus leucocephalus*), and river otter (*Lutra canadensis*) were also selected as receptors. All are present on the Duwamish River and Elliott Bay either seasonally or year-round and all have high societal value. Spotted sandpipers are protected by the Migratory Bird Treaty Act. The bald eagle has threatened status in Washington under provisions of the Endangered Species Act of 1973, as amended. The great blue heron is listed as a "priority species" by the Washington Department of Wildlife in 1991 (WDF 1991). It is assumed that each of these species is exposed through contamination of their food supply. Each species is also exposed through dermal contact with sediments but this is assumed to be a minor pathway.

2.4.1 Spotted Sandpiper (*Actitis macularia*)

Spotted sandpipers have been observed within the study area from late June through September (Cordell et al. 1996), but also are known to winter in protected embayments of Puget Sound (Paulson 1993). Over the period June through September 1995, Cordell et al. (1996) observed spotted sandpiper on 33, 69, 29, and 8 occasions at a reference location near the upper turning basin, a restoration site near the upper turning basin, Kellogg Island, and at Terminal 105, respectively.

Spotted sandpiper feed on invertebrates, e.g., amphipods, polychaetes, by probing and picking the intertidal sediments. Albright (1977) found spotted sandpiper and other shore birds feeding on the amphipod, *Corophium*, on Gray's Harbor mudflats. Leon (1980), Parametrix (1990), and Cordell et al. (1996) determined that *Corophium* was one of the most abundant amphipods on Kellogg Island mudflats.

It is not known if spotted sandpiper breed in the study area. If they do, their nesting sites will be on the ground in semi-open vegetation and close to water (Oring et al. 1997). Because of minimal disturbance and an abundant food supply, Kellogg Island is the most likely site. Egg laying in Minnesota begins in late May to early June. Females lay eggs one to six times each year and up to 12 eggs in total. Females are polyandrous meaning they lay eggs for different males. Males undertake most of the incubation and parental care. In Minnesota, densities of birds ranged between 4-13 females per hectare and 4-20 males per hectare over a 10-year period (Oring et al. 1983).

Aspects of spotted sandpiper biology influencing their exposure to CSO stressors are:

- Adult spotted sandpipers are present in the study area in summer.
- They forage over the intertidal zone probing and picking small invertebrates from the sediments, which may be chemically contaminated.
- Breeding in the study area may occur but has not been documented.

2.4.2 Great Blue Heron (*Ardea herodias*)

The great blue heron is a year-round resident of the study area and is a fish eater. They are often seen feeding in or near eelgrass in Elliott Bay but can be found in any intertidal habitat in the Duwamish River. Kellogg Island is a particularly important habitat for great blue heron. They were the most numerous of shore/wading birds recorded by Cordell et al. (1996) on the Duwamish River over the period of June to September 1995.

Heron colonies (rookeries) are usually located close to feeding areas. In the study area, a heron colony (rookery) is located in nearby West Seattle a few hundred meters from Kellogg Island. This site is used by up to 40 birds (Norman 1995). Another rookery in Renton, 12 km distant, contains 28 nests and also may contribute birds to the study area. In Minnesota lakes the distance between rookeries and feeding areas ranged between 0 and 4.2 km, averaging 1.8 km (Mathisen and Richards 1978). Parnell and Soots (1978) found that rookeries in North Carolina were located an average of 7 to 8 km from feeding grounds.

While three to seven eggs are laid over a period from early March to May, seldom more than two chicks fledge (Norman 1995). In late summer after fledging (leaving the nest), the juveniles disperse widely and do not return to their natal area until adulthood (Butler 1995). They exploit any small body of water where fish are abundant but tend to spend their winters in upland areas feeding on invertebrates and mice (*Microtis* sp.). Butler (1991) suggests that this is because they can't meet their energetic requirements in coastal estuaries in fall and winter. Birds that are observed within the study area, then, tend to be adults.

Shiner perch (*Cymatogaster aggregata*) is a major food source of the chick and female and may be related to juvenile survival (Butler 1993). Adult shiner perch are particularly abundant in the Duwamish River in May and June (Matsuda et al. 1968). Juvenile shiner perch are more abundant in the river other months of the year. Great blue herons eat fish up to 20-25 cm in length (Kirkpatrick 1940; Hoffman 1978). Adult herons provide the same food to their nestlings as they consume, although partially digested (Kushlan 1978).

Aspects of great blue heron biology influencing their exposure to CSO stressors:

- Adult great blue heron are present in the study area throughout the year; juveniles are present in the study area at least part of the year.
- They eat mostly resident fish species, e.g. shiner perch.
- They wade and fish in the shallows making contact with the sediments, which can be chemically contaminated.
- Breeding occurs in the study area.
- Young juveniles are fed the same prey that adult herons consume.

2.4.3 Bald Eagle (*Haliaeetus leucocephalus*)

The bald eagle is primarily a carrion feeder (dead and dying fish) but will also catch live fish (Brown and Amadon 1968). Spawned-out salmon are a particularly important food item in the Pacific Northwest. In eating carrion, they may ingest small amounts of sediment. Although eagles feed mainly on fish, waterfowl make up a significant portion of their food during winter months. Eagles have been observed to kill Western grebe in the Duwamish River in winter (J. Strand, Department of Natural Resources, King County, personal observation). Eagles also have been reported to prey on great blue heron chicks (Norman et al. 1989).

Resident birds are found in the study area in the summer but this may be limited to one or two pair. The closest active eagle nest is located in West Seattle, only a few hundred meters from water and in the study area (K. Stenberg, Department of Natural Resources, King County, personal communication). Migrant (wintering) birds are routinely observed in the study area beginning in October. They migrate north in late March.

Breeding pairs bond for life and when mature (four to five years) lay up to three eggs each spring. The young fledge in about 10 to 12 weeks with both sexes foraging to feed the young (Brown and Amadon 1968). With renovation, the same nest can be used year after year.

Aspects of bald eagle biology influencing their exposure to CSOs Stressors are:

- Adults are present in the study area throughout the year; juveniles are present in the study area at least part of the year.
- They eat fish (living and dead) and other wildlife from the study area. Dead fish and other carrion may be found on sediments, which can be contaminated.
- They breed in the study area.
- The young juveniles are fed the same food that the adults consume. Prey is captured close to the nest tree in the study area.

2.4.4 River Otter (*Lutra canadensis*)

From largely anecdotal information, a family of river otters lives year-round on Kellogg Island. The river otter was once harvested for its fur.

Local river otters feed largely on fish but also will feed on crabs and sometimes mussels and clams (J. Strand, King Department of Natural Resources, King County, personal observation). They are more likely to eat non-game than game species. In eating invertebrates they may ingest sediment and other material. Otters in captivity required 700-900 grams of food daily (Harris 1968). In Oregon, they have been reported to eat adult coho salmon during the period of salmon spawning (Toweill 1974). Waterfowl, gulls, and rails, particularly eggs and nestlings, comprise a significant part of their diet in pacific coast states (Toweill 1974; Grenfell 1974; Hayward et al. 1975; Verbeek and Morgan 1978).

Little is known about size of the otter's home range. It is likely dependent on habitat and the availability of food and dens. On rivers or streams, their home range may be a long strip along each shoreline. In a wetland or area with many small streams, the home range may resemble a polygon. In Sweden, the home range for a female and young was an area 7 km in diameter (Erlinge 1967). The home range for an adult male was 15 km in width with a highly variable length. Male otters also were found to forage 9-10 km a night and up to16 km have been recorded.

Mating occurs in early spring just after birth of a litter (Whitaker 1996). Uterine implantation is delayed for up to 10 months, and active gestation lasts for two months (Pearson and Enders 1944, cited in U.S. EPA 1993; Liers 1951,1958). A litter of one to six young are born in March or April (Hooper and Ostenson 1949). Adult female otters breed each year in Oregon, the closest area for which data are available (Tabor and Wright 1977).

Aspects of river otter biology influencing their exposure to CSO stressors are:

- Otters may spend their complete life cycle in the study area.
- They feed largely on resident fish, which can have higher burdens of tissue-deposited chemicals than migrant fish.
- They also eat invertebrates that are largely associated with the sediments that can be chemically contaminated.

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Issue Paper 3: HUMAN SITE USE

As part of the Duwamish River/Elliott Bay Water Quality Assessment (WQA), King County is assessing current and future risks to human health from using the Duwamish Estuary, as well as the fraction of these risks attributable to CSO discharges. The risks being assessed are those associated with pathogens and chemicals that may occur in CSO discharges, surface water, sediments, and aquatic organisms. This paper summarizes the human use of the Duwamish River and Elliott Bay and highlights uncertainties and issues regarding the level and extent of use. Additional discussion of the human site use is presented in the Appendix A - *Problem Formulation*.

The different activities in which that people are engaged in the estuary can be grouped into two general categories:

- Direct contact activities that may result in direct contact with the water and sediment of the Duwamish River and Elliott Bay. Examples of such activities include swimming and SCUBA diving, among others.

- Indirect contact activities that may result in exposures to chemicals and pathogens that have bioaccumulated in seafood. Examples of such activities include seafood consumption by recreational and subsistence fishers.

To gain a better understanding of these activities, we conducted a series of field and phone surveys to assess activities that may potentially result in exposures to chemicals and pathogens in the Duwamish River and Elliott Bay. Human activities in the Duwamish River and Elliott Bay, estimation of exposure based on human site use, and a summary of the site use are described further below.

3.1 Human Activities in the Duwamish River and Elliott Bay Estuary

Activities that may result in direct exposure with the water and/or sediment of the Duwamish River and Elliott Bay include:

- Swimming and wading
- SCUBA diving
- Boating and sailing
- Wind surfing
- Jet skiing

- Canoeing and kayaking
- Water skiing
- Parasailing
- Occupational exposures
- Line fishing and net fishing
- Collecting organisms other than fish (e.g., crabs, and squid).

In addition to these activities, the consumption of seafood collected from the study area may result in indirect exposures to the water and sediment of the study area.

Each of these human activities, along with uncertainties associated with characterizing human activities in the study area, are further described in the sections that follow.

3.1.1 Swimming and Wading

There are many access points along the shorelines of the Duwamish River and Elliott Bay at which people may swim and wade. The two primary access points are at Duwamish Park along the Duwamish River and at Duwamish Head in Elliott Bay. However, neither of these swimming locations is generally considered to be especially popular relative to other Puget Sound beaches in Seattle such as Alki Beach or Golden Gardens Beach.

We expect that swimming and wading in the river and bay is largely confined to the warmer summer months. This assumption is based on the generally cold water temperatures of the estuary. The average water temperature for the study area is about 9^{o}C (about 48^{o}F), with winter water temperatures dropping as low as 4^{o}C (about 39^{o}F) and summer water temperatures rising as high as about 16^{o}C (about 60^{o}F).

3.1.2 SCUBA Diving

We conducted a phone interview of the long-time manager of a dive shop at Seacrest Park in West Seattle to obtain anecdotal information on the number of people that scuba dive in Elliott Bay and the Duwamish River, and the frequency that they dive (Ferdico 1997). This anecdotal information indicates that the majority of the people that dive in Elliott Bay or the Duwamish River dive at the Seacrest Park in West Seattle. It was estimated that between 0 and 100 people dive at Seacrest Park on any given day, with higher numbers of divers during the summer than winter. More people also dive on weekends than on weekdays. For comparison, the number of divers at Alki Beach (outside the study area) was estimated to range between 0-20 divers per day dependent on day of the week and season. The dive shop manager suggested an average diving frequency in the study area of once per month.

3.1.3 Boating/Sailing

There is heavy boat use in Elliott Bay and the Duwamish River. Boat size varies from small, one-person sailboats to large supertankers that dock at Harbor Island. It is possible that people on boats would come into contact with the surface water or sediments of the bay or river. Such contact could occur while pulling in anchors, from wind/wave spray, or from pulling in ropes and bumpers.

We do not know how many people go boating in the Duwamish River or Elliott Bay. However, the degree of exposure by boaters is expected to be lower than experienced by people windsurfing.

3.1.4 Windsurfing/Jetskiing/Canoeing/Kayaking/Parasailing/Water skiing

We do not expect many, if any, people windsurf, jet ski or parasail in the Duwamish River. However, we do expect that people conduct these activities in Elliott Bay. We also expect canoeing and kayaking to occur in both the Duwamish River and Elliott Bay. However, we do not know how many people engage in these activities, or how frequently they do so.

3.1.5 Occupational Exposures

Occupational exposure can result for people who work in and around the study area, as they may be exposed to chemicals and pathogens in the waters and sediments of the Duwamish Estuary. For example, construction, repair and maintenance workers may be exposed if they are working on underwater pilings. Persons monitoring or sampling pilings or other underwater objects may also be exposed. However, we do not know the number of people that may be exposed through these activities or the frequency with which they engage in the activities.

3.1.6 Recreational Seafood Collection

We conducted a fishing survey during the summer of 1997 to determine how many people collected what types of organisms from the shores of the Duwamish River and Elliott Bay at what frequency, and their plans for their catch. We observed approximately 1,183 different people collecting organisms during 30 days in June, July and August. We did not interview people collecting from boats, or while scuba diving. Nor did we interview people collecting using gill nets.

The vast majority of the people interviewed during the survey collected seafood at one of three locations: Seacrest Park, Elliott Bay Fishing Pier in Myrtle Edwards Park, or from the Harbor Island Bridge over the East Waterway of the Duwamish river. A majority of the people collects seafood during the summer only, although about 10 percent collect seafood every month of the year. About 53 percent of the people collect seafood less than

12 times per year, about 29 percent collect between 12 and 52 times per year, and 18 percent collect more than 52 times per year.

Salmon accounted for the majority of the mass of the seafood collected. A total of 34 salmon were collected, weighing approximately 359 pounds, out of a total of 565 pounds of collected seafood. Other seafood collected included a variety of fishes, crabs, squid and clams. The salmon collection is seasonal and corresponds to the seasonal salmon runs. Nearly all people interviewed were line fishing, or collecting crabs using crab pots.

A survey of boating anglers in Elliott Bay and Commencement Bay conducted in 1985 (NOAA 1987) found that Elliot Bay received heavy fishing pressure from boating anglers at that time. We believe that the number of people boat fishing in Elliott Bay has likely declined since the 1985 study because of restrictions associated with salmon fishing from boats that have been implemented during recent years. For example, the salmon fishery was closed to boat fishing during the period that we conducted our fishing survey. Therefore, the applicability of this 10-year-old information on collection from boats is unknown.

3.1.7 Commercial Seafood Collection

Salmon are commercially harvested from the Duwamish River and Elliott Bay. According to the Washington State Department of Fisheries (1993), approximately 30,000 salmon were harvested in this area during 1992. This harvest included chinook, chum, coho, sockeye, steelhead and Atlantic salmon. The majority of these fishes are harvested by the Muckelshoot and Suquamish Tribes. Commercial harvesting of shellfish is currently not allowed in Elliott Bay due to the occurrence of bacteria in water and shellfish (King County 1995; WSDOH 1996).

Members of the Muckelshoot and Suquamish Tribes have treaty rights to collect salmon heading up the Duwamish River. Gill netting is the preferred method of salmon collection for tribal members. We currently do not know how many people collect fish using this method, or how many times each year they do so.

To develop data on the frequency of other commercial harvests from the study area, a phone survey of commercial fishing companies, local seafood wholesalers and distributors was conducted to determine how many companies harvest seafood from the Duwamish River or Elliott Bay. At the time of this writing, twenty-two companies had been contacted. None obtained any seafood from Elliott Bay.

In an effort to determine how many charter fishing operations might be harvesting from Elliott Bay or the Duwamish River, we conducted a phone survey of charter fishing companies regarding their use of these areas. Of the twelve local charter companies contacted, three indicated any use of the area. The use of the Elliott Bay was indicated to be rare, and two companies indicated that they had not kept the fish collected from Elliott Bay. Although this survey is not a comprehensive or conclusive survey of charter fishing in

Elliott Bay, it suggests that charter fishing in these areas occur infrequently with only limited collection.

3.1.8 Recreational Seafood Consumption

The results of King County's fishing survey indicate that not everybody that collects organisms from the Duwamish River and Elliott Bay consume their catch. Some people release what they collect, others feed their catch to animals, and some use it as bait to try to catch more desirable species. The results of our survey indicate that less than one-half of the people we interviewed eat seafood that they collected from the location at which the survey was conducted.

Those people that indicated that they consumed their catch did so at various frequencies, ranging from once per year to every day of the year. Of those that consume the seafood they collect, about 50 percent do so eight or fewer times per year and about 75 percent do so 24 or fewer times per year. However, seven people were located that collect and consume seafood from the study area every day of the year. These results indicate that the majority of people consuming seafood from the study area do so less than 24 times per year. However, the survey also identified a small population of subsistence fishers (seven people consume study area seafood on a daily basis) that obtain the majority of their protein from seafood collected in the study area.

3.2 Uncertainties in Characterizing Human Site Use

We believe that we have a fairly good understanding of the types of activities that people engage in the estuary. This belief is founded on direct observations of people using the estuary for a variety of activities. However, several uncertainties are associated with estimating the level of use of the Duwamish River and Elliott Bay. We attempted to reduce some of the largest of these uncertainties by conducting a fishing survey of recreational fishers, and a phone survey of commercial fishers. The seafood collection survey provides recent, site-specific information that can be used to derive estimates of exposure parameters for use in exposure estimation models. However, some uncertainties remain regarding seafood collection and consumption, including:

- The survey was conducted over a limited time period, and as such, only represents a snap shot in time of the fishing pressures on the system. We believe that the survey results provide a reasonable estimate of the fishing pressure during the time period that the survey occurred. We also believe that the 50^{th} percentile, 75^{th} percentile and maximum seafood consumption frequencies estimated for the survey period probably would not substantially change were the survey continued for an entire year. However, it is likely that our survey underestimates the number of people that collect and consume seafood from the study area each year.

- While we asked the ethnic background of each person interviewed, it is not possible to characterize behaviors within entire ethnic subpopulations. For example, although our survey results indicate that we interviewed 69 individuals of Vietnamese descent, it is not possible to characterize the seafood consumption habits of the entire Vietnamese community.

- Seafood collection and consumption may vary based on factors not accounted for in this risk assessment. For example, boat fishing for salmon has been tightly regulated in recent years as a method of regulating salmon populations. It is possible that similar changes may occur in the future and alter the frequency that people collect and consume seafood from the study area.

3.3 Use of Site Use Data to Estimate Human Exposures in the Risk Assessment

As part of the risk assessment, we need to estimate human exposures to chemicals and pathogens in the sediment, water and seafood of the Duwamish River and Elliott Bay. These estimates of exposure are based on a combination of the frequency and duration of exposure activities, along with estimates of the concentrations of the chemicals and pathogens in the water, sediment and seafood. Exposure is estimated separately for direct exposures to water and sediment, and indirect exposures through the consumption of seafood. For both types of exposures, we will assess a range (low, medium, high) of human exposures in the risk assessment based on our understanding of the human site use of the Duwamish River and Elliott Bay. We will estimate the range of exposures for both children and adults. Methods for estimating direct and indirect exposures are described below.

3.3.1 Methods for Estimating Direct Exposures

We have limited information regarding the amount of direct exposures that occurs from these activities, other than an average of once per month for scuba diving. We also know that members of the Muckelshoot Tribe may be highly exposed to water and sediment while pulling in gill nets during their salmon collection activities. However, we do not know how frequently this activity occurs and how many people are involved in this activity, nor do we know how many people may engage in this activity in the future.

Several of the identified exposure activities are not being explicitly quantified because we have such limited data regarding these activities and because exposures resulting from these activities are expected to be similar or less than comparable exposure pathways selected for evaluation. Direct exposures to water and sediment were assessed for (1) swimming, (2) SCUBA diving, (3) windsurfing, and (4) net fishing. Direct exposures via sailing, boating, kayaking, parasailing, water skiing and jet skiing were not evaluated because exposures from these activities are expected to be similar or less than exposures occurring while wind surfing. Similarly, direct exposures while wading were not assessed because exposures while swimming are expected to be larger and provide a more conservative estimate of exposure. Finally, direct exposures while line fishing and gathering shellfish and other organisms were not assessed because these exposures are expected to be smaller than those experienced while net fishing.

To account for variability in human behavior, we will assess low, medium and high levels of exposures to encompass all of these activities (Table 3-1). We believe that fewer people will engage in these activities at the high exposure frequency than at the low or medium exposure frequencies. Children will be assessed separately for the swimming exposure pathway. Children will not be assessed for SCUBA diving, windsurfing or net fishing because it is believed that few children engage in these activities.

Table 3-1. Exposure Frequencies to be used for Estimating Direct Exposures to Water and Sediment in Human Health Risk Assessment

	Exposure Frequency (days/year)		
Activity	**Low**	**Medium**	**High**
Swimming	2	12	24
SCUBA diving	2	12	24
Windsurfing	2	12	24
Net fishing	2	24	91

3.3.2 Methods for Estimating Indirect Exposures

Low, medium and high seafood consumption frequencies will be used to assess exposures and risks from seafood consumption. Based on the results of our survey, we will use consumption frequencies of 8, 24, and 365 meals per year to represent low, medium and high consumption frequencies, respectively. These consumption frequencies are equal to the 50th percentile, 75th percentile and maximum consumption frequencies of the people included in King County's fishing survey. Evaluation of daily (365 meals per year) seafood consumption ensures that we will not underestimate exposures and risks to subsistence fishers. We also will estimate exposures and risks per meal.

Exposures will be estimated for each seafood species for which King County has concentration data. This approach will allow for an estimate of upper and lower bounds of exposure based on different combinations of seafood consumed. Estimation of exposures and risks per meal will allow the reader to estimate exposures based on individual seafood consumption habits and overcomes uncertainties associated with different people consuming different combinations of seafood. Children and adults will be evaluated separately for seafood consumption because the fishing survey clearly indicates that children consume seafood from the study area.

3.4 Summary

The Duwamish River and Elliott Bay are obviously heavily used by a large number of people engaging in a variety of activities. Substantial site-specific information is available to characterize the extent to which some of these activities occur, while limited information is available concerning the extent other activities occur. The extent to which these can be quantified varies, as does the uncertainties associated with the differing sources of information used. We will assess these exposures as follows:

- Direct exposures to chemicals and pathogens in sediment and water will be assessed for swimmers, SCUBA divers, windsurfers and net fishers. Other activities known to occur in the Duwamish Estuary will not be quantitatively assessed because their level of exposure is expected to be similar or less than the exposures for the four selected activities. Low, medium and high exposure frequencies will be used to account for the variability in human behaviors. Swimming exposures by children will be assessed separately.

