ECOANALYSTS, INC.

PUGET SOUND EXHAUST GAS CLEANING SYSTEM (EGCS) WASHWATER ECOLOGICAL RISK ASSESSMENT

Prepared for
The Cruise Line International Association
The Port of Seattle
The Washington State Department of Ecology

Prepared by
Jay Word - EcoAnalysts, Inc.
Cindy Wang - Exponent
Joy McGrath – McGrath Consulting
William Stubblefield - Oregon State University

Date
January 2024

Report ID:
PG1632.01

All results are intended to be considered in their entirety, and the authors are not responsible for use of less than the complete report.

PREPARED BY

Jay Word

Program Manager

Authors:

Jay Word, Dayang (Cindy) Wang, Joy McGrath, William Stubblefield

CONTENTS

TABLES

FIGURES

APPENDICES

GLOSSARY ACRONYMS AND ABBREVIATIONS

Ambient Site Water	Collected during sampling from the Seawater Inlet site.
CA	Concentration Addition
CCL	Carnival Cruise Lines
CI	Confidence Interval
COC	Chain-of-Custody
Control	Laboratory Control Water – Source Hood Canal
CRDI	Common Rail Direct Injection
CTLBB	Critical Target Lipid Body Burden
DeSOx	Desulfurization
EC	Effect Concentration
ECHA REACH	European Chemicals Agency Registration, Evaluation, Authorisation, and Restriction of Chemicals
EGCS	Exhaust Gas Cleaning System
GESAMP	Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection
HAL	Holland America Line
HFO	Heavy Fuel Oil
IA	Independent Action
IMO	International Maritime Organization
MAMPEC	Marine Antifoulant Model to Predict Environmental Concentrations
MARPOL	International Convention for the Prevention of Pollution from Ships
MCR	Maximum Continuous Rating
MGO	Marine Gas Oil
NPDES	National Pollutant Discharge Elimination System
OD	Overboard Discharge Sampling Point
OECD	Organisation for Economic Co-operation and Development
PAHs	Polycyclic Aromatic Hydrocarbons
PCL	Princess Cruise Line
PD	Post DeSOx Sampling Point
PEC	Predicted Environmental Concentration
PNEC	Predicted No-Effect Concentration

POS	Port of Seattle
SFI	Standard Fuel Injection
SI	Seawater Intake Sampling Point
TLM	Target Lipid Model
TU	Toxic Unit
VGP	Vessel General Permit
VLSFO	Very Low Sulfur Fuel Oil
WDOE	Washington Department of Ecology
WET	Whole Effluent Toxicity
U.S. EPA	United States Environmental Protection Agency

EXECUTIVE SUMMARY

The goal of this study was to evaluate potential environmental risks associated with the operation of open-loop EGCS, in the waters of Puget Sound in Washington State, in the U.S. A. Study design was developed in consultation with an Advisory Panel of experts, and also with the Port of Seattle and the Washington State Department of Ecology (WDOE) and the Cruise Line International Association (supported by Carnival Corporation). A combination of analytical determinations (PAHs and metals) in pre- and post-EGCS waters, modeling of overboard discharge toxicity, pre-discharge evaluation of environmental toxicity, and post-discharge evaluation of environmental concentrations were employed. Differences in the composition of the discharge waters attributed to engine load were included as an experimental variable. Possible differences attributable to fuel type were controlled by limiting the study vessels to a single HFO source with a sulfur content of 1.1 to 1.7%. Two engine loads were evaluated, low load (≤ 50%, represents "in port operations") and high load (≥70% represents normal transit operations). A representative set of samples were collected from six ships operated under low and high engine loads (total of 12 sampling scenarios). Environmental samples were collected from the seawater intake, post-EGCS waters, and overboard discharge for each scenario. Although the low engine load was meant to provide an assessment for what these discharges would be in port, this study collected the washwater samples while in transit from Victoria BC to Seattle WA.