- Indirect exposures to chemicals and pathogens accumulated in seafood will be assessed for each of the different types of seafood with concentration data available. Per meal, low, medium and high exposure frequencies will be used to account for the variability in human behaviors. This approach also will allow the reader to estimate exposures based on different combinations of seafood consumed. Seafood consumption exposures by children will be assessed separately.

3.5 References

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Issue Paper 4: PATHOGENS

As part of the Duwamish River/Elliott Bay Water Quality Assessment (WQA), King County is assessing current and future risks to human health from using the Duwamish Estuary, as well as the portion of these risks attributable to CSO discharges. The human health risks being assessed are those associated with pathogens and chemicals that may be in CSO discharges, surface water, sediments, surface water runoff (possibly containing animal wastes), and edible aquatic organisms. The purpose of this paper is to describe the issues associated with assessing the risk of infectious disease in people as a result of using the Duwamish Estuary for recreational and commercial activities. These issues apply whenever there is a need to predict the likelihood of illness occurring as a result of environmental exposure. This paper will describe the challenges for this particular project and the approach that will be taken to assess the risks of infectious disease resulting from swimming, wind surfing, SCUBA diving (and other direct exposure activities), and/ or eating fish or shellfish from the study area.

4.1 Statement of Issues

4.1.1 What Are Pathogens and Why Are They of Potential Concern?

Pathogens are microscopic organisms that may infect a person and result in disease. Illnesses with symptoms such as nausea, diarrhea, chills, fever, stomach and intestinal cramps, and jaundice are caused by bacteria, viruses, and parasites. People who are sick and some who are not sick but just infected, have high numbers of pathogens in their bodies. The numbers and types of pathogens as well as the area of the body infected vary with the illness.

In this particular study we are concerned about the disease causing organisms that are excreted from the body. These bacteria, viruses, or parasites leave the ill person and enter the wastewater treatment system. At the wastewater treatment plants, the pathogen-containing wastewater is chemically disinfected and the solids are anaerobically digested. For combined sewer overflows, the wastewater is not treated and therefore the pathogens may be carried in discharges directly into the Duwamish River or Elliott Bay. Relatively rapid die-off of most pathogens is expected in the river and bay, although some may survive for weeks or months. These organisms may also be contained in animal wastes as intermediate hosts and may be present in surface water runoff, surface waters, and sediments.

It is possible that the presence of these pathogens in the Duwamish River or Elliott Bay may result in infection and disease in exposed populations. Infection and disease may result when people are exposed to sufficient numbers of viable pathogens.

4.1.2 What are the Pathogens Potentially Present in CSO?

The pathogens present in CSOs vary with the types of illnesses present in the population of people who use the King County DNR Wastewater Treatment services. Four types of organisms are generally associated with potential human health risks from pathogens in wastewater (Mara and Cairncross 1989; NRC 1996):

- bacteria,
- viruses,
- helminths, and
- protozoa.

Each of these is further described below. Other pathogens that naturally occur in the study area and are not associated with sewage discharges are also described, although estimation of human health risks from exposure to these pathogens is beyond the scope of this project.

Bacteria. Bacterial pathogens are single-celled organisms that reproduce inside infected people. Some bacterial pathogens can also reproduce in other host organisms (e.g., poultry, cattle, rodents), or in the environment (e.g., surface water, soils). Bacteria will survive for various lengths of time outside of an infected host. In addition, surviving bacteria may be injured to the point of complete lack of infectivity. Bacterial infections can range in severity from minor to lethal and are typically treated with antibiotics.

A wide range of bacteria potentially occurs in wastewater and CSO discharges (Mara and Cairncross 1989; NRC 1996; U.S. EPA 1992; WHO 1989). Some common examples of these bacteria include:

- *Salmonellae* sp.
- *Yersinia enterocolitica*
- *Shigellae* sp.
- *Escherichia coli*
- *Clostridrium perfringens*
- *Staphylococcus* sp.
- *Enterococcus* sp.
- *Campylobacter jejuni*

These bacteria occur to various degrees in King County. For example, there are approximately 1800 different types of salmonella most of which cause nausea, diarrhea, and fever. The most virulent form of this genus is that which causes typhoid fever, *Salmonella typhi*. While *S. typhi* is still present in this country, it is rare in King County. Other forms of Salmonella are consistently present in the King County population and accordingly present at all times of the year in the wastewater and combined sewer overflows.

Viruses. Viruses are small infectious agents that do not have true cell walls. They infect their hosts by inserting their DNA into host cells, instructing the infected cells to make copies of the virus agent. As with bacteria, viruses will survive for various lengths of time outside of an infected host. For example, small round viruses may survive for months in marine waters, while HIV virus will not survive outside of the human body except under unusual circumstances for very short time periods.

More than 100 enteric (intestinal) viruses can be found in human feces (Ahmed 1991), and hence may potentially occur in wastewater. Some common examples of the types of viruses that may be in wastewater worldwide include:

- Norwalk/Snow Mountain/small round viruses (SRVs)
- Enteric non-A, non-B hepatitis
- Hepatitis A
- Rotavirus
- Poliovirus
- Other picornaviruses

Specific viruses have been found or are thought to be present to various degrees in King County wastewater. For example, because rotavirus is the most common waterborne illness in the United States (Gerba et al. 1996), it is expected that rotaviruses are relatively common in King County and hence may be present in wastewater. On the contrary, poliovirus (other than the vaccine strain) is not expected to be common in King County because few cases of polio are reported annually in the United States.

Helminths. Helminths, generally known as worms (e.g., tapeworms, roundworms), invade their hosts in the larval stage where they migrate through the body before maturing in the gut. Helminths may cause serious tissue and organ damage, and adult forms generally cause malnutrition and anemia while residing in the gut. Helminth ova may cause infections either from direct infection without intermediate hosts, or by passing through intermediate hosts, such as fish. Removal of worms prior to cooking and thorough cooking of infected fish prevents infection by worms.

Many different helminths may be present in fish and shellfish. However, the majority are harmless to humans and are not sewage related (Ahmed 1991). Illnesses from helminths are generally associated with eating raw fish. Helminths are responsible for far fewer human illnesses than either bacteria or viruses (Ahmed 1991). Sewage-related helminths are not typically problems in developed countries (Mara and Cairncross 1989) and are uncommon in King County.

Protozoa. Protozoan pathogens are single-celled organisms that cause a variety of symptoms by colonizing the gastrointestinal tract. Protozoan diseases may be debilitating but are rarely fatal in developed countries like the United States. However, the dehydration and nutritional imbalances caused by protozoan-related diarrhea (e.g., from *Giardia*, *Cryptosporidium* and related organisms) can be fatal to infants, the elderly, and people with compromised immune systems such as those with AIDS or who are undergoing radiation treatments, chemotherapy, or organ transplants. Protozoa may be present in wastewater, treated wastewater, CSO discharges, sludge and even in untreated or inadequately treated drinking water as cysts and oocysts, which are dormant structures resistant to adverse environmental conditions and disinfection. Enteric protozoa, including *Giardia* spp. and *Cryptosporidium* spp., are of particular importance and can cause moderate to severe enteritis. Most people who become infected with *Giardia* do so by drinking untreated contaminated surface water from lakes or rivers. Because *Giardia* and *Cryptosporidium* occur in King County, it is expected that their cysts and oocysts will occur in untreated wastewater.

Other Naturally Occurring Pathogens. The bacteria *Vibrio parahaemolyticus* occurs naturally in Puget Sound. While it is not associated with sewage discharges, it can accumulate in shellfish to pathogenic levels during the warmer summer months (WSDOH 1997). Vibriosis is typically characterized by diarrhea, abdominal cramps, nausea, vomiting, headache, fever and chills. Because *V. parahaemolyticus* is not related to sewage discharges, risks from this bacteria are beyond the scope of this assessment.

Several other naturally occurring parasites commonly infect fish in the Puget Sound region. Most of these parasites remain on fish skin although some bore into the flesh of the fish. The parasite argulus, or fish lice, is relatively common on salmon in Puget Sound. These parasites do not appear to be related to sewage discharges and generally are not infectious to humans.

Finally, a wide variety of helminths not associated with sewage are known to naturally occur in fishes in Puget Sound. The most commonly occurring worm is the nematode *Philometra americana*. *Philometra* is very common in bottom fish (e.g., sole, flounder). As with the majority of the worms infecting fishes in Puget Sound, *Philometra* is not infectious to humans. The nematode *Anisakis simplex*, although not common in Puget Sound, is one example of an aquatic worm that can be transmitted to humans. Again, because these helminths are not associated with sewage discharge, the risks of their causing infection are beyond the scope of this assessment.

4.1.3 What are Other Possible Sources of Pathogens in the Study Area?

Human pathogenic bacteria, viruses, and parasites may enter the study area through a variety of sources, including CSO discharges, stormwater runoff, failed septic systems, and dumping of untreated sewage by boaters. In addition to these sources, the indicator organisms fecal and total coliforms may also originate from agricultural runoff and the feces of wild and domestic animals. Indeed, monitoring of fecal coliform concentrations in the Green River at the Tukwila gaging station, upstream of all of the CSO discharges, indicates that between 1987 and 1996, the fecal coliform concentration exceeded 130 colonies per 100 milliliters 50 percent of the time and exceeded 425 colonies/100 mL about 10 percent of the time.

4.1.4 What are Potential Routes of Exposure to Pathogens?

Exposure to sewage-related pathogens in estuarine and marine environments generally occurs through two activities:

- Consumption of raw or partially cooked shellfish.
- Incidental ingestion of contaminated water while swimming, SCUBA diving or during other recreational or commercial activities resulting in direct contact with contaminated water.

Each of these exposure pathways is discussed below.

Shellfish Consumption. Many studies have linked consumption of raw or partially cooked shellfish with viral infections (e.g., McDonnell et al 1997, Luthi et al. 1996, Le Guyader et al. 1996, Anonymous 1997). There is also some indication that eating cooked shellfish may not substantially reduce the risk of viral infection relative to eating raw shellfish (McDonnell et al. 1997). In the United States, hepatitis A was the predominant disease reported in the 1960s, but today acute gastroenteritis is most prevalent (Le Guyader et al. 1996). Many of the cases of gastroenteritis are caused by small round-structured viruses such as Norwalk and Norwalk-like viruses (Ahmed 1992, Le Guyader et al. 1996).

In the Puget Sound region, illnesses from eating shellfish may also be caused by the naturally occurring bacteria *Vibrio parahaemolyticus*. During 1997, 57 cases of vibriosis infection were reported in Washington State, the majority of which were associated with consumption of raw oysters (Therien 1998).

Disease caused by sewage-related bacteria in shellfish is less commonly reported than disease caused by sewage-related viruses in shellfish (Ahmed 1991). Sewage-related bacteria that have been proven as pathogens in seafood include *Salmonella typhi*, *Campylobacter jejuni, Escherichia coli, Yersinia enterocolitica, Clostridium botulinum*, and *Shigellae* (Ahmed 1991).

The results of the 1997 fish consumption survey indicate that some people eat shellfish from the Duwamish River and Elliott Bay. The survey observed 1 person that had collected clams, 42 people that had collected crab, 11 people that had collected shrimp, one person that had collected moon snails, and 2 people that had collected squid. A slightly larger number of people had collected fish than shellfish.

Data from the 1997 fish consumption survey also indicate that few people eat raw seafood collected from the Duwamish River or Elliott Bay. Of 105 people that provided information on preparation method in the fish consumption survey, one person indicated that they planned on eating their seafood raw. However, it is possible that people may eat raw seafood at other times of the year, or that people that eat raw seafood refused to participate in the survey.

Direct Contact with Water. We expect that people come into direct contact with Duwamish River and Elliott Bay water through a variety of activities. These activities may include swimming, wading, SCUBA diving, windsurfing, jet skiing, parasailing, sailing, boating, kayaking, line fishing and net fishing, among others. Some of these activities occur near the shores (e.g., swimming and wading), while others may occur in the middle of the bay (e.g., windsurfing). A more complete review of the human site use of the study was provided in Issue Paper 3.

People engaging in recreational water sports (such as swimming or SCUBA diving) in sewage-contaminated waters also may potentially be exposed to sewage-related pathogenic bacteria (Cabelli et al. 1982). Swimming in even marginally polluted marine bathing water has been demonstrated to be a significant route of transmission for gastroenteritis (Cabelli et al. 1982). Several studies have also been conducted that relate swimming in sewage-polluted waters with viral infections (e.g., Cabelli et al. 1982, Seyfried et al 1985). Rotaviruses and Norwalk and Norwalk-like viruses have been linked to these illnesses.

4.1.5 What are the benefits and limitations of using indicator organisms to assess pathogen contamination?

Monitoring for pathogen contamination usually encompasses using surrogate or indicator species, such as fecal coliforms, fecal streptococcus, enterococcus, or *E. coli*. Fecal coliforms, fecal streptococcus, enterococcus, and *E. coli* are present in wastewater and CSO discharges as a natural component of human feces. Indicator organisms are commonly used to assess potential exposures and risks from sewage-related pathogens in surface water and shellfish. Fecal coliforms, fecal streptococcus, enterococcus, and *E. coli* were first used because they are present in sewage in large quantities and are relatively easy to measure. Measurement of other, specific pathogens in water and tissue are much more difficult and expensive than measurement of indicator organisms. For example, to measure virus concentrations in surface water, it is commonly necessary to filter 50 to 100 gallons of water to observe one virus particle.

While indicator organisms are widely used to assess potential pathogen contamination, several studies have indicated that indicator organism concentrations in water do not accurately describe pathogenic risks from eating raw shellfish. For example, fecal coliform bacteria concentrations in surface water have been demonstrated to inaccurately predict risks of viral infection from consumption of raw shellfish (McDonnell et al 1997, Luthi et al. 1996, Le Guyader et al. 1996, Anonymous 1997). Indicator organisms for evaluating fecal contamination also are not applicable to non-sewage-related pathogens, such as the naturally occurring bacteria *V. parahaemoliticus*.

Several indicator bacteria have been proposed for assessing potential risks of gastrointestinal symptoms with swimming water quality, including fecal coliforms, *E. coli*, fecal streptococcus, enterococcus, and *Pseudomonas aeruginosa* (Cabelli et al. 1982, Seyfried et al 1985, Ferguson et al. 1996). Both total staphylococci and enterococci have been suggested as being better indicators of risk posed by sewage-contaminated bathing waters than total coliforms or fecal coliforms (Cabelli et al. 1982, Seyfried et al. 1985).

4.1.6 What are Pathogen Survival and Fate in Marine and Estuarine Systems?

Many factors influence the survival and viability of pathogens in aquatic environments. Specific pathogens may have widely different survival times in the marine environment, and may be accumulated by shellfish to various degrees (Ahmed 1991, Pitman 1995). In general, enteric viruses are more resistant than enteric bacteria to common sewage treatment processes, including chlorination as commonly practiced (Gerba 1988). Many viruses are also more persistent than bacteria in marine waters and are accumulated to a larger extent by shellfish (Gerba 1988, Chung and Sobsey 1993, Bosch et al. 1995).

Many pathogens tend to survive longer in cold water relative to warm water. For example, concentrations of the virus infectious hepatitis A remained stable for over 92 days in cold (4^{o}C) seawater, but decreased by 90 percent in 11 days at 25^{o}C (Crance et al. 1998). Similarly, many bacteria in surface water are injured or killed at higher water temperatures relative to lower temperatures (McFeters and Singh 1991). Other factors that influence survival and infectivity of pathogens in marine environments include pH, salinity, metals, u.v. radiation, nutrient restrictions, and possible presence of disinfectants (McFeters and Singh 1991).

4.1.7 What Are the Pathogen Regulations?

The United States Food and Drug Administration and the Washington Administrative Code require that approved shellfish collection areas meet water quality standards for either total or fecal coliforms. The sanitary quality of shellfish is based on an allowable standard (geometric mean) of 14 most probable number (MPN) fecal coliforms per 100 milliliters (mL) of growing water, with not more than 10 percent of samples exceeding 43 MPN fecal coliforms per 100 mL. The U.S. EPA also has set a geometric mean fecal coliform limit for bathing (swimming) waters of 200 colonies/100 mL.

Washington State has classified the Duwamish River as a class B water (good), with a fecal coliform criterion of 200 colonies/100 mL (geometric mean) and 400 colonies/100 mL (90^{th} percentile). Washington State has classified Elliott Bay east of a line between Pier 91 and Duwamish Head (approximately equal to the western boundary of the study area) as class A water (excellent). The fecal coliform criteria for marine class A waters are 14 colonies/100 mL (geometric mean) and 43 colonies/100 mL (90^{th} percentile).

No water quality requirements are currently in place for specific bacteria, viruses, or parasites.

4.1.8 What Are the Methods for Assessing Pathogen Risks?

Evaluation of specific pathogens to assess the risk of infection is a more complex method than using indicator organisms, and requires larger amounts of data. The approach used to assess risks of infection from specific pathogens is similar to that employed for chemical risk assessment. First, human exposure to specific pathogens is estimated. Exposure is likely to be expressed as the number of each type of organism ingested in one day. Second, the infectious dose, or the minimum infectious dose (MID), is determined for each pathogen. This represents the number of organisms that are required to be ingested for infection to occur to a defined percentage of normal, healthy adults. For example, about 30 *Cryptosporidium* oocysts will cause infection in about 20 percent of exposed, healthy adults, while 1,000 Cryptosporidium oocysts will cause infection in 100 percent of exposed healthy adults (DuPont et al. 1995). Finally, the exposure and infectious dose are compared to assess the potential for infection, and if possible, the likelihood that infection will result in disease.

Several viral agents, such as Norwalk and rotavirus virus, may theoretically cause infection and illness to nearly 100 percent of the exposed population through exposure to one viral particle (Ahmed 1991). Many other viruses have infectious doses of less than 100 viral particles (Ahmed 1991). Bacteria generally have larger infectious doses than viruses (Ahmed 1991, Gale 1996). For example, Shigellae are among the most virulent bacteria, yet the infectious dose is about 100 CFUs (colony forming units), or about 100 times larger than the infectious dose for Norwalk virus (Ahmed 1991). About 132 oocysts of *Cryptosporidium* are estimated to be needed to infect about 50 percent of a population of healthy adults.

Assessment of risks from specific pathogens in marine waters and shellfish is not common, but is possible to conduct. However, because pathogen concentrations in marine waters are probably too low to be measured, modeling techniques would be required using raw sewage pathogen concentrations, dilution, and survival. For example, Rose and Sobsey (1993) estimated that the national average risk of contracting viral gastroenteritis from eating raw shellfish from approved waters in the United States ranges from approximately 1 in 100 to 50 in 100 (1 in 2) per year. These estimates were calculated based on average concentrations of total viruses in shellfish from approved waters, estimates of the shellfish consumption rates in the United States, and estimates of infectiousness for different viruses.

4.2 Pathogen Data Availability

4.2.1 Indicator Organisms

Fecal coliform concentrations were measured in CSO discharges, Duwamish River and Elliott Bay surface water, and wastewater influent samples to the West Point sewage treatment plant. Pathogen concentrations in sewage treatment plant influent were measured as a worst-case estimate of possible concentrations in CSO discharges. These data were used to model fecal coliform concentrations in surface water using King County's water and sediment quality model for the Duwamish River and Elliott Bay. The modeled concentrations account for inputs from upstream in the Green River, from Puget Sound outside of Elliott Bay, from CSO discharges, and from other sources (e.g., stormwater discharges).

Fecal coliform bacteria concentrations were also measured in mussel tissues near the Brandon Street CSO outfall before, immediately after, and 24 hours after a CSO event.

4.3 Pathogenic Bacteria, Viruses and Parasites

Preliminary concentration data in untreated wastewater were obtained for Yersinia, Salmonellae, Listeria, total viruses, *Giardia* and *Cryptosporidium*. These concentration data were obtained to test the measurement capability of the laboratory for these organisms.

Concentrations of Yersinia, Salmonellae, and total virus, along with concentrations of *Giardia* and *Cryptosporidium* cysts and oocysts, were measured in seven wastewater influent samples to the West Point sewage treatment plant. Concentrations of Yersinia, Salmonellae and total viruses were also measured in mussels at the Brandon Street CSO outfall. These samples were collected before, immediately after, and 24 hours after a CSO discharge.

4.4 Proposed Method for Assessing Human Health Risks from Pathogens

We propose to assess potential risks from pathogens using two methods. The first method for assessing risks consists of a comparison of modeled fecal coliform concentrations in the Duwamish River and Elliott Bay to concentrations deemed safe for shellfish growing and to concentrations deemed safe for swimming. These comparisons will be conducted under baseline conditions with CSO discharges, as well as under the modeled scenarios assuming 100 percent CSO control. It is acknowledged that these comparisons serve as an indicator of potential risks, not as definitive risk estimates.

We are modeling concentrations of fecal coliforms in the Duwamish River and Elliott Bay because we do not currently have sufficient data for any other indicator organisms that could be used as a measure of exposure to disease causing organisms.. These modeled concentrations are calculated based on the amount of input to the system, fecal coliform die-off within the study area, survival in a marine environment, and removal from the study area.

The second method for assessing risks consists of calculating the risks of infection from viruses and *Giardia* originating from CSO discharges. Risks of infection will be estimated based on estimates of exposure from incidental ingestion of surface water, and estimates of infectivity of a virus (rotavirus) and a parasite (*Giardia*). Concentrations of viruses and *Giardia* in CSO discharges will be estimated from available site-specific and national data on their concentrations in untreated wastewater. King County's hydrodynamic fate and transport model of the Duwamish Estuary will then be used to estimate virus and *Giardia* concentrations in the river and bay resulting from these discharges. Because no other site-specific data are available on virus and *Giardia* concentrations in the river and bay, the modeled concentrations will represent those associated with the CSO discharges only, not those associated with all sources. Estimated exposures will be combined with estimates of infectivity to derive risk estimates (Regli et al. 1991; Haas et al. 1993; Rose and Gerba 1990; Rose et al. 1995). Risks of infection from viruses and *Giardia* in shellfish will not be quantitatively assessed due to a lack of data on their ability to bioconcentrate from surface waters.

To assist us with our assessment of pathogen risks, we have hired Dr. Joan Rose, an expert on microbiological risk assessment. Dr. Rose will advise us on the best method for assessing pathogen risks under baseline conditions, as well as the portion of the risks attributable to CSO discharges. Dr. Rose will visit the project team in June 1998 and will prepare a report on pathogenic risks based on available data.

4.5 Summary

- Pathogens in CSO discharges may result in the increased likelihood of exposure and illness in people who come into direct contact with contaminated water and sediment or who eat shellfish.

- Pathogen regulations are based on concentrations of indicator organisms in surface water, not concentrations of specific pathogens of concern.

- Exceedance of indicator organism concentrations deemed safe by regulation may not accurately correlate to risks of infection.

- Indicator organisms, as well as specific pathogens, in the study area may originate from many sources, including many non-sewage-related sources.

- Specific pathogens of interest are extremely difficult and expensive to measure in environmental samples.

- Specific pathogens of interest survive in estuarine systems to various degrees and are accumulated by shellfish to various degrees.

- Sufficient fecal coliform data in the study area are available to predict exceedances of regulatory prescribed indicator organism concentrations.

- Sufficient virus and *Giardia* concentration data for untreated wastewater are available to conduct a quantitative assessment of the risk of infection from direct contact with surface water (e.g., through swimming or SCUBA diving) resulting from CSO discharges. Insufficient data are available to assess risks of infection from viruses and *Giardia* from sources other than CSOs.

- Insufficient data are available to estimate risks of infection from specific pathogens in shellfish based on predicted surface water concentrations and the ability of pathogens to bioconcentrate in shellfish.

- Comparison of surface water fecal coliform concentrations to health-protective levels, and estimates of risks of infection from viruses and Giardia will be used in the risk assessment to evaluate the baseline conditions and possible risk reductions associated with CSO control.

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Issue Paper 5: PHYSICAL STRESSORS

CSO discharges can impact the environment either chemically, microbiologically or physically (King County 1997). Of these, evaluating the risks to aquatic life from physical stressors presents a unique set of issues. This is because physical stressors have rarely been evaluated in risk assessments, which have traditionally focussed solely on chemicals released to the environment by human activities. In the Water Quality Assessment (WQA) Problem Formulation, the following physical effects (termed stressors) were identified:

- Release of suspended solids in CSO discharges as well as resuspension of deposited sediments which would increase total suspended solids in the water column (TSS) and increase the amount of settleable solids deposited in the study area (sedimentation);

- Disruption of the sediment surface causing loss of sediment habitat to benthic organisms;

- Reductions in water column salinity;

- Reductions in water column dissolved oxygen;

- Changes in water column acidity (pH);

- Changes in water column temperature; and

- Displacement of fish and other water column organisms by increases in water velocity.

To identify any risks to aquatic life from these physical effects, we must address the following three major issues for each of these effects:

- <u>Define effects thresholds</u>: For each effect, we will need to find out if regulatory standards exist (e.g., EPA or WDOE), or whether we will have to develop a criterion (effects threshold). Developing a criterion will require identifying a method and surveying the scientific literature for relevant data to determine the effect threshold that corresponds to a significant impact on receptor populations.

- <u>Define exposure parameters</u>: We will need to determine how much of each effect is present in the study area. In general, exposure will be defined either from project team knowledge of the river (qualitative), from the WQA sampling program, or from the EFDC hydrodynamic model.

- Characterizing risk: Once the effects threshold and exposure levels have been defined, we will need to decide on how we will combine these to characterize risks to aquatic life. Available methods include (1) comparing the exposure level to the effects level to see if it is exceeded, (2) calculating a specific risk number (e.g., hazard quotient), or (3) using a probabilistic approach to evaluate the community level response.

In addition to each of these major issues, physical effects can be manifested in the area directly around the CSO discharge (termed the nearfield) as well as in areas that are relatively far from the direct CSO discharge (termed the farfield[1]) from individual discharges. Consequently, this paper will identify the scale at which these assessments will be made. Table 5-1 identifies the main issue(s) that must be addressed for each physical stressor we propose to evaluate.

Table 5-1. Main Issues of Physical Stressors Evaluated

Physical Stressor	Main Issue(s)
TSS, Sedimentation, Erosion, and Salinity	No regulatory criteria for TSS, sedimentation, erosion, or salinity effects thresholds are available in Washington State. We need to determine what information we will use to develop these criteria and what methods we will use in their development.
Dissolved Oxygen, Acidity (pH), and Temperature	While regulatory criteria for dissolved oxygen, acidity (pH), and temperature are available, the EFDC hydrodynamic model will not provide data for these effects. Consequently, we need to evaluate what data is available, and whether it will distinguish CSO contributions from other sources.
Displacement	No regulatory criterion for displacement is available. We need to determine what information we will use to develop this criterion and how we will apply it.

5.1 Suspended Solids – TSS/Erosion/Sedimentation

During CSO events, inorganic and organic particulate matter is discharged to the Duwamish Estuary. This particulate matter is composed of both settleable and total suspended solid (TSS) levels. Settleable solids are larger, heavier particles (e.g., sand) that quickly settle to the streambed (e.g., p. 2-57 of APHA 1995). Conversely, TSS are smaller, lighter particles (e.g., silt and clay) that remain suspended for a longer period

1 The EFDC model used to calculate the exposure levels for TSS, sedimentation, erosion, and salinity is a farfield model that calculates a single value for each stressor over the entire cell at each time step.

(e.g., see p. 2-56 of APHA 1995). High TSS levels can be lethal to aquatic organisms. Young fish and invertebrate larvae appear most sensitive. Furthermore, stress to benthic organisms can occur from smothering and interference with filter feeding and breathing (ventilating).

Sedimentation (sediment deposition) is the settling of solids at the sediment-water interface and is measured as the depth of solids accumulating over time. Sediment deposition is a direct measure of how solids can cover and subsequently smother benthic organisms. Erosion is the removal of sediment from existing habitat by currents that resuspend and move sediment downstream. Sedimentation and erosion affects epibenthic and infaunal species. When sediment is deposited at high rates sessile and slow moving species can be smothered. Smothering of the organisms generally lethal and considered an acute effect. Alternatively, benthic organisms can be displaced and exposed to predation by erosion and loss of sediment habitat. Organisms may also leave the area where there is a high sediment deposition rate. Increases in TSS levels and sedimentation rates have been documented as causing mortality to aquatic organisms in Africa, Canada, and the U.S. (Orme 1975; Turk et al. 1980; Lemly 1982).

We propose that the assessment endpoint for TSS, sedimentation, and erosion is the maintenance of sustainable populations of aquatic life. It will be measured by comparing modeled TSS, sedimentation, and erosion rates (measures of exposure) to their respective threshold tolerances (measures of effect) to characterize the risk from increased TSS, sedimentation, and erosion rates.

5.1.1 Suspended Solids – TSS/Erosion/Sedimentation Effects Characterization

Our proposed approach to defining TSS, sedimentation, and erosion effects on aquatic life consists of evaluating effects data from the literature for quality and relevance to the types of exposure characteristic downstream from CSO discharges. TSS data have been selected rather than turbidity to more reliably assess the adverse effects of particulates on aquatic life. This is because turbidity is not a mass-based measurement. It characterizes waters light-scattering properties which generally increases with higher suspended solids. Data are relevant when they address situations applicable to the habitats typifying the Duwamish Estuary; i.e., if they apply to rivers and estuaries with silted bottoms. This criterion excluded considerable literature that focused on gravel bottomed streams. Data for fresh and saltwater species are being evaluated, since there is no evidence that either TSS or sedimentation should affect fresh or saltwater species differently. The data are then compiled into computer spreadsheets to be sorted from the most to least sensitive species.

An initial literature review has discovered 27 review papers and original studies for TSS effects, and 14 review papers and original studies of sedimentation effects on fish and invertebrate communities and individual species. These papers are comprised of studies of effects on fish and invertebrate communities as well as individual species. Each of these studies has been evaluated for appropriateness and quality for all data, and data were rejected for inadequate documentation or the health of control specimens. For example, sedimentation effect data will be rejected if there was inadequate documentation of the health of control specimens or if sediment deposition rates were not reported.

Studies passing this data review will be used to derive effects criteria for TSS, sedimentation, and erosion following the California water quality marine standards process (Klapow and Lewis 1979) and U.S. EPA water quality criteria process (Stephan et al. 1985). This will involve tabulating the data into tables, and using the most sensitive endpoint for each test species, the data will be ranked from most to least sensitive species. Data ranked by sensitivity to the sedimentation or TSS effect will then be used to calculate effect thresholds, defined as the lowest TSS concentrations or sedimentation rates expected to protect 95, 90, 85, and 75 percent of the exposed aquatic species. This approach is consistent with methods proposed and being used for this purpose in Europe (Kooijman 1988; OECD 1992) and in the U.S. (Stephan et al. 1985).

Initial work conducted by Parametrix (1977) has identified the acute and chronic effect thresholds for TSS as 1,000 and 250 mg/L, respectively. These effect thresholds should protect 90 percent of the species[2]. This same effort has identified the chronic effect threshold for sediment settling as approximately 21 mm/month (see Table 1). This sediment settling rate is considerably higher than rates observed in the open ocean, 1 mm/year (ref) and even higher than in a flood influenced estuary, 11 mm/month (Zedler and Onuf 1984). Thresholds providing other levels of ecological protection from sedimentation are reported in Table 5-2.

Table 5-2. Chronic Effects Thresholds for Sedimentation

Percent of Species Protected	Chronic Effect Threshold (mm/month)
95%	21
90%	37
85%	47
75%	60

2 The national criterion development process outlined in Stephan et al. 1985 does specify that criterion can be adjusted in the adoption of state standards to be protective of ecologically important species, which would be the case with a numerically dominant species that would fall in the 5% most sensitive species. In the case of the WQA study, the numerically dominant benthic species observed in the study area was a polychaete, *Pygospio elegans* (See Section 7 of Appendix B4). The most sensitive polychaete species used in the development of the TSS criteria had a NOEC value of 90 mm/month. This suggests that this criteria would be unlikely to under-represent risk to the numerically dominant species.

For erosion, we propose to establish effects levels using literature information on the organisms that occupy different layers of the 0 to 15 cm biologically active zone of sediment (Table 5-3). Thus, for example, loss of the top two centimeters would mean the displacement of 18 different types of taxa from that habitat, constituting a CSO impact.

Table 5-3. Zonation of Groups of Benthic Organisms Occupying the Top 10 Centimeters of Estuarine Sediment

Taxon	Sed. Depth (cm)		Food Habitats
	Upper	Lower	
Chironomidae	0	0	Filters water column/sediment surface feeder
Epitonium spp.	0	0	Predator
Cumella vulgaris	0	0	Surface detrital feeder
Eudorella pacifica	0	0	Surface detrital feeder
Euphilomedes carcharodonta	0	0	Surface detrital feeder
Euchone sp.	0	0.5	Filters water column
Manayunkia aestuarina	0	0.5	Filters water column/sediment surface feeder
Pseudeopolydora kempi	0	0.5	Filters water column/sediment surface feeder
Pygospio elegans	0	0.5	Filters water column/sediment surface feeder
Corophium salmonis	0	0.5	Surface detrital feeder
Corophium spinicorne	0	0.5	Surface detrital feeder
Hobsonia florida	0	0.5	Surface detrital feeder
Oligochaeta	0	1	Sediment feeder
Eogammarus confervicolus	0	1	Surface detrital feeder
Capitella capitata	1	2	Sub-surface sediment feeder
Axinopsida serricata	2	2	Surface detrital feeder
Grandidierella japonica	0	2	Surface detrital feeder
Psephedia lordi	0	2	Surface detrital feeder
Aphelochaeta sp.	0	3	Surface detrital feeder
Macoma carlottensis	1	4	Surface detrital feeder
Cossura pygodactylata	0	5	Sub-surface deposit feeder
Scoletoma luti	1	5	Sub-surface sediment feeder
Neanthes sp.	0	5	Surface detrital feeder
Parvilucina tenuisculpta	0	5	Surface detrital feeder
Clinocardium sp.	0	6	Filters water column/sediment surface feeder
Heteromastus sp.	0	15	Sub-surface sediment feeder

5.1.2 Suspended Solids – TSS/Erosion/Sedimentation Exposure Characterization

Evaluating risks to aquatic organisms in the Duwamish Estuary from TSS in CSO discharges as well as erosion and sedimentation rates will require information on the magnitude of rate increases of TSS, erosion, and sedimentation. Data sources will involve measurements of CSO discharges and river concentrations for TSS, as well as predictions by the hydrodynamic model of TSS concentrations, and erosion/sedimentation rates in specific model cells.

These exposure concentrations will be incorporated into summary statistics that will represent acute and chronic exposures. Acute exposures will be calculated as the 95th percentile of the data population. This value will represent the maximal exposure that an organism could be encounter from a CSO discharge. Chronic exposures will be calculated as the 95th upper confidence limit of the mean. This value will represent the maximal mean concentration that a population of aquatic organisms could encounter over a lifetime. These values will be calculated for the two main impact areas of the study area—the Duwamish River and Elliott Bay

5.1.3 Suspended Solids – TSS/Erosion/Sedimentation Risk Characterization

Estimations of risks will be conducted in a tiered approach. To evaluate TSS, erosion, and sedimentation risks, a hazard quotient will first be calculated for each type of stressor as the ratio of the acute or chronic exposure concentration and the appropriate effects concentrations:

$$\text{Hazard Quotient} = \frac{\text{Exposure Concentration}}{\text{Effects Concentration}} \qquad \text{Equation 5-1}$$

Stressors with hazard quotients greater than 1.0 will be further evaluated probabilistically to determine the number of species at risk in each impact area from that stressor type. Preliminary review of the TSS data collected in the calibration sampling program indicates that it is unlikely that risks from TSS will be found in the river.

5.2 Reduction In Salinity

During and following rainstorms, large amounts of freshwater enter the Duwamish River from CSO discharges. If these are of sufficient magnitude and duration, the salinity in the river's estuarine reaches and Elliott Bay can decline substantially, creating conditions lethal to stenohaline[3] species and stressful to some euryhaline[4] and immobile species. Differing regions of the U.S. have seen kills of organisms following freshwater inundation of these habitats (Boesch et al. 1976; Jarvis 1979). The following discussion explains how we propose to select the different salinity regimens occurring in the Duwamish Estuary and the assessment and measurement endpoints we have identified for this physical stressor.

We propose that the assessment and measurement endpoints for salinity are the same as those defined for chemical stressors. The assessment endpoint, which seeks to maintain sustainable aquatic life populations, will be measured by comparing modeled salinity concentrations and the duration of declines to the salinity tolerances of representative organisms. The measurement endpoint for salinity tolerance will be death or a similar endpoint that translates into adverse effects on the population.

5.2.1 Reduction in Salinity Effects Characterization

As discussed earlier, rainstorms can result in salinity declines in the estuarine portion of the Duwamish River and Elliott Bay. These declines can adversely impact stenohaline and euryhaline organisms. To assess the effects of salinity declines on these organisms in the Duwamish River, salinity tolerance ranges will be identified for different groups of organisms (i.e., freshwater, estuarine, and marine). Within these groups, the types of organisms that would be most at risk (e.g., pelagic fish vs. benthos) will be identified, as well as the durations over which they could tolerate abrupt salinity fluctuations.

To evaluate these objectives, the literature is being consulted for studies that document the salinity tolerance ranges for various aquatic organisms. The first study objective is to define typical salinity ranges that different organisms are able to tolerate. Salinity tolerance ranges are highly species-dependent. The relevance of much of the literature on salinity tolerance ranges of aquatic organisms is difficult to assess because few surveys exist on the distribution of biota along the Duwamish River. Most freshwater fishes are not found at salinities above 3 to 5 parts per thousand (Moyle and Cech 1988). Many species of marine organisms cannot tolerate estuarine situations or low salinities.

Many intertidal invertebrates are osmotic conformers (i.e., they do not expend energy to maintain osmotic concentrations in their body fluids that are different from the environment) (Mitchell et al. 1988). Bivalve mollusks can avoid unfavorable salinities by burrowing or closing their shells. For example, the mussel *Mytilus edulis* closes its shell valve when seawater is suddenly diluted, thereby delaying internal concentration changes

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[3] Stenohaline organisms can tolerate only a narrow range of salinity. Here, they are mainly marine species colonizing Elliott Bay and the lower reaches of the Duwamish River during times of high stable salinity.

[4] Euryhaline organisms can tolerate significant changes in salinity because they are physiologically capable of regulating their ionic balance.

(Lockwood 1976). Still, *Mytilus edulis* can acclimatize and tolerate salinities below 5 ppt (Lockwood 1976). Because the sediment interstitial water salinity approximates the average in the overlying water (Lockwood 1976), mobile benthic invertebrates can adjust their environmental salinity by migrating over short (3 to 6 cm) distances.

The second study objective will be to identify which types of estuarine and marine organisms would be most at risk from salinity declines. Benthos that cannot avoid freshwater (e.g., by closing their shells or emigrating) appear to be those at greatest risk. In the Georgetown reach of the Duwamish River and Elliott Bay, a saline lens of water will persist along the bottom, in the absence of high river flows scouring out the saline layer, thereby maintaining a saline environment for the benthos. In addition, planktonic organisms, which include algae, zooplankton, and the larval stages of fish and invertebrates, can passively or actively sink to the more saline water below (Lockwood 1976).

This information will be used to develop specific scenarios to predict the conditions that will result in risk to aquatic organisms intolerant of the changes in salinity that can be encountered in the Duwamish Estuary following a CSO discharge.

5.2.2 Reduction in Salinity Exposure Characterization

To assess risks to aquatic organisms in the Duwamish River estuary from an influx of freshwater and decline in salinity requires an understanding of the magnitude and duration of declines. With this information, and knowledge of the salinity tolerance ranges of the selected receptors, risks to aquatic organisms can be estimated. Salinity declines in the Duwamish River estuary will be estimated with the same model used for the chemical stressors. As discussed in earlier, the aquatic receptors being evaluated are stenohaline marine organisms and euryhaline estuarine organisms that may be stressed from rapid, large declines in salinity or prolonged exposure to freshwater. These scenarios will be the focus of the exposure characterization. Using these scenarios and the data provided by the hydrodynamic EFDC model, areas will be identified within the Duwamish Estuary that would result in exposure of the sensitive species identified above.

5.2.3 Reduction in Salinity Risk Characterization

Potential risks from salinity declines to the aquatic receptors being evaluated (i.e., stenohaline and euryhaline aquatic organisms) will be estimated by combining the results of the exposure and effects characterizations. The salinity tolerances of Duwamish River estuarine biota have not been fully characterized; therefore, risks are estimated based on the responses of representative groups of organisms. For example, between the turning basin and Elliott Bay, certain euryhaline organisms unable to withstand or avoid the salinity stress are expected to be at risk. Stenohaline organisms are not expected to reside in this portion of the estuary because salinity varies too much for them. As stated above, the presence or absence of a saline lens at the river's bottom would dictate which types of organisms would be most at risk. For example, if a saline lens persists, certain bottom-dwelling organisms appear protected from the overlying freshwater. If CSO discharges result in completely freshwater conditions in the river where normally a salt wedge is present, non-mobile benthos like polychaetes, oligochaetes and mollusks are at greatest risk, followed by plankton and relatively immobile fish and invertebrates. Although some mobile organisms may be able to avoid the freshwater, it is the abrupt drop in salinity that may stress the organisms. The results of this analysis will be description of areas of risk generated by salinity reductions related to CSO discharges.

5.3 Reduction In Dissolved Oxygen

Reductions in dissolved oxygen (DO) may result from the biochemical oxygen demand (BOD) of particulate matter in CSO discharges (Welch and Lindell 1992). The effects of CSO discharges on receiving water DO levels may be mediated by the increased river flow which can occur in conjunction with CSO discharge events (SPCC 1981; Welch and Lindell, 1992). Those waters near and in the sediment typically have the lowest DO concentrations, owing to the degradation of organic matter (sediment oxygen demand). In urban environments, sediment DO is often well below the minimum threshold of 3 mg/L (Davis 1975a,b) due to the cumulative effects of waste and nutrient loading (ERF 1997; NOAA 1996; Rabalais and Harper 1992).

Reductions in DO can effect the behavior, metabolism, growth, reproduction, and survival of aquatic organisms. Key variables defining risk are the magnitude and duration of the DO minima, and the species-specific adaptations to living in low and fluctuating DO environments. Typically, early life stages except embryos are the most sensitive phases, with juveniles appearing to be the most sensitive (U.S. EPA 1986a). Fish seem to be the aquatic life group most sensitive to DO declines, owing to their higher metabolic requirements per unit mass.

Receptors evaluated in this pathway of the risk assessment excludes fish inhabiting the upper water column, but includes benthic invertebrates, bottom-dwelling fish, plankton, and other relatively immobile aquatic organisms. These represent aquatic life that cannot readily avoid areas of low DO by emigrating. It is expected that pelagic fish will be able to emigrate or find refugia of elevated dissolved oxygen.

Aquatic organisms use several compensatory mechanisms when they encounter low DO (U.S. EPA 1986a). Mobile organisms respond through avoidance, but those with limited or no mobility have to create their own currents through body movements, close their shells (bivalves) or suspend activity. Organisms with limited mobility may also make physiological compensations for low DO. These may involve reduced heart rate and increases in both hemoglobin, red blood cells and similar respiratory tissues (Heath 1995).

The assessment endpoint for DO is the maintenance of viable populations of aquatic life downstream of CSOs. Data on the effects of DO on aquatic life comprise the measure of effect. The measures of exposure are DO concentrations at specific river locations.

5.3.1 Reduction in Dissolved Oxygen Effects Characterization

In contrast with chemical stressors, it is low DO concentrations that can result in adverse effects; thus, DO effect thresholds are minima not maxima. The State of Washington has already established a series of dissolved oxygen criteria designed to protect aquatic life based on the classification of surface bodies by the State (WAC 1995). The Duwamish River has been designated Class B – Good. The dissolved oxygen criteria associated with this classification are:

- Freshwater — Dissolved oxygen shall exceed 6.5 mg/L
- Marine Water — Dissolved oxygen shall exceed 5.0 mg/L. When natural conditions, such as upwelling, occur, the dissolved oxygen can be degraded by up to 0.2 mg/L by human –caused activities.

We propose using these criteria to evaluate the calibration sampling program data to determine the risk to aquatic life from reductions in dissolved oxygen.

5.3.2 Reduction in Dissolved Oxygen Exposure Characterization

No data will be available from the hydrodynamic/EFDC model concerning dissolved oxygen concentrations. This is because the Biological Oxygen Demand (BOD) component was not implemented due to programming needs and computing power. Consequently, exposures to reduced dissolved oxygen concentrations will be characterized from the calibration sampling program data.