Analytical determinations (PAH, metals, nitrates, nitrites, pH and additional constituents) were conducted on all seawater samples. Total metals were also measured in the fuel feeding the diesel generator set which was associated with the EGCS line during the sampling. The Target Lipid Model (TLM; DiToro et al 2000) was used to predict the toxicity of petroleum components (PAHs) in EGCS overboard discharged waters while evaluation of the toxicity of metal components was conducted in accordance with the evaluation of metals mixtures as described by methods published in Nys et al (2018). The Marine Antifoulant Model to Predict Environmental Concentrations (MAMPEC) Estuarine Harbour and Shipping Lane modules were used to model the Predicted Environmental Concentrations (PECs) of EGCS-associated water components discharged from vessels while at berth and during transit. Both modules used a simplified geometry of the receiving water, the ship visit pattern, the ambient water characteristics, the discharge configuration and characteristics, the sampled background concentrations, and properties of the various constituents of interest. In addition to PEC modeling efforts, short-term chronic whole effluent toxicity (WET) tests were performed for each overboard discharge sample to determine potential effects on growth, reproduction, and survival to the organisms in the receiving waters. Whole effluent toxicity tests were conducted with the Inland Silverside (*Menidia beryllina*), mysid shrimp, (*Americamysis bahia*), and sand dollar (*Dendraster excentricus*), following Washington Department of Ecology (WDOE, 2016) and U.S. EPA methodologies (U.S. EPA, 2002; U.S. EPA, 1995). The Silverside, mysid shrimp, and sand dollar bioassay methods are used by WDOE for monitoring National Pollutant Discharge Elimination System (NPDES) permitted discharges to the Puget Sound.

The marine diatom bioassay with, *Skeletonema costatum*, was added to the test design based on the advice of the external Advisory Panel to decrease the uncertainty of the risk the washwaters would pose to the marine community. Inclusion of the microalgae in the test design allows for an empirical measure of the response to a surrogate for this component of the community (i.e., primary producers).

Measurable concentrations of some metals (e.g., chromium, lead, nickel, and vanadium were noted in both DeSOx, samples taken immediately after the scrubber tower and prior to the addition of any

buffering water, and in the overboard discharge water, presumably originating from fuel combustion since they were also present in the uncombusted fuels. Chromium and lead concentrations were lower than the European predicted no-effect concentrations (PNEC), the U.S. EPA Ambient Water Quality Criteria and the Washington state standard. Measured nickel concentrations were slightly higher than the PNEC in the high engine load overboard samples, however the overboard discharge results for both engine load scenarios were lower than both the U.S. EPA Ambient Water Quality Criteria and the Washington state standard. Vanadium results were elevated above the PNEC and currently no U.S. EPA criterion or Washington state water quality standard exists for vanadium, so no additional comparison could be made. As concentrations for both nickel and vanadium slightly exceeded available PNEC; nickel and vanadium were further studied during the subsequent modeling activities.

Measurable concentrations of PAHs were noted in waters sampled following exhaust gas desulfurization (DeSOx) treatment and in the overboard discharge water when vessels were operated in either low or high load conditions. The Target Lipid Model (TLM) was used to predict toxic units based on PAH concentrations in the discharge samples prior to any dilution series. For this study, 60 individual PAHs were analyzed, and these results were used to calculate toxic units. The TLM predicted that with the exception of three of the twelve samples, the acute toxic units were above one suggesting that toxicological effects may occur from exposure to the concentrations of petroleum hydrocarbons in the overboard discharge. The acute and chronic toxicity calculated by the TLM were similar between engine loads for each ship; however, differences were predicted among ships. The TLM suggested that the biggest contribution to the overboard discharge water toxicity for PAHs was associated with C1-C4 phenanthrene/anthracenes concentrations.

In contrast to the TLM model predictions, limited toxicity was observed in empirical WET tests conducted with discharge water samples as discharged. Effects on organism survival were only observed in a single undiluted effluent sample. Effects on chronic endpoints indicated the greatest sensitivity was observed in the sand dollar larvae that developed normally and the growth (i.e., biomass) endpoint for the silverside test. Chronic effects were noted in 50% (based on the EC25 endpoint) of the discharge samples; however, these effects were generally ameliorated when the sample was diluted with clean seawater by 50 percent. These results suggest that chemistry-based toxicity predictions (e.g., TLM) may overstate cumulative risks when compared to observed results from laboratory-based empirical toxicity tests. Several reasons may exist for these discrepancies including: 1) uncertainty in characterizing the toxicity assessments used in model predictions, 2) environmental fate processes may decrease exposures under lab and field conditions thus reducing observed toxicity, and 3) possible interactions among effluent constituents may be over-estimated in the TLM model due to the assumption of response additivity.

A MAMPEC model (Marine Antifoulant Model to Predict Environmental Concentrations) was used to assess the fate and distribution of EGCS associated waters when discharged. MAMPEC has the ability to estimate hydrodynamic exchanges in six generalized “typical” environments two scenarios were evaluated in this study. The Shipping Lane Module was used to predict the fate of the constituents associated with the discharge and Predicted Environmental Concentrations (PEC) during transit while the Estuarine Harbor Module was used for assessing the fate of the constituents associated with the discharge and the PECs while in port. MAMPEC modeling results of simulated PECs in the discharge areas showed no significant increase from the background concentration for constituents which were identified over time, primarily due to local conditions and the high flushing rate of the constituents from the discharge area. The MAMPEC model produced estimates for dilutions of 205-fold (while in transit) and 40-fold (while in port).