5.3.3 Reduction in Dissolved Oxygen Risk Characterization

As presented above for other stressors, risks from reduced dissolved oxygen concentrations will be characterized by calculating a hazard quotient for the Duwamish River impact area and the Elliott Bay impact area. If either area has hazard quotients greater than 1.0, they will be further evaluated probabilistically to determine the number of species at risk in each impact area. Preliminary review of the dissolved oxygen data collected in the calibration sampling program indicates that it is unlikely that risks from TSS will be found in the river.

5.4 Change in pH

Changes in pH levels are typically not directly toxic to aquatic life, but instead can result in risk from the effect of pH on other toxicants (U.S. EPA 1986b). A review of the effects of pH on freshwater fish by the European Inland Fisheries Advisory Commission (1969) (as cited in U.S. EPA 1986b) concluded:

> *"There is no definite pH range within which a fishery is unharmed and outside which it is damaged, but rather, there is a gradual deterioration as the pH values are further removed from the normal range. The pH range which is not directly lethal to fish is 5-9; however, the toxicity of several common pollutants is markedly affected by pH changes within this range, and increasing acidity or alkalinity may make these poison more toxic. Also, an acid discharge may liberate sufficient CO2 from bicarbonate from the water ether to be directly toxic, or to cause the pH range 5 to 6 to become lethal."*

We propose that the assessment endpoint for dissolved oxygen is the maintenance of sustainable populations of aquatic life. It will be measured by comparing field evaluations of pH to freshwater and marine threshold tolerances (measures of effect) to characterize the risk from decreases in dissolved oxygen.

5.4.1 Change in pH Effects Characterization

As stated above, pH has both maximum and minimum criteria for the protection of aquatic life. The State of Washington has already established a series of pH criteria designed to protect aquatic life based on the classification of surface bodies by the State (WAC 1995). The dissolved oxygen criteria associated with the B classification of the Duwamish River are:

- Freshwater pH shall be within the range of 6.5 to 8.5 with human-caused variation within a range of less than 0.5 units
- Marine Water pH shall be within the range of 7.5 to 8.5 with human-caused variation within a range of less than 0.5 units

We propose using these criteria to evaluate the calibration sampling program data to determine the risk to aquatic life from changes in pH.

5.4.2 Change in pH Exposure Characterization

No data will be available from the hydrodynamic/EFDC model concerning pH levels. Consequently, exposures to changes in pH levels will be characterized from the calibration sampling program data.

5.4.3 Change in pH Risk Characterization

As presented above for other stressors, risks from changes in pH will be characterized by calculating a hazard quotient for the Duwamish River impact area and the Elliott Bay impact area. Two sets of hazard quotients will be calculated using by the lower and upper criteria appropriate for the classification of each impact area. If either area has hazard quotients greater than 1.0, they will be further evaluated probabilistically to determine the number of species at risk in each impact area. Preliminary review of the pH data collected in the calibration sampling program indicates that it is unlikely that risks from pH will be found in the river.

5.5 Change In Temperature

Temperature is a critical measure of the suitability of a particular environment for the presence of specific aquatic life species. Subsequently, temperature changes in water bodies can alter the existing aquatic life community (U.S. EPA 1986b). Both algal and fish communities will change as the temperature increases. Sufficient increases in temperature can change a coldwater fishery to a warm-water fishery as temperature may be directly lethal to adults or fry, causing a reduction of activity or limitation of reproduction (Brett 1960; U.S. EPA 1986b). Below lethal upper and lower temperature limits, temperature can dramatically affect diet, activity, and the general health of aquatic organisms.

We propose that the assessment endpoint for temperature is the maintenance of sustainable populations of aquatic life. It will be measured by comparing field evaluations of temperature to freshwater and marine threshold tolerances (measures of effect) to characterize the risk from increases in temperature.

5.5.1 Change in Temperature Effects Characterization

The State of Washington has established a series of temperature criteria designed to protect aquatic life based on the classification of surface bodies by the State (WAC 1995). The temperature criteria associated with the B classification of the Duwamish River are:

- Freshwater — Temperature shall not exceed 21^{o}C due to human activities. Incremental temperature increases resulting from point source activities shall not, at any time, exceed t=34/(T+9)[a]. Incremental increases resulting from nonpoint source activities shall not exceed 2.8^{o}C

- Marine Water — Temperature shall not exceed 19^{o}C due to human activities. Incremental temperature increases resulting from point source activities shall not, at any time, exceed t=16/T. Incremental increases resulting from nonpoint source activities shall not exceed 2.8^{o}C

[a]"t" represents the maximum permissible temperature increase measured at a mixing zone boundary; "T" represents the background temperature as measured at a point or points unaffected by the discharge and representative of the highest ambient water temperature in the vicinity of the discharge.

We propose using these criteria to evaluate the calibration sampling program data to determine the risk to aquatic life from changes in temperature.

5.5.2 Change in Temperature Exposure Characterization

No data will be available from the hydrodynamic/EFDC model concerning changes in temperature. Consequently, exposures to changes in temperature will be characterized from the calibration sampling program data.

5.5.3 Change in Temperature Risk Characterization

As presented above for other stressors, risks from changes in water column temperature will be characterized by calculating a hazard quotient for the Duwamish River impact area and the Elliott Bay impact area. If either area has hazard quotients greater than 1.0, they will be further evaluated probabilistically to determine the number of species at risk in each impact area. Preliminary review of the temperature data collected in the calibration sampling program indicates that it is unlikely that risks from increases in temperature will be found in the river.

5.6 Displacement

Influx of water from a CSO discharge can result in displacement of organisms from appropriate habitats (based on salinity, temperature and structure providing escape from predators) to habitats that are inappropriate. This displacement could result in acute, lethal and chronic, sub-lethal effects. However, it will not be possible to determine which affect is occurring as the displacement merely makes organisms vulnerable to other kinds of stresses. We propose that the assessment endpoint for displacement is the maintenance of salmonid outmigrants in the Duwamish River. It will be measured by comparing nearfield model predictions of river velocities following CSO discharges to sustainable swimming speeds of salmonid fry and smolt (measures of effect) to characterize the risk from increases in river flow.

5.6.1 Displacement Effects Characterization

Juvenile salmonids are capable of two types of swimming—burst and sustained swimming. Burst swimming represents maximum swimming speeds that juveniles would use to escape predation. Typically, juvenile salmonids can only maintain burst swimming speeds for approximately 10-20 seconds, a time period unreflective of CSO discharges (Feist and Anderson 1991). Sustained swimming speeds can maintained for periods of minutes to hours, dependent on the type of species, as well as the age and physical condition of the individual fish. For example, Chinook salmon that are 3 to 5 cm long are capable of swimming at 1.0 to 1.2 fps velocities for an hour or more (Feist and Anderson 1991). In contrast, Brown, brook, and lake trout, and Atlantic salmon have critical velocities > 1.6 fps (Feist and Anderson 1991). These fish are unlikely to be able to maintain themselves at these speeds for more than 20 to 30 minutes.

Alternatively, a recent presentation by Washington Department of Fish and Wildlife (WSDFW February 1997) on coho fry and fingerling passage up culverts indicated that juvenile swimming speeds might not be relevant indicators of passable river velocities. This is because juveniles use the turbid boundary layer along the substrate to "rock hop" upstream in high velocities. Thus, measurable water column velocities don't bear much relation to juvenile swimming speeds. The culvert study found that 100 percent of coho fry were able to pass through a smooth-walled 12-in diameter pipe at 1 fps, but no fish passed at velocities greater than or equal to 2 fps. Fingerlings in a corrugated culvert were able to use the corrugations to withstand average velocities of 4.6 fps. Consequently, the critical issue in setting a displacement velocity is not "what river velocities can juveniles tolerate?" but "what happens to the boundary layer velocity (location and area) when velocities increase?" We will continue to investigate this issue to establish a sustained swimming speed effects threshold to evaluate the potential for displacement of salmonid fry by CSO discharges.

5.6.2 Displacement Exposure Characterization

Velocities in the vicinity of CSO discharge locations will be predicted by the nearfield component of the hydrodynamic/EFDC model. This component of the model will predict the speed and direction of river flow resulting from the specific CSO discharge at each location.

5.6.3 Displacement Risk Characterization

As presented above for other stressors, risks from exceeding displacement velocities will be characterized by calculating a hazard quotient for each CSO location. Risks to salmonid fry will be identified as being associated with that specific CSO discharge when hazard quotients are greater than 1.0.

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Issue Paper 6: AQUATIC LIFE AND WILDLIFE TOXICOLOGY

To assess ecological risks[5] from chemicals in the Duwamish River/Elliott Bay WQA, we need to identify the minimum amount of a chemical present in water or sediment that would harm invertebrates or fish (the "effects level"). We will identify the effects level for each chemical being evaluated using the basic principles of toxicology, as well as a series of assumptions founded on these principles. This paper presents these elements of toxicology, and explores the basis for the assumptions we will make in this evaluation.

6.1 The Principles of Toxicology[6]

Toxicology has three basic principles that we rely on in assessing any risks posed by chemicals in the environment. These principles are:

- Cause and effect relationships exist

The existence of cause and effect relationships between the exposure to a chemical and the observed effect in the exposed animal is a cornerstone of toxicology (Bacci 1994, Rand 1995). This principle allows us to quantify the effects of exposure to a chemical of interest. However, until it is experimentally demonstrated that exposure to a specific amount of chemical results in a particular response, this relationship is only a reasonable working hypothesis.

- Concentration/Dose[7]-response relationships exist

An additional foundation of toxicology is the idea that "it's the dose that makes the poison." Stated another way—increases in concentration or dose, result in increases in the effect on the exposed animals over a specific duration. Above some threshold, the magnitude of the effect is proportional to the amount of toxic chemical reaching the site of toxic action (the tissue or organ within the exposed animal where the toxic effect is realized).

- Effects of toxic chemicals are reproducible

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5 For the purposes of this paper, ecological risks will be defined as the risks posed to aquatic life and wildlife by chemicals in the environment.

6 These general principles are taken from "Fundamentals of Aquatic Toxicology" by G. Rand, editor, 1995.

7 Concentration/dose refers to the amount of a chemical to which an organism is exposed. Aquatic organisms are exposed to concentrations from the water or sediment in which they live. Wildlife receive a dose from the food and water they ingest.

A variety of different effects on animals exposed to toxic chemicals have been measured within the field of toxicology. The ability to measure and tally both the exposure concentrations and effects of these concentrations is required to sustain the first two principles. In order to be evaluated in a risk assessment, the effects of a toxic chemical must be both measurable and repeatable (Calow, 1993).

6.2 Statement of Issues

Having stated these principles, there are a number of data gaps that must be overcome in applying them to any quantitative risk assessment. These data gaps represent the issues that need to be addressed in the conduct of the Duwamish Estuary WQA. These issues are:

- *Some chemicals being evaluated in the WQA have not been tested for toxicity.*

  The effort by the scientific community to determine chemical toxicity has not kept pace with the development and introduction of the approximately 30,000 chemicals currently in use in the United States (U.S. EPA 1994; Rand 1995). When such data are unavailable for a risk assessment, it is necessary to extrapolate from available toxicity information in order to conduct the risk assessment.

- *Not all WQA receptor species have been tested with every chemical of interest.*

  The receptor species selected for the WQA reflect the site-specific organisms that are potentially at risk in the Duwamish estuary. However, these organisms (e.g., bald eagles, river otters, fish and aquatic invertebrates) have only rarely been tested for toxicity with the chemicals of interest for our study. Reasons for this range from restrictions on use (such as the bald eagle being an endangered species) to the practicalities of using these organisms as test species. When species-specific data are unavailable, it is necessary to extrapolate from the response of surrogate species in order to conduct the risk assessment.

- *Only a limited range of concentrations/doses and exposure durations have been tested for some chemicals being evaluated in the WQA.*

  Typically, the literature studies available for each combination of receptor type and chemical did not examine the specific range of concentrations/doses and exposure durations that are required for a study such as the WQA. Frequently, many studies fail to bracket the threshold concentration that produces the specific effect that we require for our study. This means testing a concentration that just produces the effect (e.g., death of 50% of the exposed animals) and a concentration just under this concentration that does not produce this effect. Additionally, available literature studies have not been conducted for the same exposure durations that relate to the site-specific conditions of the Duwamish Estuary. When only effect producing concentrations or no-effect producing concentrations have been studied, it is necessary to extrapolate from one concentration to the other to conduct the risk assessment. Additionally, when inappropriate test durations have been used, it is common practice to apply safety factors[8] to tests to lower the effect concentration for use in the risk assessment.

- *A limited range of effects/endpoints has been evaluated for some of the chemicals being evaluated in the WQA.*

  Risk assessments such as the WQA are commonly based on a set of effects (termed endpoints) such as acute and chronic responses. Acute responses are those that result in the death of some percentage of the exposed organisms. Chronic responses are those that result in sub-lethal effects such as decreases in growth and reproduction. Additional endpoints can be evaluated such as biomarkers, which are biochemical or physiological responses in exposed organisms. Additionally, assessed endpoints can be the result of interactions between the chemicals being evaluated. Chemical interactions can involve additivity (where effects add linearly), synergism (where the resultant effect is more the sum of the individual components), and antagonism (the resultant effect is less than the sum of the individual components). In conducting a risk assessment and setting adverse levels for identifying risk, it is necessary to specify the effects and the possibilities for interaction between chemicals that are relevant to the receptors being evaluated.

- *Extrapolation from laboratory to field conditions.*

  The vast majority of toxicity studies have been conducted under laboratory conditions. The scientific and regulatory community has determined that the benefit of being able to carefully control the conditions under which these tests are conducted far out-weighs the loss of site-specificity. These benefits are the ability to control organism health, age, diet, and exposure amount and duration that allow attribution of observed effects to the chemical being tested. These tests allow us to rule out competing stresses (e.g., predation, other chemicals) that could confound the dose-response relationship. Additionally, laboratory tests allow for repeatability that is required under the principles identified above. Typically, none of these factors can be controlled in the field, severely limiting our ability to attribute observed effects solely to the

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8 A safety factor is a value that is designed to ensure some greater margin of conservatism. In the context, of risk assessment, a safety factor is employed to account for possible greater sensitivity of the target receptor.

chemical of interest. However, use of laboratory tests requires us to extrapolate to field conditions in order to conduct the WQA.

- *Percent of species protected by an aquatic Toxicity Reference Value (TRV).*

  The techniques used to develop ambient water quality criteria (AWQC) used by the U.S. EPA and the State of Washington require identification of the percent of the community of species to be protected. The current practice in the United States is to protect 99 percent of the individuals in 95 percent of the exposed species from adverse effects[9]. This is achieved by using a species effect curve, which represents the cumulative impact of increasing concentrations on the community. In the conducting the WQA, we will strive to identify specific species of interest (particularly salmonids) in the species effect curves we will use.

Every risk assessment conducted must address these issues, and develop strategies for filling these gaps. The following sections expand on the issues presented above, and how we will address them in the ecological risk assessment.

6.3 Selecting Toxicity Reference Values

A fundamental component of any ecological risk assessment is determining the toxic effects threshold concentration (called the toxicity reference value or TRV), against which environmental concentrations are compared to identify risks. For evaluating aquatic life, it is possible to adopt standards developed by regulatory agencies (e.g., EPA or Washington Department of Ecology) for evaluating chemical risks. If such standards are unavailable (or when the assumptions used in developing these standards don't apply to a particular study), it is necessary to develop a TRV from research reported in the scientific literature. No state or national water quality standards have been adopted for the specific protection of wildlife. Consequently, the following discussion addresses aquatic life and wildlife separately.

6.3.1 Aquatic Life TRVs

When selecting a water column TRV to use in evaluating aquatic ecological risks in Washington, there are a number of potential choices.

Potential water column values are:

- Washington State surface water quality standards;

[9] This level can be superceded by the need to protect species of special concern (i.e., endangered) or species of particular commercial value. Thus concentrations can be set below the 95 percent effect concentration to protect this specific species.

- Federal ambient water quality criteria;
- Scientific literature studies.

Potential sediment values are:

- Washington State Sediment Management Standards;
- U.S. EPA Sediment Quality Criteria;
- Ontario Freshwater Sediment Guidance (Persaud *et al*. 1993);
- Scientific literature studies (Long and Morgan 1990, Ingersoll *et al*. 1996);
- Calculation methods to for untested sediment chemicals—Ecotox Threshold (U.S. EPA 1996) and equilibrium partitioning (Di Toro et al. 1991, Ankley et al. 1996).

To address the variety of available criteria, we developed a selection hierarchy that is presented in the Analysis Plan (Parametrix 1997a) and reproduced here in Tables 6-1 and 6-2. Water quality and sediment criteria will be selected following these hierarchies. If no U.S. EPA ambient water quality criterion is available, then the lowest literature value available will be used after applying a safety factor of 20. Additionally, when only acute values are available for chemicals, an acute to chronic ratio (ACR) will be applied to develop a chronic value. Examples of chemicals for which we have applied an ACR to develop the chronic value are 1,4-dichlorobenzene, total PCBs, chrysene, and benzo(a)pyrene.

A safety factor of 20 is derived from the Tier II process developed in the Final Water Quality Guidance for the Great Lakes System (U.S. EPA 1995). This guidance established a procedure for extrapolating from the National Guideline requirements of eight genus mean acute values (GMAV) representing specific taxa needed to establish a criterion to data sets containing less information. This process applies ever-increasing safety factors as you move from data sets containing seven GMAVs to those with only one. An adjustment factor of 21.9 was recommended by EPA for use with only one study. This is modified in this approach to a safety factor of 20 for all chemicals, regardless of the number of studies available.

Sediment criteria are selected following the same hierarchical process using the criteria identified in Table 2. For some sediment chemicals, the U.S. EPA has published criteria developed using the Tier II process termed Ecotox Thresholds (U.S. EPA 1996). Finally, if no sediment criteria are available for a nonionic organic chemical, then one will be calculated using the equilibrium partitioning approach (Di Toro et al. 1991) to compare with the chronic water quality criteria.

Table 6-1. Water Column Selection Hierarchy

Freshwater Criteria	Washington State Surface Water Quality Standards (Title 173-201A WAC), or U.S. EPA Ambient Water Quality Criteria (AWQC) (U.S. EPA 1994), or Parametrix Criteria for Manganese and Cobalt (Parametrix 1997b), or Lowest Literature Value Divided by 20
Saltwater Criteria	Washington State Surface Water Quality Standards (Title 173-201A WAC), or U.S. EPA Ambient Water Quality Criteria (U.S. EPA 1994), or Parametrix Criteria for Manganese and Cobalt (Parametrix 1997b), or Lowest Literature Value Divided by 20, or Freshwater Criterion when no saltwater criterion/literature values are available

Table 6-2. Sediment Criteria Selection Hierarchy

Freshwater Criteria	U.S. EPA Sediment Quality Criteria[10],or Ecotox Threshold (U.S. EPA 1996), or Long and Morgan (1990), or Ingersoll et al. (1996), or Ontario Freshwater Sediment Guidance (Persaud et al. 1993), or Application Equilibrium Partitioning (EqP) to Chronic Water Quality Criteria after applying the 20 safety factor
Saltwater Criteria	Washington State Sediment Management Standards (Title 173-204 WAC), or Long et al. (1995), or Ecotox Threshold (U.S. EPA 1996), or Application Equilibrium Partitioning (EqP) to Chronic Water Quality Criteria after applying the 20 safety factor

6.3.2 Wildlife TRVs

As no U.S. EPA or state of Washington[11] wildlife criteria or standards currently exists, wildlife toxicity values will be obtained from the scientific literature. The toxicity data for wildlife receptors used in this risk assessment will be reviewed from the U.S. Fish and Wildlife Service Contaminant Hazard Review series (e.g., Eisler 1986), the Agency for Toxic Substances and Disease Registry (ATSDR) documents (e.g., ATSDR 1994), and from the general scientific literature.

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10 U.S. EPA Sediment Quality Criteria are found in U.S. EPA 1993a, 1993b, 1993c, 1993d, and 1993e.

11 The state of New York has developed such criteria for PCBs (Newell *et al*. 1987) that will be evaluated in this project.

Both chronic and acute data will be identified. Chronic toxicological effects (e.g., those effects elicited following exposure over a significant portion of an animals' lifetime) were generally based on the highest reported no-observed-adverse-effect levels (NOAELs) or lowest-observed-adverse effect levels (LOAELs) for reproductive and developmental effects. In the absence of toxicological data for these preferred endpoints, impacts on growth or other systemic effects were substituted. Systemic effects include organ toxicity, effects on the immune system, or a decrease in body weight. A safety factor of 10 will be used to extrapolate from a LOAEL to a NOAEL, as recommended in EPA guidance (EPA 1996).

Safety factors are used when there are limited toxicity data available for a chemical or the receptor being evaluated. Safety factors are applied to reduce the possibility of eliminating a chemical that may pose risk because data are lacking for a specific species and exposure period. For some chemicals, only acute (short-term) toxicity data are available. For these chemicals, a chronic (long-term) toxicological effect level must be estimated (effects from chronic exposures rather than acute are typically assessed). Other safety factors include extrapolations from an effect level to a no-effect level, and extrapolations from one species to another.

Since few toxicity data exist for the spotted sandpiper, great blue heron, or bald eagle, surrogate species will be used. Most toxicological effect data are available for domestic and laboratory animals, such as rats, mice, chickens and quail. The standard laboratory animal that is the most closely related (in terms of diet and phylogeny) to the wildlife receptor will be selected as the surrogate to derive a toxicological no effect (“safe”) dose.

Surrogate species are typically used to assess adverse toxicological effects to wildlife species of concern because there are limited laboratory toxicity data available for most wildlife species. Typically, toxicological effect data are available for domestic, livestock, and laboratory animals, such as rats, mice, chickens, mallards and quail. When surrogate species are used to assess the toxicity of a chemical to a site receptor, the test species that is most closely related (in terms of phylogeny[12]) is typically used. In doing this, it is assumed that toxicological responses are similar or more similar then they would be across those species not closely related. Because laboratory toxicity data are not available for most wildlife species, there are levels of uncertainty in assessing the potential toxicity of a chemical to a wildlife species.

The wildlife toxicity data used are based on daily dose levels normalized for body weight of the test species. Evaluating wildlife toxicity data on an amount of chemical per body weight per day (i.e., mg/kg/day) allows for comparisons across tests and species (Sample *et al*. 1996). For mammals, an additional modification to the toxicity dose is recommended, that is, scaling the dose based on the body weight of the test and receptor species. This is performed because studies have shown that numerous physiological functions such as metabolic rates and responses to toxic chemicals, are functions of body

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[12] The lines of descent or evolutionary development of any plant or animal species.

size (Sample *et al*. 1996). Differences in metabolic rates can lead to more resistance to toxic chemicals because of the rate of detoxification through metabolism and excretion of the chemical (Sample *et al*. 1996). This relationship of scaling the dose for body weights has been shown for mammals (Travis and White 1988, Travis *et al*. 1990 and U.S. EPA 1992). Body weight scaling, however, has not been found to be appropriate for birds (Fischer and Hancock 1997). For birds, differences in toxicological reactions appear to be more a factor of when comparing passerine[13] and nonpasserine birds (Fischer and Hancock 1997).

6.4 Laboratory Studies and Field Populations

We would like to discuss further the broad issue of how we can use the available laboratory-based literature data to assess risks to field populations of aquatic life and wildlife. It is important to appreciate that predictive assessments of the effects of chemicals on fish, birds, and mammals, such as the Duwamish River/Elliott Bay Water Quality Assessment (WQA), inevitably involve extrapolations from laboratory studies to the field and across species due to the limited amount of available information (Suter 1993). This has been a major issue in ecotoxicology for years (e.g., see Suter et al. 1985). For example, for even a well-tested chemical like phenol, the tested species constitute less than 1 percent of the North American ichthiofauna (Suter et al. 1985). The U.S. EPA water quality criteria explicitly adopt the laboratory to field extrapolation approach (e.g., see sections addressing "Applicability of National Water Quality Criteria To Specific Sites" and "National Water Quality Criteria versus Field/Microcosm Studies" in Hansen (1989). Many studies sponsored by U.S. EPA have shown that laboratory derived water quality criteria are protective in field situations (Mount et al. 1984, 1985, 1986a-c, Mount and Norberg-King 1985, 1986, Norberg-King and Mount, 1986).

Additionally, only laboratory tests have been conducted on the vast majority of those chemicals that have been tested for toxicity. The reasons for this are the difficulty of conducting these tests in the field and the degree of control afforded by laboratory tests over field experiments. For example, the U.S. EPA has stated that

> *"if it were feasible, a freshwater (or saltwater) numerical aquatic life national criterion[14] for a material should be determined by conducting field tests on a wide variety of unpolluted bodies of fresh (or salt) water. It would be necessary to add various amounts of the material to each body of water in order to determine the highest concentration that would not cause unacceptable long-term or short-term*

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[13] Passerine are the perching songbirds (such as starlings) and account for approximately half of all known bird species. These birds have a very different physiology that influences their response to toxicants and distinguishes them from the nonpasserine birds (such as ducks and chickens).

[14] A numerical aquatic life national criterion is a specific concentration of a chemical that is protective of the aquatic life in that water body.

> *effect on the aquatic organisms or their uses. The lowest of these highest concentrations would become the freshwater (or saltwater) national aquatic life criterion for that material." (Stephan* et al. *1985).*

However, this approach has proven infeasible due to a variety of difficulties, such as locating populations of animals not currently exposed to man-made stressors that can confound the field study. Consequently, the scientific and regulatory community has turned to laboratory tests to determine chemical toxicity. These laboratory tests are used over field studies to derive toxicity dose-response data because the laboratory setting provides more control over other stressors (e.g., predation) that can confound the results of the study. Confounding factors that can be controlled in a laboratory study, but not in a field study, include other chemical stressors (e.g., PCBs, DDT), predation, malnutrition, competition from other species, and physical stressors, such as flooding and abnormal ambient temperature changes. When these other stressors are acting upon a population of organisms, it is difficult, if not sometimes impossible, to develop a cause and effect relationship for the chemical toxicant of concern. For these reasons, field data will be difficult to interpret.

The receptors being assessed at our site will not be exposed to same combination of stressors as a species at other sites. Therefore, even if a chemical concentration is the same at two sites, effects will vary across sites because of other stressors. If these other stressors are not comparable, the effects seen at one site may not be used to determine effects at another site because of differences in other stressors (e.g., predation). The advantage of laboratory studies is that they allow us to control other stressors and obtain an unconfounded estimate of a chemical's toxicity.

Thus, risk assessments have explicitly assumed that toxicity studies of laboratory populations of fish, birds, and mammals can be extrapolated to field populations (Hoffman, et al. 1990). This assumption has been validated for the National Water Quality criteria which were established to be protective of aquatic life (and wildlife and humans for specific chemicals) (e.g., see Arthur 1988; Geckler et al. 1976; Mount et al. 1984, 1985, 1986a-c, Mount and Norberg-King 1985, 1986, Norberg-King and Mount 1986). These criteria have been field tested in field/microcosm studies, and the results from single species tests predicted effects on populations in these experimental streams and marine microcosms (Arthur 1988; Hansen 1989). Extrapolations for birds and mammals attempt to limit the safety involved by limiting the taxonomic distance (e.g., only extrapolating from birds to birds and not birds to mammals). These extrapolation concepts have been proven to work for wildlife (Sample et al. 1996) as well as aquatic life (i.e., see Suter and Rosen 1986).

Techniques have been advanced to extrapolate from chemical toxicity data generated from laboratory studies to receptor populations and communities (Kooijman 1987; OECD 1991; Emans et al. 1993). The WERF methodology being used in the WQA is one of these (Parkhurst et al. 1994; Cardwell et al. 1993). This method compares the chemical sensitivity of a number of test species (represented in a species-sensitivity curve) to the range of chemical concentrations (the expected environmental concentration curve) found at the study site. This comparison allows us to estimate what percentage of all species in the community a particular chemical adversely impacts. This provides an estimate of community risk. A community extrapolation approach also is used in setting water quality criteria that represent the chemical concentration protecting 95 percent of the species in a generic aquatic ecological community (Stephan et al. 1985).

6.5 Predicting Toxicity Of Untested Chemicals

Risk assessors faced with evaluating chemicals for which no toxicity data are available have two options—identify a data gap and say risks are totally uncertain, or extrapolate across chemicals to predict the untested chemical's TRV. For aquatic life, these predictions can be calculated using structure-activity relationships (SARs) based on the similarity between chemicals for which aquatic toxicity has been previously measured and untested chemicals (U.S. EPA 1994; Rand 1995). These relationships have been developed by toxicologists. They relate toxicity to the molecular weight of a compound and the measured or calculated "octanol/water partition coefficient" (K_{ow})[15]. The U.S. EPA has regularly used this approach since 1981 to predict the aquatic toxicity of new industrial chemicals in the absence of test data (U.S. EPA 1994).

Structure-activity relationships are regression equations relating the water solubility of a chemical (as measured by K_{ow}) to measured aquatic toxicity for a class of chemicals (U.S. EPA 1994). Using the K_{ow} (either measured or estimated) and molecular weight of the untested chemical with the class specific regression equation, it is possible to predict an aquatic toxicity value for this chemical. This approach has been used to develop toxicity relationships for fish (both fresh and saltwater), water fleas (daphnids), green algae, and earthworms (U.S. EPA 1994).

We have only used this technique to develop TRVs for high-molecular polyaromatic hydrocarbons (specifically pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, dibenzo(a,h)anthracene, benzo(g,h,i)perylene, and benzo(e)pyrene) in developing the screening criteria used to identify COPCs for this project. We propose to use these same values in conducting the aquatic life risk assessment of the WQA.

No such relationships are regularly used to predict toxicity for mammals or birds. In contrast, the common practice has been to use toxicity values of structurally related chemicals in developing wildlife TRVs. Toxicity data for these chemicals, known as surrogates, are used for directly for closely related chemicals. Examples are, again, for HPAHs where test values for a four ring HPAH will be used for other four ring HPAHs for which no data are available. While differing in specific approach, the concept of

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15 The octanol water partition coefficient is a measure of how a chemical partitions between an organic phase (octanol) and water. A chemical with a low K_{ow} tends to be found dissolved in water; a chemical with a high K_{ow} tends to associate with organic material. Chemicals with different K_{ow}s tend to have different toxicity because they occupy different environmental media.

structurally and physically related chemicals having the same or similar toxic effect is the same.

6.6 Biomarkers

The WQA ecological risk assessment follows the current United States practice of measuring environmental concentrations in abiotic[16] media (e.g., air, water, and sediment) to compare with toxicity reference values. An alternative approach that is gaining interest in the risk assessment community to is to measure biochemical, physiological, and historical changes in organisms to estimate either exposure to chemicals or resultant effects. These changes in an organism are collectively referred to as *biomarkers* (Huggett et al. 1992). One clear advantage of using biomarkers in a risk assessment is a reduction in the uncertainty in exposure modeling. This is because a biomarker that has been clearly linked experimentally to a particular chemical can provide direct evidence that exposure has occurred and, in some cases, the amount of exposure. This advantage only applies to risk assessments of current conditions, though. For example, we cannot measure biomarkers to assess no CSO risks in the Duwamish River and Elliott Bay. This lack is related to our inability to predict biomarker concentrations from the EFDC hydrodynamic model used to predict the no CSO conditions for this project.

Before biomarkers are more readily incorporated into risk assessments, it will be necessary to demonstrate that they have met an objective set of criteria that have been established for their use (Huggett et al. 1992). Examples from Huggett et al. (1992) of these criteria are:

- *Relative Sensitivity*—How sensitive is the biomarker compared to conventional endpoints such as lethality, reproduction, or growth impairment?
- *Biological Specificity*—How unique or general is the measured response to a chemical by different groups of biological organism?
- *Clarity of Interpretation*—How clear-cut is the endpoint as an indicator of exposure or effects of anthropogenic stress?
- *Permanence of Response*—Is the biomarker a transient or a permanent manifestation of exposure and/or effect?
- *Linkage to Higher Level Effects*—Can the biomarker be linked to effects at higher levels of organization (e.g., growth or reproduction)?

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16 Abiotic means non-living (without life).

Of these criteria, the most critical one, in our opinion, is the need for linkages with higher level effects. In current risk assessment practice, adverse effects to reproduction, development and survival are typically the endpoints used for assessing risk to aquatic life and wildlife. These endpoints are used because they can have a direct impact on a species population. Other endpoints, such as systemic effects (e.g., kidney lesions) and reduced growth (such 5 to 10 percent), generally are not considered because these effects are specific to an individual and cannot be linked to population effects. These effects typically do not keep a species from reproducing, developing normally and maintaining a sustainable population.

The ecological risk assessment will be primarily conducted using the current practices identified above. However, we have a unique opportunity to employ a biomarker database developed by the Northwest Regional Fisheries Center for flatfish exposed to PAHs in Puget Sound (as reviewed in Meyers et al. 1990). This research program has established that various flat fish (particularly English sole) develop a biomarker, termed Fluorescent Aromatic Compounds (FACs) in bile after exposure to sediment PAHs. FACs are metabolic by-products from the degradation of PAHs in flatfish. Furthermore, exposure to PAHs has been linked with development of liver lesions in this fish. Research has established that liver lesions form only after a threshold concentration has been exceeded. We will use the regression relationships developed by NRFC to predict FAC concentrations in English sole, which will then be used to further predict the probability of liver lesions in this fish.

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Issue Paper 7: HUMAN HEALTH TOXICOLOGY

To assess risks to human health from chemicals in the Duwamish River and Elliott Bay, the Water Quality Assessment (WQA) requires two types of information:

- The amount of chemical that people are exposed to when engaged in different activities (exposure)
- The amount of chemical taken into the human body that is considered to be either “safe”, or associated with a defined level (generally small) of health risk

This paper discusses the assumptions and uncertainties of the information used to establish “safe” doses as defined using accepted principles of toxicology (Doull et al., 1980). These principles are:

- There must be a chemical or physical agent capable of producing a response (a “chemical”).
- There must be a biological system with which the chemical interacts to produce a response (an “exposure”).
- There must be a response that can be considered to be deleterious to the biological system (an “effect”).

Establishing toxicity values involves two basic steps: hazard identification and dose-response assessment. In hazard identification, the toxicology data for a chemical is reviewed to determine whether exposure to a chemical can cause an adverse health effect (e.g., cancer, reproductive toxicity) and whether these health effects are likely to occur in humans. Hazard identification is largely a qualitative evaluation. The second step in establishing toxicity values is the dose-response assessment. The dose-response assessment quantitatively expresses the relationship between the amount of a chemical taken into an organism (the dose) and the incidence of adverse health effects (the response) in the exposed population of animals and humans. The hazard identification and dose-response assessment steps, as with many aspects of risk assessment, involve assumptions which this paper broadly identifies and discusses.

The general issues related to the development of the human health toxicity values used in this project are:

- *Toxicity values are derived differently for certain types of human health effects.*
- *Toxicity values are based largely on animal rather than human toxicology studies.*

- *Health-protective assumptions are incorporated in the derivation of human toxicity values.*
- *Chemical toxicity can be affected by where chemicals enter the human body.*
- *Limited data are available on how mixtures of environmental chemicals affect toxicity.*

The general methods, assumptions, and uncertainties embodied in the human health toxicological assessment for the Duwamish River/Elliott Bay WQA are discussed below for each of these issues. Cited references provide additional detailed information for interested readers.

7.1 Toxicity Values are Derived Differently for Certain Types of Health Effects

The U.S. EPA has developed specific methodologies for establishing numerical toxicity values for chemicals causing cancer or other types of toxic effects (non-cancerous) in animals and humans (U.S. EPA 1986a,b;1987; 1991; 1995). These U.S. EPA values were developed for use in human health risk assessments such as the WQA, and were derived using a number of general assumptions. Internationally, the World Health Organization (WHO) has also developed procedures (WHO 1996) and these procedures largely parallel those of the U.S. EPA, with the exception of the procedures for some cancer-causing chemicals assumed by the WHO to have toxicity thresholds.

Methods used to develop toxicity values by the U.S. EPA and other organizations such as the WHO are discussed below for non-cancer and cancer health effects, as are the impacts on the risk assessment of using the different approaches.

7.1.1 Non-cancer Health Effects

Non-cancerous health effects include a wide range of endpoints such as organ-, tissue-, or system-specific toxicity, reproductive toxicity, developmental toxicity, and endocrine disruption. The toxicity values developed by the U.S. EPA for “critical” health effects, and used in the WQA, are based on health effects other than endocrine disruption. This is because the available scientific data suggest that endocrine disruption is not a toxic endpoint *per se*, but rather a mechanism of toxic action leading to health effects such as the development of cancer, or reproductive and developmental effects (U.S. EPA 1997) for which toxicity values have been traditionally established.

Examples of Non-cancer Health Effects. As part of the development of non-cancer toxicity values, the general toxicity data is reviewed to determine a chemical’s potential for causing health effects in animals and humans. Many types of toxicity data are examined and from this review a “critical” adverse health effect is identified. Critical health effects can include a wide-range of lesions, functional impairments, or impacts to the organism as a whole. Examples of organs, tissues and systems that can be affected include (but are certainly not limited to): skin, liver, lung, kidney, heart, brain, thyroid, gastrointestinal system, endocrine system, blood-forming system, and central nervous system (Doull et al., 1980). Examples of toxic effects include cellular effects, reductions in weight (organ-specific or organism bodyweight), reduced neurological function, reduced enzyme activity and gastrointestinal effects. Most toxicity values developed by the U.S. EPA and used in the Duwamish River and Elliott Bay WQA are based on these types of general toxic effects. A key assumption in the development of toxicity values for non-cancer effects by the U.S. EPA is that by protecting against the critical toxic effect for a chemical, other toxic effects that could be exhibited at higher exposures are similarly protected.

Reproductive toxicity can include (but is not limited to) effects on the following: fertility, impotence, menstruation, pregnancy term, birth weight, premature, and reproductive senescence (U.S. EPA 1996a). Developmental health effects focus on impacts to survival, fetal abnormalities, alterations in growth and reduction in the physiological competence of an organism. Though a number of chemicals have available toxicity data on these types of effects, other types of critical health effects often occur at lower doses. Therefore, fewer U.S. EPA non-cancer toxicity values are based on reproductive effects.

Endocrine disruption, though not an effect *per se* as mentioned above, refers to impacts by chemicals that mimic natural hormones. Endocrine disrupting chemicals can inhibit hormonal function or induce alterations in the function of the immune, nervous, and endocrine systems (U.S. EPA 1997). Some types of health effects of endocrine disrupting chemicals observed in animals include abnormal thyroid function, decreased fertility, defeminization and demasculinization and reduced offspring survival. In general, few chemicals are tested for endocrine disrupting effects, though the U.S. EPA's Endocrine Disrupter Screening and Testing Advisory Committee is currently finalizing a toxicology testing program for these types of effects (Fairly, 1998).

Non-cancer Toxicity Values For Different Exposure Routes. Toxicity values developed by the U.S. EPA for the health effects (except endocrine disruption) discussed above are referred to as reference doses (RfDs) for ingestion exposures and reference concentrations (RfCs) for the inhalation exposure route. These U.S. EPA toxicity values will be used in the Duwamish River and Elliott Bay WQA. RfD and RfC values are defined as estimates of a daily exposure level for the human population, including sensitive subpopulations, that are likely to be without an appreciable risk of deleterious effects during a lifetime (U.S. EPA 1989). Based on the Problem Formulation prepared for the Duwamish River and Elliott Bay WQA, inhalation exposures are not being evaluated in the human health risk assessment and is not discussed further in this issue paper.

Toxicity values are not currently available for assessing toxicity from the dermal exposure route, which is an important exposure route for the Duwamish River and Elliott Bay WQA. However, as discussed later in this paper (see Chemical Toxicity Affected

By Where Chemicals Enter the Body), we will use toxicity values derived from ingestion exposures to assess risks from dermal exposure.

Chemicals Lacking U.S. EPA Non-cancer Toxicity Values. For cases where U.S. EPA ingestion RfDs are not available for WQA chemicals (few instances) other toxicity values can be considered, such as the WHO Tolerable Daily Intakes (TDI). The TDI is defined identically to the U.S. EPA's RfD and RfC values, and is similar in the underlying methods of its development (WHO, 1996). In general, the TDI values developed by the WHO are comparable to the RfD values developed by the U.S. EPA.

Toxicity as a Function of Length of Exposure. Several types of RfDs are available for health risk assessment depending on the exposure route (ingestion, inhalation), the critical health effect (organ toxicity, etc.), and the length of exposure (long-term, intermediate, short-term) being evaluated. As a result of Problem Formulation, the WQA is focused on long-term or *chronic* exposures to sediments, water, and seafood in the Duwamish River/Elliott Bay so chronic RfDs are used . In using chronic exposure scenarios in the WQA, a key assumption is that we are not likely to underestimate health effects for individuals or groups exposed for shorter periods of time.

Deriving Non-cancer Toxicity Values. For most non-cancer health effects, protective mechanisms in the human body are believed to exist that must be "overcome" before an adverse effect is manifested (Rodericks 1992; U.S. EPA 1986a, 1989). These protective mechanisms include (but are not limited to) the capacity of an organism to detoxify chemicals and cellular repair mechanisms. For example, if a large number of cells of the body perform the same or similar function, the population of these cells may have to be significantly depleted through the toxic action of a chemical before an adverse effect is actually observed. As a result, a range of exposures to a chemical from zero to some finite value can be tolerated with essentially no chance of expression of an adverse effect. The upper end of this range, where an adverse effect would theoretically occur, is referred to as a "threshold".

The approach used by the U.S. EPA for developing RfD values (also TDI values) assumes that a threshold (tolerance range) for toxic effects exists. To develop an RfD, the upper end of the tolerance range is identified; in other words, an RfD is the highest dose that is not associated with health effects (adverse health effects, or non-adverse health effects). In general, once a key study has been identified for developing the RfD, the highest dose tested that is without adverse effects (including the critical health effect) is the key piece of data taken from the study and used to establish the RfD. This dose is called the No-Observed-Adverse-Effect Level (NOAEL).

As discussed later in this paper (see Toxicity Data Based on Animal Studies; Health-Protective Assumptions), most data for establishing RfDs are taken from animal studies and a series of "adjustments" to the NOAEL dose. The adjustments, or the dose selected from the critical study if a NOAEL is not available, are made to ensure the protectiveness of the toxicity value derived from the selected study. The effects of these adjustments are to make the resulting toxicity value more protective, and thus a higher chance that the toxicity of a chemical to humans is overestimated.

7.1.2 Cancer Health Effects

Cancer is most simply defined as unregulated (i.e., by the body's normal mechanisms) cellular growth, or the development of tumors. The development of cancer has, until very recently, been considered to be a process for which the assumption of a "threshold" does not apply. In other words, unlike chemicals causing non-cancer health effects, any dose of a cancer-causing substance, no matter how small, may result in changes in a single cell that could lead to uncontrolled cellular growth and eventually a clinical state of cancer (Wilson 1996; U.S. EPA 1986a, 1989).

How Chemicals Cause Cancer Varies and Can Affect Predicted Cancer Risks Our understanding of the biological processes related to birth and growth of cancer has greatly expanded since the adoption of the "no threshold" policy decades ago. For example, it is now understood that two different biological processes affected by chemicals can increase cancer risk:

- Increasing the rate at which cells divide
- Increasing the rate at which mutations occur, independently of cell division (Wilson 1996)

Because it is known that cell division is normally under close physiological control and kept within certain limits (Wilson 1996), scientists are now recognizing that a threshold likely exists for chemicals that induce cancer through non-mutagenic mechanisms. Carcinogenic chemicals that induce cancer through non-mutagenic mechanisms---in other words inducing cancer in a way that does not involve chemical action on genetic material---are referred to as non-genotoxic carcinogens (Wilson 1996; WHO 1996).

The toxicity values for cancer-causing chemicals ("slope factors", defined below), as currently developed by the U.S. EPA, do not generally consider possible threshold mechanisms for non-genotoxic chemicals, though this is likely to change[17]. Some of the carcinogenic chemicals evaluated in the WQA, such as arsenic, are thought to be non-genotoxic. The result of not considering the possibility of a threshold to the Duwamish River and Elliott Bay WQA is a potential to overestimate the cancer risk to people from exposure to cancer-causing chemicals in WQA site media (fish, sediment, water). King County has however, elected to be conservative in their evaluation of potential health risks in the WQA, and thus, U.S. EPA slope factors are used in evaluating human cancer risks.