The highest toxic units resulting from the TLM, and the WET tests were used to estimate *Risk Characterization Ratios* (RCR)[1] in the nearest cell to the discharge by dividing these values by the MAMPEC derived dilution estimates. The resulting RCRs were less than 1.0 suggesting minimal potential for ecological risk in the characterized area.

The Cornell Mixing Zone Model (CORMIX) was used to refine the estimates from the MAMPEC model for the in-port scenario. CORMIX model estimates were simulated under two discharge scenarios, discharge during slack tide and one-hour post slack tide. The CORMIX model estimated a dilution of 16.3-fold (model extent was 51.5 meters) and 24.1-fold (model extent was 57 meters) would occur along the centerline of discharge. Reduction of toxic units from TLM calculations and WET tests were estimated based on the dilution of the overboard discharge waters by distance along the centerline of discharge using the dilution curves from the CORMIX model. Dilution modeling estimated reduction of the toxic units from the conservative TLM to result in a RCRs of less than 1.0 between 4 and 50 meters along the centerline of discharge. Using toxic units calculated from the WET tests this estimate would result in an RCRs of less 1.0, between 0 and 6 meters; again, suggesting minimal ecological risk.

[1] RCR = Predicted Environmental Concentration (PEC)/Predicted No Effect Concentration (PNEC))
An RCR < 1 indicates that the identified risks are acceptable.

1. INTRODUCTION

Exhaust emissions from the combustion of heavy fuel oil (HFO) by ocean-going vessels have been suggested to increase risks to human health and the environment. Beginning 1 January 2020, the International Maritime Organization (IMO) International Convention for the Prevention of Pollution from Ships (MARPOL) has required vessels to comply with new air emission standards for sulfur by: 1) using a very low sulfur fuel oil (VLSFO) with low (< 0.5%) sulfur content while operating globally or 0.1% sulfur marine gas oil (MGO) within designated emission-controlled areas, or 2) by installing an "appropriate exhaust alternative method" (IMO, 2019). The IMO has identified waters offshore of both coasts of North America extending 200 nautical miles from shore as emission-controlled waters which includes the entire Puget Sound (U.S. EPA, 2010). To assist with reducing aerial emissions of sulfur oxides and comply with these new standards, the majority of the cruise ship industry has installed Exhaust Gas Cleaning Systems (EGCS) on their vessels. The EGCS system removes sulfur oxides emissions by spraying ambient sea water into the engine exhaust prior to release via the vessel's stack. In addition to sulfur oxides, the EGCS system removes a portion of both pyrogenic and petrogenic polycyclic aromatic hydrocarbons (PAHs), metals, and other exhaust-associated constituents during this process (Fridell & Salo, 2016).

1.1 Ecological Risk Assessment Goals and Scope

The goal of this study was to provide empirical and modeled data to better understand the potential ecological risks from buffered EGCS discharge waters released to the Puget Sound in Washington, related to cruise ships operating open-loop EGCS. The Study originated in consultation with the Port of Seattle, the Washington State Department of Ecology, CLIA, and the Study Team. The focus of the agreement was to develop a program to perform a risk assessment following U.S. laws, WDOE standards and procedures, and recent IMO recommendations to better understand the potential impacts of EGCS discharges to the Puget Sound.

There are three major designs for an EGCS system, closed-loop, open-loop, and a hybrid configuration that can be operated in open- or closed-loop mode. Closed-loop systems recirculate EGCS scrubber waters multiple times with the aid of chemical treatment and temporarily store the recycled EGCS residues in tanks onboard the vessel until they can be discharged to a designated shore reception facility. Open-loop systems operate as a "once through" system, discharging buffered DeSOx tower outlet water to sea after contact with exhaust gases, thus there is no retention of waters associated with the EGCS onboard the ship. When operating in U.S. Vessel General Permit (VGP) waters (within 3 nautical miles of the U.S. coast), open-loop EGCS operators need to buffer their overboard discharge waters with seawater to attain a regulated pH level of $\geq$ 6.0 prior to discharge (VGP, 2013). Hybrid systems have the capability to run as either an open (during sea transits) or closed-loop system (for in port). Carnival Corporation, which provided the ships for sampling for this study operates vessels in the Puget Sound with open-loop EGCS systems, therefore the open-loop system was the focus of this project. Figure 1-1 provides a visual diagram of an open-loop EGCS system. These systems are outfitted with two sets of filters, one prior to the DeSOx tower and one set that filters the discharge water after exiting the DeSOx tower. The filters that are before the DeSOx tower are meant to remove particulates that could clog the nozzles that spray the ambient seawater into the tower. The filters that are after the DeSOx tower are intended to remove particulates from the discharge water and are used primarily in port. The vessels that were sampled during this project were all outfitted with filters to reduce the constituents in the DeSox discharge water, however these filters were not used during the sampling stage of the study to allow for a “worst-case”analysis .