Slope Factors Are Developed for Chemicals Causing Cancer in Humans or Animals. Under current EPA protocols, to determine if a chemical is a human carcinogen both human and animal studies in the scientific literature are carefully reviewed leading to assignment of a "weight-of-evidence classification" (U.S. EPA 1989; WHO 1996; IARC

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[17] In 1996 the U.S. EPA, recognizing the need to update the science underlying the current cancer assessment process, proposed revisions to their 1986 Cancer Assessment Guidelines (U.S. EPA 1996b). These changes include examining the mode of action of some carcinogens that could lead to the establishment of threshold-based toxicity values.

1992). The U.S. EPA has defined several weight-of-evidence classifications, which are defined in Table 8-1 below; the International Agency for Research on Cancer (IARC 1992) has also identified similar procedures. Following the assignment of a weight-of-evidence classification, toxicity values are developed for known or probable carcinogenic chemicals (A, B1, or B2 only). A number of the chemicals being evaluated in the Duwamish River and Elliott Bay WQA are either known (arsenic) or probable (benzo(a)pyrene, PCBs) human carcinogens.

Table 8-1. U.S. EPA Weight-of-Evidence Classifications for Carcinogenic Substances

Group	Description
A	Human carcinogen
B1	Probable human carcinogen; limited human data
B2	Probable human carcinogen; sufficient animal evidence, limited human evidence
C	Possible human carcinogen
D	Not classifiable as to human carcinogenicity
E	Evidence of noncarcinogenicity in humans

The toxicity values developed for known or probable human carcinogens are called "slope factors". A slope factor is defined as a plausible upper-bound estimate of the probability of a response per unit of chemical intake over a lifetime. Because the toxicity values for carcinogenic chemicals are expressed as a "probability of response" rather than a dose (as is the case for non-cancer causing chemicals), the resulting risk estimates are also expressed as a probability of contracting cancer.

Slope Factors are Based on Extrapolation From High Doses To Low Doses Using Mathematical Models. Health risks are difficult to measure in human or animal studies at the very low exposure levels typical of environmental risk assessments. Therefore, the development of slope factors using current EPA protocols depends on the use of mathematical models to extrapolate from the high doses typical of laboratory animal studies (or from epidemiological studies with humans), to the lower exposure levels expected in the environment. Several models have been evaluated by the U.S. EPA for generating slope factors, each with strengths and weaknesses. The use of these models introduces an additional source of uncertainty (biased toward health protectiveness in general) in the slope factors used to estimate cancer risk potential.

The details of estimating slope factors are discussed in greater detail in several U.S. EPA publications (U.S. EPA 1986a, 1989, 1996b) for the reader interested in more detailed information. Briefly though, the choice of a low-dose extrapolation model is governed by consistency with current understandings of the mechanism of carcinogenesis and not just the fit of the model to the tumor data. Typically, in the face of limited data sets for chemical carcinogens, a model called the linearized multistage model is used by the EPA, which tends to result in the prediction of more conservative slope factors than other types of mathematical models.

Extrapolation from Low Dose Exposures Assumes Linearity in the Dose-Response Curve. After an appropriate study(s) has been identified from the literature and the appropriate mathematical model has been selected, a dose-response curve is generated by the model for low dose exposures and the slope of the resulting dose-response curve is calculated and represents the "slope factor". Chemicals with steep dose-response curves have higher slope factors and thus are considered more potent in their ability to potentially induce cancer at low environmental doses. The slope factor is derived from the linear portion of the dose-response curve predicted by the selected mathematical model. Linearity of dose-response curves is thought to occur at only the lowest environmental exposures for most chemicals. The assumed linearity of the dose-response curve in the low-dose region is another source of uncertainty in the slope factors used to estimate human cancer risk potential because it assumes that cancer risk increases linearly with exposure (see Conservatism's in Human Toxicity Values for further discussion). Thus at high exposures, an assumption of linearity may not be valid for some chemicals.

7.2 Human Health Toxicity Values are Frequently Derived from Animal Toxicity Studies

As discussed above, human health toxicity values used in the WQA are derived differently for different types of health effects (cancer, non-cancer). The toxicity values are based on scientific studies (discussed above) that may be based on either human or animal toxicity studies. The number, quality, and types of toxicity studies available in the scientific literature for a chemical influence the selection of the critical study used as a basis for the toxicity value.

7.2.1 Different Types of Toxicity Data Are Available in the Scientific Literature

Studies from the scientific literature used to develop the toxicity values are either based on human studies or laboratory studies using different animal species. Studies of human toxicity are derived from either case reports of clinical toxicity, usually accidental poisonings reported by physicians, or epidemiological studies. Case reports are not based on controlled scientific investigations and therefore it is often difficult to identify the specific dose of the chemical that induced toxicity. Thus, fewer toxicity values developed by the U.S. EPA are based on clinical case reports.

Epidemiological studies are a far more important source of human toxicity data. The epidemiologist tries to learn how specific diseases are distributed in various populations of individuals. For example, epidemiological studies may focus on whether certain groups of individuals experiencing common exposures, such as workers engaged in a common activity or people taking the same medicine, experience unusual rates of certain diseases. Though these are valuable sources of information directly relevant to human exposure, in some cases these studies may not be carefully controlled and thus when used as a basis for toxicity studies, they can introduce uncertainties into the dose-response assessment. An example would be the epidemiological study underlying the arsenic toxicity value. Arsenic is a COPC for the Duwamish estuary WQA. The study used as a basis for the cancer slope factor represents a cross-sectional study of 40,000 Taiwanese exposed to arsenic in drinking water that found significant excess skin cancer prevalence by comparison to 7500 residents of Taiwan and Matsu who consumed relatively arsenic-free water (Tseng et al. 1968, Tseng 1977). Although this study demonstrated an association between arsenic exposure and development of skin cancer, it has been shown to have several weaknesses and uncertainties, including poor nutritional status of the exposed populations, possible genetic susceptibility and other exposures to inorganic arsenic from non-water sources that are not accounted for. Thus, in human studies there are also important differences that can affect predicted toxicity including variability in genetic constitution, diet, occupational and home environment, activity patterns, and other cultural factors.

Laboratory studies are usually carefully controlled scientific studies using animals (mice and rats most frequently) exposed to chemicals for short or long periods of time. The doses in these studies are frequently higher than would be expected in the environment, and thus the extrapolation of these high doses to typically lower human environmental exposures is a source of uncertainty. There are also important species differences in uptake, metabolism, and species and strain differences in susceptibility to certain toxic effects that can also introduce uncertainties into the derivation of toxicity values from lab animal studies. Where data are available, these differences can be incorporated into the derivation of toxicity values.

7.2.2 Human Toxicity Data are Used Preferentially When Available

For obvious reasons it is preferable to use human data instead of animal data in developing toxicity values for a given chemical. Many toxicity values developed by the U.S. EPA are based on human toxicity data and these data are largely derived from epidemiological studies rather than clinical case studies. As discussed above, arsenic is an example of a WQA chemical that is based on human epidemiological data. Often however, toxicity values used to evaluate toxicity to humans are based on studies of toxicity in animal species, out of necessity, thus introducing a significant source of uncertainty. Where animal toxicity data are the only data available, preference is given to species that are more closely related to humans where possible. Toxicity values for chemicals evaluated in the WQA represent a mix of animal and human-based values, though most toxicity values are based on animal data.

7.2.3 Impacts of Using Animal Versus Human Data on Toxicity Values

The impact of using animal data for predicting toxicity in humans (i.e., is the toxicity value too high or too low for humans) is not well understood for most chemicals, though some suggest that the tendency is to overpredict the potential for cancer and non-cancer health effects (Milloy 1995). To compensate for the lack of non-cancer toxicity data for humans, or in some cases the lack of good animal data, the toxicity value is typically adjusted to incorporate a margin of safety for human health protection. These conservative adjustments involve the use of uncertainty factors, which are used to further lower the toxicity value derived from the critical study (discussed earlier). In using animal data for developing cancer slope factors conservative assumptions are also employed, as are other adjustments based on physiological differences. Further discussion on the uncertainties and conservatisms associated with animal to human extrapolations are discussed in the following section (Health Protective Assumptions).

7.3 Health Protective Assumptions are Incorporated in Human Toxicity Values

As discussed previously, data for developing toxicity values may be limited and often the available data is for animal species rather than humans. Given these limitations, a number of conservatisms are built into the development of RfDs and slope factors that should be illuminated because their impact on predicted toxicity (and hence risk) can be significant.

7.3.1 Conservatisms of Non-cancer Toxicity Values

As mentioned previously, RfDs frequently include in their derivation adjustments, often large, termed uncertainty factors, as well as an implicit assumption that humans are *at least as sensitive* as the most sensitive animal species tested for that chemical. In fact, where concern for sensitive human subpopulations is an issue for a chemical (this is almost universally the case), then the assumption is that these groups are *more sensitive* than the most sensitive species tested.

Depending on the strength of the underlying toxicological database for a chemical, and concerns for sensitive members of the general human population, one or more uncertainty factors can be used to adjust the NOAEL (or other dose) identified from the critical study discussed previously. These uncertainty factors, UF (similar to safety factors for wildlife and aquatic life), are applied in factors of 10 as shown below:

- UF of 10 to account for variability in the general population, including sensitive subpopulations (elderly, children)
- UF of 10 to address extrapolation of animal toxicity data to humans

- UF of 10 when a NOAEL for critical effect is derived from a study with subchronic (intermediate exposure) rather than a chronic (long-term) exposure

- UF of 10 when a dose associated with an adverse effect is used (i.e., instead of using a NOAEL associated with no adverse effect)

In addition, a modifying factor (up to 10) may be applied in addition to the UFs identified above. The modifying factor is used to address any additional uncertainties in the critical study and in the entire toxicity database. The default value for the MF is 1.

Given these uncertainties, it is possible for RfD values for non-cancer effects to include uncertainty factors of up to 10,000. Where data from animals are used, typically the minimum UF applied is 100; this assumes that (sensitive) humans are at least 100 times more sensitive than animals which may or may not be the case. The U.S. EPA recognizes these uncertainties in their definition of the chronic RfD: “an estimate (with uncertainty spanning perhaps an order of magnitude or greater) of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a lifetime”.

7.3.2 Conservatisms Built Into Cancer Slope Factors

Current protocols for developing slope factors (U.S. EPA 1986a; 1989) embody several key conservatisms. A number of these conservatisms are being addressed by the U.S. EPA in proposed revisions to Cancer Risk Assessment Guidelines (U.S. EPA 1996b). Some of the key conservatisms in the derivation of cancer slope factors include:

- A substance causing cancer in animals is assumed to cause cancer in humans

- No threshold dose exists for carcinogens

- Humans are assumed to be as susceptible to cancer as the most sensitive species tested

- Benign tumors are generally combined with malignant tumors unless compelling evidence suggests otherwise

- Animal doses are converted to equivalent human doses using body surface area rather than bodyweight

- At low exposures the dose-response curve is assumed to be linear

- Frequent use of the upper 95 percent confidence limit on the estimated slope factor as the toxicity value

Without the assumption that animal carcinogens equate to human carcinogens few substances would be considered human carcinogens (NTP 1994). The U.S. EPA recognizes that this assumption contributes a high level of uncertainty as well. An equally uncertain assumption is the assumed equality of sensitivities of humans to the most sensitive animal tested for a carcinogenic chemical. Public policy has, however, largely dictated that a conservative approach be taken to managing risks from chemical carcinogens. The effect of assuming equal sensitivity of humans to the most sensitive animal tested could overpredict risks for some chemicals (Milloy 1995).

The concept of a possible threshold for non-genotoxic carcinogens was discussed previously and suggests that for some chemicals, where a threshold does exist (arsenic for example), the current U.S. EPA slope factors overestimate the potential health risks from cancer. For example, recent research regarding inorganic arsenic suggests that cancer may only develop if the arsenic intake is high enough to saturate the enzymes responsible for detoxifying inorganic arsenic to methylated forms (Milloy 1995). Others have suggested that arsenic is an example of a chemical that may act primarily as a tumor progressor, i.e., a chemical that affects the progression stage of carcinogenesis (Barrett 1993) rather than as an initiator of carcinogenesis (genotoxic). The concept of thresholds for certain types of cancer-causing chemicals is currently being re-evaluated by the U.S. EPA in revisions to the Guidelines for Carcinogen Risk Assessment (U.S. EPA 1996b).

The number of tumors is a key factor in determining the potency of a cancer-causing chemical (U.S. EPA 1986a). By including benign tumors in addition to malignant tumors, the cancer risk may well be overpredicted, though the degree of potential overestimation is uncertain. An additional uncertainty in the derivation of the slope factor involves the approach used to convert the chemical dose in animal tumor studies to an equivalent human dose . Because the current procedures use body surface area rather than bodyweight as the basis for scaling the doses, the equivalent human dose that causes cancer is made smaller and the result is a higher cancer slope factor. Changes to the procedures for scaling are proposed in revised EPA cancer risk assessment guidelines (U.S. EPA 1996b).

The assumption that cancer potential increases linearly at low-dose exposures is also highly uncertain. For example, some well-studied genotoxic carcinogens do not exhibit a linear dose-response relationship (SAB 1997). The validity of this assumption has been addressed in the proposed revisions to U.S. EPA Cancer Risk Assessment Guidelines (U.S. EPA 1996b). The result of changes in the linearity assumption could mean either increases or decreases in the carcinogenic potency for cancer-causing chemicals.

Current practice in developing slope factors is to identify confidence limits around the estimated slope factor. In particular, the 95 percent upper confidence limit is taken as the actual estimate of the slope factor for many (not all) chemicals. By using the 95 percent confidence limit there is a 95 percent chance that the carcinogenic response would not be greater than predicted based on the experimental data and the selected extrapolation model (U.S. EPA 1989). An additional conservatism in using the 95 percent upper confidence limit as the slope factor arises when cancer risks are summed in situations where multiple cancer-causing chemicals are present. Because the slope factors generally represent the 95th percentiles of probability distributions, and 95th percentiles of probability distributions are not strictly additive (U.S. EPA 1989), the total cancer risk predicted becomes (artificially) higher as risks from a number of cancer-causing chemicals are summed.

7.4 Chemical Toxicity Is Affected by Where Chemicals Enter the Body

Previous discussions have touched on the variety of health effects that can be induced from chemical exposures and the toxicity values (RfDs, RfCs, slope factors.[18]) that are used to quantify health risks from different exposure routes. As indicated previously, the Problem Formulation for the Duwamish River and Elliott Bay WQA has not identified inhalation as a significant exposure pathway, thus this route of exposure is not further discussed.

7.4.1 Evaluation of Toxicity From Ingestion Exposure

Toxicity of chemicals can be markedly different depending on the exposure route of the chemical entering the body (for example, ingestion vs. dermal exposures). To reflect these differences, toxicity from different exposure routes evaluated in the WQA are evaluated with route-specific toxicity data. Available ingestion RfDs developed by the U.S. EPA are used to evaluate all exposure pathways where ingestion is the pathway of entry into the body. These pathways include the ingestion of seafood, sediment, or surface water.

7.4.2 Evaluation of Dermal Toxicity

Currently no toxicity values exist for exposures that occur through the skin (dermal exposure pathway). Chemicals in contact with skin can elicit toxic effects two different ways: either through direct action at the site of skin contact, or at other locations in the body (e.g., liver) following absorption through the skin. Risk assessments typically focus on the systemic toxicity[19] of chemicals rather than direct toxicity. This is because of a general lack of data for direct toxicity and because these types of effects tend to occur at higher environmental concentrations than systemic effects.

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18 Toxicity values for carcinogenic effects can also be expressed in terms of risk of contracting cancer per unit concentration of the chemical in the exposure medium. These measures, called unit risks, are calculated by dividing the slope factor by an adult bodyweight and multiplying by an adult inhalation rate. Unit risk values are how inhalation toxicity values for carcinogenic chemicals are typically reported by the U.S. EPA.

19 Systemic toxicity refers to toxicity that is exhibited at locations within the body other than where the absorption of chemical occurs (i.e., liver effects from absorption through the skin would be an example).

Given this lack of toxicity data, and the potential importance of this type of exposure pathway for the WQA, dermal exposures are frequently evaluated using the ingestion RfDs. Large uncertainties are associated with using ingestion RfDs for dermal exposures, because it is possible that the absorption efficiency of the test animal from ingestion exposures is not the same as that observed during dermal exposure (U.S. EPA 1989, 1992).

The U.S. EPA (1989, 1992) has recommended a conservative approach whereby the ingestion RfD is modified to account for possible differences in dermal absorption. This adjustment of the ingestion RfD results in the estimated dermal toxicity value being more conservative than the ingestion RfD. However, because of the uncertainties associated with modifying the ingestion RfD to be used for dermal exposure, this approach will not be used in this project.

The accuracy of the toxicity value extrapolation all contribute to uncertainties in the predicted risks from the dermal pathway. Unfortunately, few data are available to allow general comparisons to be made of the relative toxicity among exposure routes across chemicals that can provide a better understanding of the implications of ingestion to dermal extrapolations.

7.5 Limited Data are Available for Evaluating Environmental Mixtures of Chemicals

In general, few data are available that can be used to develop toxicity values for mixtures of environmental chemicals (U.S. EPA 1986c). The prediction of how specific mixtures of toxicants will interact once inside the human body should be based on an understanding of the mechanisms of such interactions.

7.5.1 Chemical Interactions of Mixtures

Although researchers use differing classification schemes when discussing the ways in which chemicals interact, it generally is recognized that chemical interactions can occur during any of the toxicological processes that take place with a single chemical: absorption, distribution in the body, excretion, and effect at the site (tissue, organ, etc.) of toxic action (Doull et al., 1980; U.S. EPA 1986c; Rodericks 1992).

The effect of two or more chemicals taken into the body simultaneously can produce a toxic response that is simply additive of their individual responses, or the toxic response may be greater or less than that expected by addition of their individual responses (Doull et al., 1980). Several terms have been developed to describe pharmacological and toxicological interactions of chemicals. Additive toxicity is a situation where the combined effect of multiple chemicals is equal to the sum of the effect of each individual chemical. The effect most commonly observed when two or more chemicals are taken together is an additive effect. A synergistic effect is a situation where the combined effect of two or more chemicals is much greater than the sum of the effect of each given individually. Antagonism is a situation where two or more chemicals together interfere with each other's actions such that the resulting toxicity is greatly reduced. The latter is desirable in toxicology and is the basis for many antidotes used in modern medicine (Doull et al 1980).

Toxicity values are available for very few mixtures of environmental chemicals. This is because mixtures can occur uniquely in the environment and resources are finite for toxicity testing. Given this limitation, the toxicity of chemical mixtures in the environment is often assessed using the toxicity data for individual chemical components of the mixture under the assumption of additive toxicity. This is expected to be the case for WQA chemicals as well. Rarely are the tools available to permit the evaluation of chemical toxicity from mixtures of chemicals using the other types of chemical interaction models.

7.5.2 Chemicals in Mixtures May Exert Toxicity Differently

The study of chemical interactions often leads to a better understanding of the mechanism by which chemicals exert their toxic actions. A knowledge of the mechanisms through which toxic effects are exerted and the target organ or systems affected by the chemical are important in characterizing health risks in situations where multiple chemicals are present. Current practice in risk assessment assumes that all types of cancer, regardless of location or type, are additive, which introduces uncertainty in combined risk estimates for mixtures of cancer-causing chemicals. This approach will also be taken for the evaluation of risk for exposures involving multiple carcinogens for the WQA.

Chemicals can exert non-cancer health effects on different parts of the body and through different mechanisms of action. Again, current practice in risk assessment is to sum non-cancer health risk estimates, though on the basis of "like toxic effect". By segregating non-cancer risks in this manner, it can be more accurately determined whether potential effects to a particular organ or system in the body may be manifested. This approach will also be used in evaluating mixtures of noncarcinogenic chemicals in the WQA.

7.6 Summary of Key Human Health Toxicology Issues for the WQA

- Toxicity values developed by the U.S. EPA are available for evaluating both cancer and non-cancer health effects for WQA chemicals. Other sources of toxicity values are also available for the one or two chemicals lacking data.

- A wide range of non-cancer health effects are represented by the toxicity values used in the WQA. These health effects are based on long-term (chronic) exposures, which are the focus of the WQA.

- Toxicity values for WQA chemicals are based largely on animal toxicity studies, though for some WQA chemicals the toxicity values are based on human epidemiological data.

- Uncertainty factors are incorporated in the toxicity values for non-cancer health effects to ensure adequate health protection, including sensitive subpopulations such as the elderly or children. The number of uncertainty factors varies by chemical and is based on the amount and types of available toxicity data (animal vs. human, intermediate vs. long-term exposure, adverse-effect dose or no-adverse-effect dose available etc.).

- Toxicity values for evaluating cancer health effects incorporate many conservatisms to ensure health protectiveness. Key conservatisms include the extrapolation of animal data to humans, choice of mathematical models, use of surface area to scale animal doses to humans, assumption of low-dose linearity and use of 95th percentile upper confidence limits to establish the toxicity value.

- Toxicity values for WQA chemicals are available for ingestion and inhalation exposure routes, though not for dermal exposures. Toxicity for dermal exposures is evaluated using ingestion toxicity values.

- Toxicity values are generally not available for mixtures of chemicals. Evaluating toxicity from mixtures of WQA chemicals will be based on additivity of non-cancer effects where chemicals are known to exert toxicity in the same manner or on the same organ or system.

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Issue Paper 8: INTERFACING THE DUWAMISH RIVER/ELLIOTT BAY MODEL WITH THE RISK ASSESSMENT

8.1 Introduction

Daily and seasonally changing conditions in the Duwamish River and Elliott Bay, along with the size and diversity of the study area, complicate the job of the Water Quality Assessment team. Ideally, we would be able to take enough samples of waters and sediments to know when, where and how severely water quality is affected by combined sewer overflows (CSOs) and other sources of water and sediment "constituents of potential concern" (COPCs).[20] These samples would provide both hydrodynamic data and water and sediment quality data. Unfortunately, four factors limit our ability to collect the desired samples:

1. The cost of analyzing samples limits the number of water and sediment quality data we can obtain.

2. To do the Water Quality Assessment, we need to know whether measured COPCs come from CSO or non-CSO sources. This is at best very difficult to ascertain from field samples alone.

3. There is a logistical problem of getting people and equipment to the right places at the right times to collect samples. CSO discharges can occur with little warning, any time of the day or night. Moreover, they occur when strains on the sewage system require attention from the people whom otherwise could collect samples. Also, they tend to occur at multiple locations, and affect locations throughout the study area more or less simultaneously.

4. We need to assess conditions that haven't actually occurred. Specifically, we need to determine what the hydrodynamics and water and sediment quality in the study area would be if there were no CSOs.

For all these reasons, the WQA requires a model the Duwamish River and Elliott Bay. The model gives a detailed, mathematical description of the movement of water and particles in the study area. It also describes the fate of chemicals in the water and sediments. The model is an excellent substitute for extensive data collection because it can be used in conjunction with specific field data to predict what will occur in the Duwamish River and Elliott Bay under many different conditions. Considered alone, the data that have been (or could be) collected are insufficient for purposes of

[20] The Duwamish River and Elliott Bay Water Quality Assessment is evaluating 24 COPCs. They are: arsenic, bis(2-ethylhexy)phthalate, cadmium, copper, 1,4-dichlorobenzene, fluoranthene, lead, fecal coliforms, 4-methylphenol, mercury, nickel, phenanthrene, tributyltin, total PCBs, zinc, pyrene, chrysene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)anthracene, dibenzo(a,h)anthracene, benzo(g,h,I)perylene, benzo(a)pyrene, and benzo(e)pyrene.

the WQA. However, they can be used to calibrate the model, i.e., to tune it so it reflects the flows and concentrations observed in the river and bay. The model has been calibrated in this manner. Model calibration is covered in Volume 1, Appendix B-1. The calibrated model provides the data needed for the Water Quality Assessment, because it allows us to fill the gaps in the field data with model forecasts.

8.2 Is the Model the Right Tool for the Job?

The Duwamish River and Elliott Bay model is built using detailed, well-established principles of environmental fate and transport. The model is calibrated with extensive field data. Therefore it is the best available tool for predicting flows and concentrations when and where field data are unavailable. Future data could be used to verify the model's predictions, refine its calibration, and/or test the validity of its underlying principles. Presently though, it is reasonable and appropriate to assume, based on current information and understanding, that the calibrated model's predictions are accurate. As such, the model provides the best obtainable water and sediment quality information for the Duwamish River and Elliott Bay. Therefore, we are using the model to help us assess the level to which aquatic life, wildlife and people who use the river and bay are exposed to "constituents of potential concern" (COPCs), both under baseline conditions[21] and under a hypothetical scenario with no CSO discharges.

8.3 Using the Model to Assess Exposure: Post-Processing the Raw Data

Using a model is different than using field data to assess exposures to COPCs. We have already mentioned three important differences. First, the model can be used to predict exposures under conditions that haven't actually occurred (for example, if there were no CSOs). Second, it can be used to help identify the sources of COPCs in the study area. Third, the model can be used to fill data gaps. The model represents the Duwamish River and Elliott Bay as 5,120 water and 512 sediment "grid cells," (512 locations at ten depths from the surface to the bottom plus one sediment layer (0 to 10 cm)). It predicts concentrations of each COPC in each of these grid cells every 15 minutes, for as long a period of time as needed (we are using the model to assess exposures over the course of a year). Given its high level of spatial and temporal resolution, the model gives us a more or less "complete" hydrodynamic and water and sediment quality data set for the Duwamish River and Elliott Bay, for the modeled time span.

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[21] "Baseline" is meant to represent existing conditions in the Duwamish Estuary. Baseline conditions are modeled using September 1996 – April 1997 and May – August 1981 rainfall and hydrograph data. These data represent a typical year in terms of annual CSO volume, and include large CSO events during both high and low river flow conditions.

The fact that the output data set from our model is relatively complete raises an important issue, which is that the model's output data are not all relevant to the exposure assessment. Further analysis of the data is needed to pull out the relevant exposure information. This further analysis is called "post-processing." A series of three examples will help explain the issue:

1. When? If aquatic organisms are exposed to a relatively low COPC concentration, followed by a sufficient period of exposure to a higher concentration, the exposure to the higher concentration will have greater bearing on their risk. The model, though, provides all the exposure data--both higher and lower concentrations--so post-processing is necessary to pull out the higher exposures. Extracting the higher exposure concentrations is the first step in the post-processing needed to use the water quality modeling results in the risk assessment.

2. How long? Assessing risk involves comparing field exposures to effect thresholds. Effect thresholds are found by experimentally exposing organisms (typically in a laboratory) to different levels of a COPC for a specified period of time, and observing whether a particular effect is occurring. The effect threshold is the experimental exposure level that separates occurrence and non-occurrence of the effect. Risk estimation involves comparing effect thresholds, derived using experimental exposures, to predictions of actual (field) exposures. To obtain accurate risk estimates, field and experimental exposures have to be defined the same way (or in practice, as similarly as possible). In other words, we have to "compare apples to apples." One thing we can do, to help ensure the consistency of field and experimental exposures, is to use the same exposure duration in both cases. Chronic (long-term) effect thresholds are based on experimental exposures of several days to a few weeks. Therefore, the field exposure that should be compared to a chronic effect threshold also should be averaged over days to weeks. Even acute (short-term) effect thresholds are based on exposures of hours to a few days. The Duwamish River and Elliott Bay model predicts concentrations on 15-minute intervals, so time averaging of the model's concentration data (which estimate field exposures) is necessary to assess acute and chronic risks. We averaged concentrations over one hour and four days (i.e., we took the average of each concentration with the next three concentrations to get the one-hour running average, and with the next 383 concentrations to get the four-day running average). *Time averaging* is the second step in the post-processing needed to use the water quality modeling results in the risk assessment.

3. Where? The previous two examples explain why post-processing is needed to deal with questions of *when* and *how long* exposures occur that have a bearing on risk. Another question that has to be addressed by post-processing the raw data is *where* exposures occur that have a bearing on risk. There is clearly a strong habitat component to the *where* question: we want to assess exposure where populations are using the Duwamish River and Elliott Bay. One thing we need to do, is describe how risk varies across locations that could be occupied by individuals of a receptor population. Therefore, we need to pull out the right locations, and combine their data to describe the spatial pattern of peak, time-averaged exposure concentrations. *Describing spatial variability* is the third step in the post-processing needed to use the water quality modeling results in the risk assessment.

In summary, post-processing is necessary because the model describes COPC concentrations without regard for when, how long and where the exposures determining the level of potential risk occur. These are the three questions we need to address by post-processing the output data set from our model. We also want to retain as much flexibility as possible for answering questions from Stakeholder Committee members: for example, about risks at a particular location. This is an important point. It means:

1. We need to post-process the raw model output data enough to cull the information that's potentially relevant to the questions about risks, that are laid out conceptually in the Duwamish River and Elliott Bay Water Quality Assessment Problem Formulation.

2. At the same time, we have to be careful not to post-process the model output data too far. We want to avoid boiling down the raw data to the point where they're very easy to use to answer a set of pre-determined questions about risks, but too refined to be useful for answering unanticipated questions, for example from the Stakeholder Committee or the WERF peer review panel.

We have two back-ups to the post-processed exposure data: (1) we will maintain the raw data for potential future use, and (2) the model could be used to generate additional data. However, neither of these back-ups can be evoked quickly to answer questions about risks.

What follows is a brief discussion of how and why we are post-processing the model output data to assess risks to aquatic life, wildlife, and people. Post-processing of water concentration predictions for the aquatic ecological risk assessment is the most complicated post-processing step, so we will start there.

8.3.1 Aquatic Life- Water Exposure

The first post-processing question we will address is the *how long* question: what should be the exposure durations for assessing acute and chronic risks? To answer this question we referred to the U.S. EPA's Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses (Stephen et al. 1985). The Guidelines are about setting national water quality criteria, not about assessing risks to aquatic life. Nonetheless, their advice about averaging periods provides a reasonable approach, which we have chosen to follow. The Guidelines recommend a one-hour averaging period for assessing risks from acute exposures, and a four-day averaging period for assessing risks from chronic exposures. Our model outputs COPC concentrations every 15 minutes, so we calculate running averages to get our one-hour and four-day average exposure concentrations.

The Guidelines give two reasons for using a one-hour averaging period for acute exposure. First, some substances, for example ammonia or low dissolved oxygen, are fast acting toxicants. For these chemicals exposure to acutely toxic concentrations for about one to three hours is sufficient to cause death to aquatic organisms. Second, even with substances that are not fact acting toxicants, organisms may suffer delayed effects from one-hour exposure to acutely toxic concentrations. Therefore, a one-hour averaging period provides an accurate-to-conservative (depending on the substance) exposure duration for comparing to acute toxicity reference values.

The Guidelines also give two reasons for using a four-day averaging period for chronic exposure. First, they argue that the averaging period should be shorter than the duration of chronic toxicity tests (20 to 30 days) because substantial fluctuations within the test period result in increased adverse effects. Second, they argue that the results of chronic toxicity tests are, at least in some cases, determined by a sensitive life stage occurring during the chronic test. They reason that a four-day averaging period is probably sufficiently short to prevent increased adverse effects on sensitive life stages.

The next post-processing question we will address is the where question. We are assessing risks to aquatic life in the Duwamish River and Elliott Bay as two separate areas. The Duwamish River portion of the aquatic ecological risk assessment is focusing on the reach from the Norfolk CSO (just upstream of the turning basin) to the downstream end of Harbor Island. Norfolk is the uppermost of the King County CSOs on the Duwamish River. The reach from Norfolk to the end of Harbor Island is estuarine throughout. While freshwater is present, its distribution throughout this area is transient. This is due to the tidal influence in this area, and the mixing of the river with marine waters. Thus, our overall conclusion is that the available habitat for use by the identified receptors in the Duwamish River is estuarine. This conclusion is supported by the presence of marine shellfish as far up the Duwamish River as the turning basin. The portion of the study area below the downstream end of Harbor Island is Elliott Bay, which of course is marine habitat as well.

We will assess risks in each of the two areas (the Duwamish River and Elliott Bay) as a whole. Our reasoning is that we are assessing risks to the aquatic ecological community, so we should evaluate risks on a community-level spatial scale. Thus, we are assessing risks to the Duwamish River aquatic ecological community, and the Elliott Bay aquatic ecological community. When we post-process the model output data, we will retain information about which model grid cell each data point came from. This will enable us to go back and look at the exposures and risks in particular locations of interest, e.g., locations of important habitat for juvenile salmon close to a CSO, during months juvenile salmon might be present. This issue is discussed at greater length below, in the section Critical Habitat Areas.

The last post-processing question we need to address is the when question. During the course of a one-year simulation, at any particular grid cell, the model will produce many peaks and valleys in COPC concentrations. We are analyzing risks associated with the maximum monthly peaks in each cell. The decision to focus on monthly peaks is based on the following considerations:

1. Assessing risks from sequential peaks that occur close together starts to lose meaning as the peaks become closer together in time. The ecosystem doesn’t have a chance to recover between events; so, at least as an approximation, the sequence of peaks is a single exposure event. We used the highest four-day average concentration within a month to estimate the chronic exposure concentration for that month.

2. The time required for the Duwamish River and Elliott Bay to recover from peak exposures (assuming the peak exposures are high enough to cause risks) is unknown. Some recovery probably would occur within a small number of tide cycles, full recovery only after several to many seasons.

3. Assessing risks on a monthly basis will allow us to consider information about seasonal changes in the ecosystem, for example, whether or not migratory species would be in the Duwamish River or Elliott Bay when exposure concentrations are high enough to pose risks. These seasonal issues are discussed in Issue Paper No. 2, Aquatic Life and Wildlife Site Use. For example, Table 1 of Issue Paper No. 2 gives approximate dates of migrations of salmon in the Green/Duwamish River.

4. Assessing risks to the aquatic ecological community on a monthly basis gives us a representative sampling of the seasonally varying rainfall and flow conditions in the Duwamish River and Elliott Bay watershed.

The Duwamish estuary stratifies into a "freshwater lens" on the surface, with a saline layer below. Marine and estuarine organisms will avoid the freshwater lens, for example by closing their shells, swimming away, or sinking to more saline waters below the fresh surface layer. By avoiding the freshwater, organisms will avoid exposure to any COPCs found therein. Conversely, for organisms that fail to avoid the freshwater, risk from osmoregulatory failure will override risk from exposure to COPCs. Therefore, we will only analyze risks from exposures to COPC concentrations that occur when salinity is greater than 5 parts per thousand, which represents a threshold between freshwater and saline conditions (Moyle and Cech 1988).

The tendency for estuarine organisms to avoid freshwater conditions may turn out to be a significant mitigating factor in the risk assessment. CSOs and other discharges to the Duwamish River and Elliott Bay are freshwater sources. COPCs discharged into the freshwater lens will be dispersed down-current, only gradually mixing into the saline layer, resulting in lower exposure concentrations for aquatic organisms. Thus, the model may demonstrate that the hydrodynamics of the Duwamish estuary disperses COPCs before aquatic organisms are exposed, thereby mitigating risks.

In summary, we have to perform a series of post-processing steps to use the water concentrations predicted by the model in the aquatic ecological risk assessment:

1. Compute one-hour (acute) and four-day (chronic) running average concentrations for each COPC, for each model grid cell from the Norfolk CSO to the downstream model boundary in Elliott Bay.

2. For each COPC in each grid cell, select the maximum one-hour (acute) and four-day (chronic) running average concentration, for each calendar month of the simulation, excluding times when the salinity in the grid cell was below 5 parts per thousand.

3. Separate the grid cells into two sets: those above the downstream end of Harbor Island (the Duwamish River set), and those below it (the Elliott Bay set).

4. Group the maximum concentrations by month (e.g., group all the maximum acute Duwamish River concentrations for January, February, etc.)

These post-processing steps will produce two data sets (acute and chronic maximum exposure concentrations, outside the freshwater lens) for each of our nineteen COPCs, for each month of the year, for each of the two aquatic ecological communities (the Duwamish River and Elliott Bay). Each data set will contain the peak concentration from each model grid cell. It will also retain the information about which grid cell each concentration came from, so we can go back and evaluate particular grid cells of interest.

8.3.2 People and Wildlife-Water Exposure

The post-processing for exposure for wildlife and people to COPCs in the waters of the Duwamish River and Elliott Bay is somewhat simpler than for aquatic life. Because we are assessing risks to wildlife and people from cumulative exposures, we can use average water concentrations.

As discussed previously, we don't want to boil away too much information in the post-processing, so we are not going to average away all the variability in the water concentration data. We will calculate the average water concentration in each grid cell, for each month of the year. We will then calculate monthly average water concentrations over groups of cells used by the various wildlife populations and people. We refer to these groups of cells as "patches." For example, we have a patch for the great blue heron, and patch for the bald eagle, a patch for people who windsurf on Elliott Bay, a patch for net fishermen on the Duwamish, etc. Table 1 provides a summary of the patches we have defined for the risk assessment.

Table 8-1. Patches defined for people and wildlife using the Duwamish River and Elliott Bay.

Receptor	Patch[a]
Net fishermen	All Duwamish River cells
Scuba divers	Cells adjacent to Myrtle Edwards and Seacrest Parks, all depths
Swimmers	Surface layer cells adjacent to Duwamish Park and Duwamish Head
Wind surfers	All surface layer cells, Elliott Bay
Bald eagle	All surface layer cells
Great blue heron	May – July: Shoreline surface layer cells within one mile of Kellogg Island
	August – April: The following shoreline surface layer cells- Duwamish River south of Harbor Island; West Seattle; Myrtle Edwards Park and north, except between the fishing pier and Pier 91;
River otter	All shoreline cells, all depths
Spotted sandpiper	All intertidal habitat areas (see Figure 1)

[a] *We will assume exposure occurs all months of the year unless otherwise noted.*

We also will calculate the annual average water concentrations for each of the above-mentioned patches. Great blue heron will tend to concentrate along the Duwamish River, within about a mile of Kellogg Island, during the fledging season (May through July), but use a wider habitat during the rest of the year, as described in Table 8-1. Water concentration averaging for the great blue heron will be conducted accordingly.

8.3.3 Aquatic Life- Sediment Exposure

We will compare bulk sediment concentrations to effects thresholds selected from the following hierarchy, described in Issue Paper No. 7, *Aquatic Life and Wildlife Toxicity*:

1. Washington State Sediment Management Standards (Title 173-204 WAC), or
2. Long et al. (1995), or
3. Ecotox Threshold (U.S. EPA 1996b), or
4. Application Equilibrium Partitioning (EqP) to Chronic Water Quality Criteria after applying the 20 safety factor

This will require minimal post-processing. We will use dry-weight sediment COPC concentrations averaged over the biologically active depth profile (0 to 10 cm). We will use an annual average sediment concentration, because the 0 to 10 cm depth profile is comprised of sediments deposited over a period of years, thereby damping seasonal variability in deposited concentrations.

8.3.4 People and Wildlife- Sediment Exposure

The post-processing of sediment data for people will be similar to the post-processing of water data for these receptors. We will calculate the monthly average sediment concentration for each grid cell in the patches defined in Table 8-1. We will then average across the grid cells in a single patch, and average the monthly average concentrations in each patch across the year (using seasonal patches for the great blue heron as described above).

8.4 Critical Habitat Areas

The issues of habitat loss and remaining habitat areas were discussed in Issue Paper No. 1, *Study Area Description* and Issue Paper No. 2, *Aquatic Life and Wildlife Site Use*. A map of current habitat was provided in Issue Paper No. 1, which we reproduce here as Figure 1. The study area description issue paper noted that 98 percent of approximately 1,270 acres of tidal marsh and 1,450 acres of flats and shallows, and all of about 1,250 acres of tidal wetland, have been eliminated from the Duwamish River. Most of the current habitat has been created or restored in recent years, including Kellogg Island and other small, intertidal areas of marsh and unvegetated marsh habitat. These areas have been the focus of habitat restoration efforts by the Muckelshoot Tribe, the Port of Seattle, and Coastal America, a federal program with participation by the U.S. Army Corps of Engineers, U.S. Fish and Wildlife Service, and the U.S. Environmental Protection Agency.

Because so little habitat remains, and in light of the effort being spent on habitat restoration, all existing habitat areas on the Duwamish River are obviously critical. Therefore, we will pay special attention to these areas in the risk assessment. These areas will be considered in a separate section of the risk assessment report, in which we will discuss:

1. Model predictions of sediment and water COPC concentrations at these locations;
2. Field data on water, sediment and prey item COPC concentrations at these locations;
3. Field data on the benthic communities at these locations;
4. Field data on and potential risks to wildlife at these locations;
5. Potential risks to juvenile salmon, at locations that provide habitat for juvenile salmon.

8.5 Summary

We have covered a variety of issues in this paper. First, we discussed why a model is necessary to perform the Water Quality Assessment. Second, we discussed whether the model is sufficient for the job. Third, we discussed how the model output will be post-processed to provide the information needed for assessing risks. Along the way we addressed specific issues about what information is needed for assessing risks to people, wildlife and the aquatic ecological communities of the Duwamish River and Elliott Bay. Finally, we discussed the extra steps we'll take to address risks at critical habitat areas. The key take-home messages are:

1. The Duwamish River/Elliott Bay water quality model is a state-of-the-art tool for predicting water and sediment quality for the Duwamish River and Elliott Bay. It is uniquely capable of providing the data needed for the Water Quality Assessment.
2. The model provides a comprehensive description of Duwamish River and Elliott Bay water and sediment quality. Not all this information is pertinent for assessing risks to people, wildlife, and aquatic life. We need to extract the information that is, by post-processing the model output.
3. We have to be careful not to post-process the output to the point where it's too refined to answer unanticipated questions.
4. We have two back-ups for the post-processed data. We will maintain the raw model output for potential future use, and the model could be used to generate additional output.
5. The Duwamish River and Elliott Bay aquatic ecological communities are made up of estuarine and marine organisms, whose survival depends on their ability to avoid freshwater. Therefore, we are assessing risks from exposures to COPC concentrations that occur when salinity is greater than 5 parts per thousand.
6. We are assessing risks to wildlife and people from cumulative exposures, so we are using spatially and temporally averaged water concentrations. We have the information we need to address uncertainties related to when and where exposures occur.
7. Habitat for wildlife and aquatic life on the Duwamish River is extremely limited. Therefore, we will pay special attention to those locations where habitat currently exists.

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Issue Paper 9: RISK PREDICTIONS AND AQUATIC COMMUNITY RESPONSES

9.1 Introduction

The work of the stakeholders will begin in earnest when you are presented with risk predictions for the aquatic life, wildlife, and people using the Duwamish Estuary. To help you in this work, this paper discusses three specific issues associated with this assessment of risks to the aquatic community. These issues are:

- How can predictions of risk to the aquatic community be interpreted?
- How quickly can aquatic communities recover after a disturbance?
- What are the consequences of the loss of species from a community?

Interpreting risk predictions requires a basic understanding of how these predictions are formed, and how they relate to the field data collected as part of this project. The first section of this paper summarizes the ongoing discussions concerning risk assessment methods that have taken place in the stakeholder working sessions and the limits of these techniques. The second section addresses an issue you may want consider in determining the benefits of reducing CSO discharges to the Duwamish Estuary. Specifically, what is the scientific evidence concerning the ability of aquatic communities to recover after exposure to a stressor. The third section of this paper summarizes the ongoing discussions and investigations within the scientific community over the consequences of species loss from a community. As you will find, this is an area of intensive debate with minimal amounts of data to distinguish between differing perspectives. The material in this section summarizes these positions to provide a range of views in our discussions of the risk assessment results.

9.2 Risk Predictions

9.2.1 Summary of Aquatic Life Risk Prediction Methods

The aquatic ecosystem will be evaluated for risks from chemicals in the waters and sediments of the Duwamish River and Elliott Bay and from changes in water quality parameters (DO, pH, salinity, temperature, and TSS) and physical impacts (scouring, sedimentation, and displacement). We will evaluate aquatic life chemical risks in two stages. First, we will compare chemical concentrations to screening thresholds (U.S. EPA water quality criteria), then we will perform a more detailed evaluation of each stressor that exceeds its criterion, along with the physical stressors. The screening thresholds will be designed to be protective of sensitive and commercially valuable species. Chemicals that never exceed their screening threshold concentrations during the course of the one-year simulation, at any location in the Duwamish River or Elliott Bay, will be considered not to pose risk to aquatic life in the water column. Chemicals that exceed screening levels at any location, at any time during the one-year simulation will be further evaluated in the risk assessment.

This evaluation will use the WERF Tier 3 methodology (WERF 1996). The WERF Tier 3 methodology is a process that calculates the percent of species whose acute or chronic toxicity thresholds will be exceeded by the exposure concentrations. To perform this analysis, it was necessary to make a series of assumptions and decisions, such as:

- For estimating acute exposure, we will use monthly maximum one-hour concentrations (Stephan et al. 1985) in each model grid element with salinity greater than five parts per thousand (‰). We will not estimate risks from chemical exposures in waters at less than 5‰ because at those salinities, estuarine organisms either have a mechanism to avoid water exposure, or the freshwater itself is the toxicant.

- To estimate chronic exposure, we will use monthly maximum four-day running average concentrations (Stephan et al. 1985) in each grid element with salinity greater than 5‰.

- We will use dissolved chemical concentrations derived from model output to estimate exposure concentrations in the water column.

The evaluation of risks to benthos will compare measured nearfield and model-predicted farfield sediment COPC concentrations to state sediment management standards. There are no state standards for sedimentation and scouring, so we derived criteria for these two stressors. The model-predicted sediment concentrations will be for the top ten centimeter layer, at the end of the one year baseline and without CSO simulations. In addition, we will conduct a benthic survey comparing a nearfield site at the Duwamish/Diagonal CSO and storm drain to a farfield site at Kellogg Island.

9.2.2 WERF Methodology

The Water Environment Research Foundation (WERF) methodology (Parkhurst *et al*. 1994) we will use to predict risk to the aquatic community involves constructing two types of cumulative distributions—an effects curve and an exposure curve. The effects curve represents the concentrations at which any particular percentage of species in the aquatic ecological community exhibits a particular effect (acute or chronic) in response to a toxicant concentration (see the example in Figure 2). To represent how this curve works, we have drawn in a line from the y-axis representing the point at which half the species are effected to the effects curve itself. The intersection of the line with the curve is then extended down to the x-axis to identify the concentration that acutely affects 50 percent of the exposed species (6,000 µg/L in this hypothetical example).

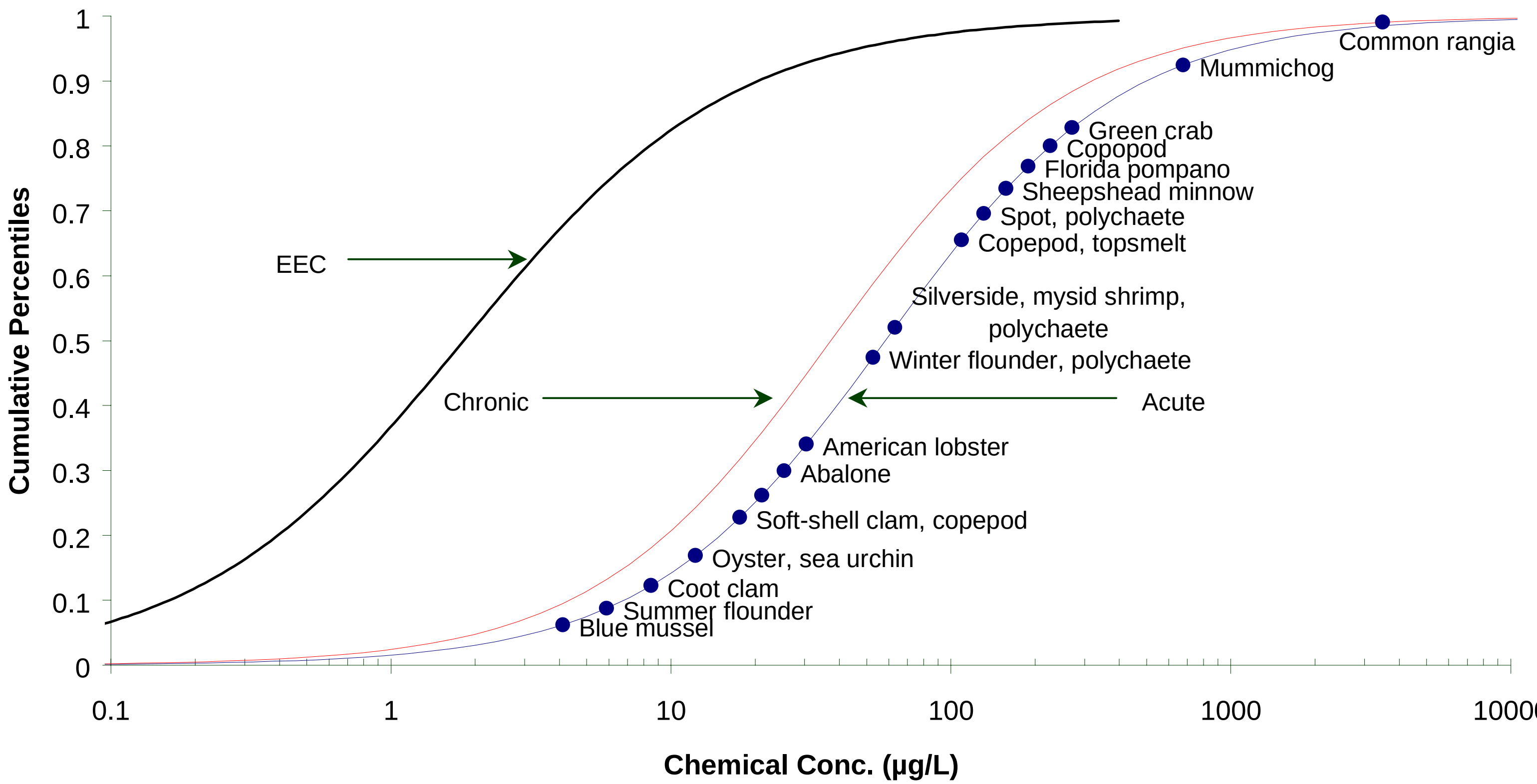

Figure 2.
Theoretical EEC and acute and chronic response curves for an individual chemical stressor

The exposure curve (termed the estimated exposure concentration (EEC) curve) presents the peak concentrations to which organisms are exposed across their ranges in the Duwamish Estuary. EECs will be developed for each receptor from the specific geographical areas and seasonal periods that represent the places and times they use in the study area. For example, only the peak concentrations from model cells in Elliott Bay will be used to estimate concentrations for marine species, such as rockfish. Using these two curves in combination will generate our risk predictions. This will be done mathematically, but can be visualized as extending the overlap of the maximum EEC with the effects curve down to the x-axis to identify the percent species at risk.

To interpret these risk predictions we need to consider:

- The type of effect used to make the prediction
- Predictions based on exposure to single versus multiple stressors
- Field measurements of species impacts

9.2.3 Endpoints

In this project, we have used the term endpoint in several different contexts, such as measurement and assessment endpoints. Here we specifically define an endpoint as the type of effect impacting a certain percentage of the individuals in a population. For example, a chronic endpoint could be a reduction in reproduction for 25 percent of the most sensitive individuals in a test population. The standard acute endpoint is the death of 50 percent of the exposed population. This definition of an endpoint is important because to interpret a risk prediction you must know the type of "endpoint" used to develop the effect curve used in making that prediction. The actual prediction we will make is that a specific endpoint will occur for some percentage of the exposed receptor species.

Let's postulate that a risk prediction is made for acute risk to 6 percent of the exposed aquatic community. In this example, LC50 values (the median lethal concentration or the concentration that kills 50 percent of the exposed individuals) were used to develop the effects curve. This risk prediction means that half of the exposed individuals that make up the populations of those 6 percent of the most sensitive species are predicted to die from exposure to that stressor. Stated alternatively, the EEC exceeds, at a minimum, the LC50 of these sensitive species.

Further examination of the effects curve will reveal that within the species predicted to be affected, not all will be affected equally. The EEC will be equal to the LC50 of the least sensitive species at risk, but could be several times the LC50 of the most sensitive species. All individuals in the population of the most sensitive species are likely to be at risk, while only half the individuals of the least sensitive species will be at risk. In such a situation, the most sensitive species is likely to be completely excluded by exposure to this EEC, whereas the least sensitive species (still within the most sensitive 6 percent) will likely survive, although half the individuals may not. Ultimately, this prediction would mean that somewhere between 0 and 6 percent of the species affected will be excluded from the community by this stressor.

Interpreting predictions of chronic risk is more difficult than interpreting predictions of acute risk, because we often have less data and more uncertainty about chronic endpoints. Examples of chronic endpoints are when 25 percent of the exposed individuals experience reduced reproduction effort or 15 percent of the exposed individuals grow significantly slower than non-affected individuals. Chronic risk predictions will indicate the percent of species exhibiting a detectable decrease in growth or reproductive effort. As illustrated above for acute risk predictions, a chronic risk prediction of 17 percent of the exposed species based on a 25 percent reduced reproduction endpoint would mean that at least 25 percent of the exposed individuals of the 17 percent most sensitive species would have significantly lower levels of reproduction than non-affected individuals.

To extrapolate from this type of endpoint and make predictions on whether species will be excluded from the study area would require the use of a concentration-population response model (e.g., Barnthouse et al. 1988; Suter et al. 1993). These models combine population dynamics and chronic toxicity information to get an overall picture of how the risk to individuals will affect the population. These models are difficult to build because the factors affecting population dynamics are difficult to quantify and because they may change as environmental conditions change. In other words, populations adapt to their conditions, and these adaptations are difficult to capture in a model. In fact, it is beyond the current state of the sciences of ecology and toxicology to quantitatively translate the effects of individual chronic risk predictions to populations or communities. Nonetheless, risks to individuals are likely to be conservative as indicators of risks to populations (because populations adapt to stressors), so the chronic risk predictions contain some (unquantified) margin of safety.

9.2.4 Single versus Multiple Stressors

An additional obstacle to interpreting risk predictions is that they are based on exposure to single stressors, while actual field exposures involve multiple stressors. The interactions among these stressors are complex and, in general, not understood well enough to allow us to combine a set of single stress predictions into one multiple stressor prediction of risk. For example, different stressors may affect the same species, or they may affect different species. The former would result in fewer species at risk than the latter, although the risk to those species could be higher.

If two stressors affect the same species, the net result is not necessarily additive. Other interactions that could occur are synergism (enhanced activity) and antagonism (depressed activity). The quantitative assessment of risks from multiple stressors is a topic currently being studied and debated in the scientific community. Both WERF and the Society of Environmental Toxicology and Chemistry sponsored expert workshops this past summer to address the assessment of multiple stressors. King County’s John Strand attended of the WERF workshop. He felt WERF was able to frame the questions we should be asking, but no one yet had answers about how we should address multiple stressors. At present, the best we can do with the predicted single-stressor risks is to apply considered professional judgement to qualitatively describe the overall risk.

9.2.5 Field Measurements of Species Impacts

Additional insight into the interactions of stressors being evaluated individually can be gained by comparing our predictions of risk to the field measurements of benthic organisms that have been made at the Duwamish/Diagonal storm water/CSO discharge pipe. Measurements of the benthic species number have been made in areas directly in the outfall footprint of this discharge and at similar sediment locations near Kellogg Island, a reference site in the Duwamish River. Using these data, we will be able to determine the species absent from the outfall area compared to the reference site, as well as changes in abundance in the remaining species. Species excluded from the footprint would be the result of all stressors, both chemical and physical at this location. Comparing our predictions for individual stressors in this nearfield environment to the field measurements will provide a benchmark against which to evaluate these predictions. For example, we may find that the amount of impact on populations is similar to what we predict from the one or two stressors posing the greatest risk, indicating that these stressors determine community structure at the site. In this case, risks to the community would be lower than the sum of the individual risks.

9.3 Recovery After Disturbance

In evaluating risks to the aquatic community, it is necessary to consider not only the frequency and type of risk posed but also how the aquatic community will respond to that risk. The responses can be divided into direct responses to stress (addressed above) and the recovery of the community once the stress ends. This is an important concept to consider in discussing the consequences of removing CSO discharges from the Duwamish Estuary. The initial scenario we will consider is the complete removal (zero-CSOs), but additional scenarios we may consider would involve changing the frequency of CSO discharges from more than once per year to only one discharge per year. The recovery of the aquatic community will depend, in part, on the frequency and timing of the disturbance, alteration of habitat quality associated with the disturbance, and the availability of colonizing populations to repopulate impacted habitats (which is a function of the abundance of these populations as well as their distance from the disturbance site) (Wallace 1990; Yount and Niemi 1990).

While aquatic life communities can change significantly when exposed to chemical and physical stressors, they generally are resilient and can achieve pre-impact characteristics in relatively short times. For example, in a review of 150 case studies, Niemi et al. (1990) found that most recovery times by fish, macroinvertebrates, and phytoplankton populations were less than 3 years following exposure to chemical and non-chemical stressors. Systems that did not recover in this time period had experienced either physical habitat alterations, residual pollutant concentrations, or were isolated and recolonization was suppressed (Niemi et al. 1990). For example, rocky shore communities impacted by an oil spill were still not completely recovered after 10 years, in part due to residual toxicity (Southward and Southward 1978). Similarly, fish species composition, species richness, and total density all recovered within one year for over 70 percent of the 49 systems studied (Detenbeck *et al*. 1992) when habitat quality was not affected by the original disturbance. When habitat quality was affected, recovery times ranged from >5 to >52 years (Detenbeck *et al*. 1992).

Response of epibenthic macroinvertebrates can be very rapid to some types of physical disturbances (on the order of days) (Tikkanen *et al*. 1994). Response by aquatic macroinvertebrates communities to decreased metal inputs from mine drainage was rapid (Nelson and Roline 1996), with impacted sites comparable to upstream reference sites in two years. Nelson and Roline (1996) suggest that aquatic communities impacted by metals, in the absence of degraded habitat and with nearby recruitment areas, will recover quickly if low instream concentrations of toxicants are achieved.

Recovery of macroinvertebrate community was monitored in a freshwater stream after three years of seasonal insecticide applications (Whiles and Wallace, 1992, 1995). While total abundance of this community was unchanged during these three years, biomass and production were decreased during treatment. Two years after treatment, biomass, productivity, and ecosystem function had returned to pre-treatment levels. While these community elements did recover, it is critical to note the certain species did not return to pre-disturbance levels. However, the function performed by these species in the ecosystem (shredding leaf litter into very small pieces that could be eaten by other animals and microbes) was filled by other species that expanded their abundance during the recovery period (Whiles and Wallace, 1992, 1995).

Finally, recovery of macroinvertebrate and fish communities can vary seasonally. Timing of the disturbance relative to the life history characteristics, particularly spawning, of the impacted community can be critical to the response time (Detenbeck et al. 1992; Tikkanen et al. 1994). Disturbances that occur outside of critical life stages (e.g., spawning) have a much more limited impact on species than they can otherwise (Wallace 1990). This is particularly true for macroinvertebrates who have evolved a number of life history characteristics to allow them to persist in areas with naturally existing disturbances (e.g., floods) (Wallace 1990). For example, even in areas with high flooding rates, densities and diversities of sediment insects returned to pre-flood levels during periods of low flow in the same year (Cobb *et al*. 1992).

9.3.1 Recovery in the Duwamish River and Elliott Bay

As discussed above, the critical element in an ecosytem recovering after a disturbance is the time between disturbances. Included in this the concept is whether or not the stressor itself is removed from the system shortly after the disturbance occurs. Ecosystems with persistent stressors, such as chemical contamination present in sediments, frequently take very long times to recover. In considering the Duwamish River and Elliott Bay, the periods between CSO discharges range over days to months. On average, the longest period between discharges is on the order of six months. This is unlikely to be adequate to observe any significant recovery.

9.4 Species Loss, Biodiversity And Ecosystem Function

The implications of both local and global species loss have been the focus of intense interest and discussion both within the scientific community as well as our society at large (Ehrlich and Ehrlich 1981; Grime 1997; McDonald 1997). Indicative of this was an article appearing in the New York Times as this paper was being written. This article states that one in every 8 plant species in the world is under threat of extinction (Stevens, 1998). In discussions of these issues, scientists investigating species loss and the consequences of extinction have put forth a broad spectrum of varying viewpoints. In this section, we summarize these positions to provide a range of perspectives for our discussions of the risk assessment results. In considering these perspectives, each of us needs to consider that "good science is necessary, but isn't sufficient" (Plummer 1998). Plummer (1998) goes on to say "all the scientific information in the world…cannot provide us with the answers to many of the hard choices ahead. Science provides information of the ecologically possible, but cannot indicate which among many possibilities is the best." To decide this, our society must also address economics and politics (Plummer 1998), as well as its values.

9.4.1 Perspectives on Species Loss

In discussing aquatic community risk predictions, it is critical to distinguish between the exclusion of species from a community and the total extinction of that species. Everything we have addressed so far has dealt only with the former. However, Ehrlich (1995) has noted that "populations are disappearing at a high rate, and that will have disastrous consequences…regardless of the fate of species. After all, if population extinctions reduced all remaining species to single minimum viable populations, no further species extinctions would have occurred. Nonetheless, an extinction catastrophe would have taken place…through the interruption of ecosystem services [function]." In the remainder of this section, we go beyond our ability to predict risks to single populations and address the question of what is the consequence of species loss.

One commonly stated position concerning species loss is that we must act to protect every species (Ehrlich 1990). Reasons advanced by Ehrlich (1990) for this position are ethics, aesthetics, economic consequences, and the basic functioning of ecosystems. Paul and Anne Ehrlich in 1981 (Baskin 1994) have summarized this last perspective as the "rivet-popping hypothesis". They liken species in an ecosystem to the rivets holding a plane together. Once enough rivets have popped-off, the plane crashes. As we don't know which "rivet" will be the critical one, they propose we should protect all species possible to prevent catastrophic breakdowns.

However, some scientists investigating the relationship between the number of species and the functioning of ecosystems have stated that not all species are equally important to the functioning of the ecosystem. Ecosystem functions are basic processes, such as the creation of new growth by plants and algae (primary productivity), the flow of nutrients such as nitrogen and phosphorus (nutrient cycling), and the decay of dead plants and animals so that their mineral components are available for use by new living organisms (decomposition rates). Each function is typically carried out by a number of species in each ecosystem, and their research has demonstrated that not all species contribute equally to these functions (Grime 1997). This research indicates that some threshold number of species are required for ecosystem function, but exceeding these thresholds adds little detectable benefit to these functions (Baskin 1994).

It now appears that ecosystem functions typically reflect the features of the few dominant species that perform them (Grime 1997). Additionally, many ecosystems have a number of species performing the same function, creating functional redundancy (Walker 1992, 1995; Baskin 1994). An example of redundancy was presented in the previous section concerning the change in composition of species shredding detritus in freshwater streams but continued function in the ecosystem (Whiles and Wallace, 1992, 1995). Thus, these scientists argue that loss of species making the major contribution to an ecosystem function, or performing functions with little or no redundancy, are those on which we should focus our protection efforts.

Another group of scientists has examined the geological record of extinctions over the last three billion years to better understand the consequences of current species losses. Two perspective have grown out of this examination. Paleontologists have found from examining the fossil record that of all the species that have lived on the Earth since life first appeared, only about one in a thousand are still living today (Newman and Roberts 1994). During this time period there have been at least five mass extinctions, resulting in between 15 percent and 50 percent of biological families[22] disappearing (Bagheera and R&E Online, Inc. 1996). During the Permian period (approximately 250 million years ago), it is estimated that more than 60 percent of all the earth's plant and animal species were lost (Columbia University Press 1994).

Additionally, paleontologists have observed that species diversity has recovered after each mass extinction with reappearance of species not recorded in the fossil record for some time (so-called Lazarus species) as well as the evolution of new forms not previously observed (Allmon and Morris 1995; Kaufman and Ermin 1995; Erwin 1998). Recovery periods to reach new levels of biodiversity have typically taken 1 to 2 million years (Kaufmann and Erwin 1995). Kaufman and Erwin (1995) have concluded that

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[22] A biological family is an organizational unit used to classify organisms and is composed of number of similar genera.

"global biodiversity crises leading to mass extinction are a normal part of Earth history, and have dramatically altered the course of evolution".

Other scientists have stated that the current pace of extinction is unlike that observed in earlier mass extinctions. Pimm (1995) proposes that "over the past few thousand years species have been becoming extinct at rates up to 1000 times greater than those deduced *from the fossil record.*" These rates of extinction are projected by some scientists to result in the possible elimination of between one quarter and half of all current species (Myers 1993). These projections are highly uncertain due to the very limited knowledge we have of the number of species on the earth—projected to be anywhere between 2 and 50 million species (Myers 1993; Pimm 1995). Myers summarizes the importance of these issues by stating that they are "central to our understanding of Earth's biota - and what we can still do to stem the biological debacle threatening to reduce the planetary stock of species by anywhere between one quarter and one half within the foreseeable future."

Examination of available data has called these projections of current mass extinction into question (Easterbrook 1995). For example, the Pacific Northwest is one of the best studied ecosystems in the world, where a number of environmental stresses have been well documented—logging, industrial activity, and residential growth resulting in fragmented forests, divided into checkerboards by roads and clearcuts. Yet, "there are no known cases of extinction of vertebrates in the Pacific Northwest forests during the postwar era" (Easterbrook 1995). Additionally, "the Nature Conservancy reports no known extinctions of vascular (loosely, green-stemmed) plants during the same period" (Easterbrook 1995). Easterbrook's perspective is not that we shouldn't protect species, but rather that we should and that our efforts are acting to improve our environment.

Regardless of which, if any, of the above perspectives are correct, it is probably best to end this section by quoting Paul Ehrlich and Edward Wilson as Easterbrook does in his book: "Because Homo sapiens is the dominant species on Earth, we and many others think that people have a moral responsibility to protect what are our only known living companions in the universe."

9.4.2 Evaluating Species Loss in the Duwamish River and Elliott Bay

Evaluating the consequences of species loss from the Duwamish River and Elliott Bay aquatic life community hinges on our knowledge of their ecosystem function. Loss of species which play key roles in maintaining community and ecosystem structure could result in permanent changes. Examples of these types of species would be primary producers, such as a dominant algae, or key predators that determine the distribution and abundance of their prey items. Frequently the impact of these "keystone" species is disproportionate to their numerical abundance Recent studies in the Northwest have documented this phenomena for two different predators, a dog-whelk and a seastar (Sanford, E. 1999, Berlow, E.L. 1999). Alternatively, species loss has resulted in the replacement by other species and the continuing functioning of this ecosystem. This has been documented in Willapa Bay, where an exotic aphid introduced from Japan has become the dominant food source used by the fish community. Based on our current knowledge of the aquatic life community in the study area, we are unable to predict the loss that would result from the loss of a particular species.

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