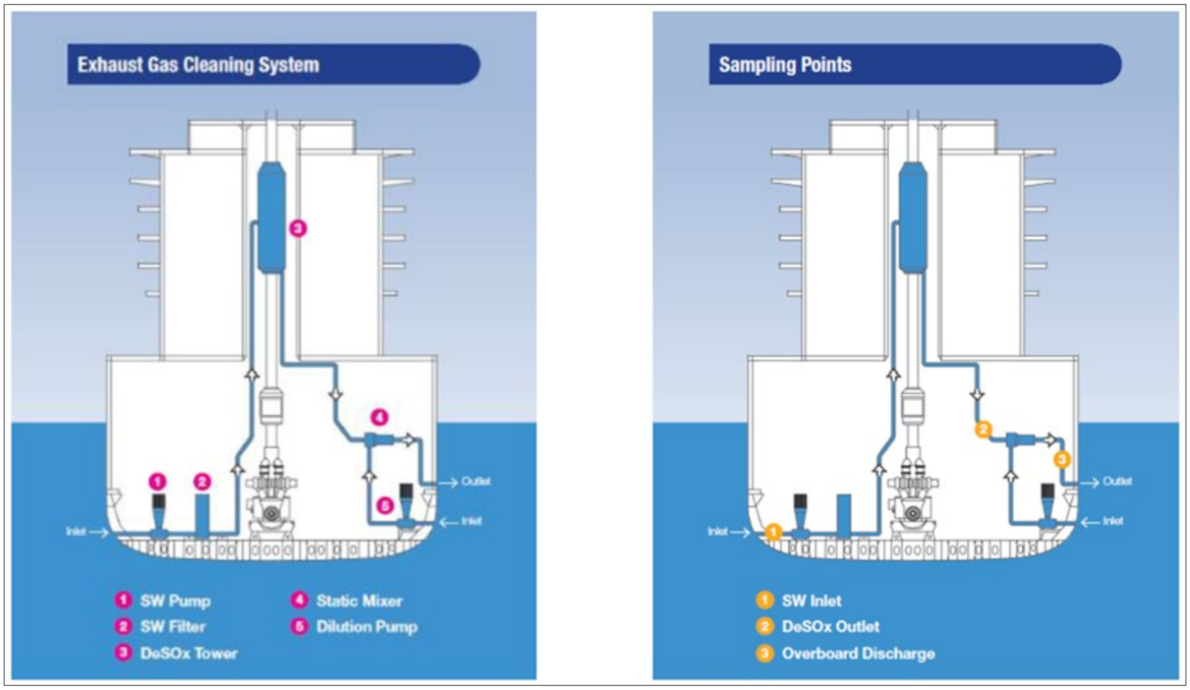

Figure 1-1. Exhaust Gas Cleaning System Diagram and Sampling Locations

It is important to note that modern cruise ships are typically powered by electric motors and that the electricity for these motors is supplied by independent diesel generators (DGs) consisting of a diesel engine and an associated generator. There are usually five or six of these DG sets onboard each ship and the power rating for each can be anywhere between 8 to 16 MW. In the ships used for this study, an EGCS is installed on the exhaust system of an individual diesel generator set and the discharges from each system are independent from other generators on the ship. The sampling points along the EGC system allow for collecting samples from the seawater intake, post DeSOx tower outlet and overboard discharge for each unique EGCS/generator configuration on the ship. Figure 1-1 illustrates the general configuration of these systems.

1.2 Study Team

The study team for this project is comprised of recognized experts in the fields of ecotoxicology, environmental fate and effects modeling, and ecological risk assessment. Table 1-1 provides the role and brief background for the major contributors.

Table 1-1. Study Team Description.

Name	Role	Background
Dr. Bill Stubblefield	Program Manager/Risk Assessment Expert	Oregon State University professor who is an internationally recognized expert in the field of environmental toxicology, with research that has focused on the areas of aquatic toxicology and ecotoxicology. He has extensive experience on performing evaluations of metals and hydrocarbons.
Dr. Cindy Wang	Fate and Transport Modeling Expert	Expert in the field of fluid mechanics which she uses to solve complex environmental problems involving contaminant fate and transport. She specializes in the study of mixing, flow, and transport processes via laboratory and field experiments as well as analytical and numerical modeling.

Name	Role	Background
Ms. Joy McGrath	PAH Risk Assessment Expert	Expert with over 20 years of experience in risk assessment, specializing in bioavailability of sediment-associated contaminants to the aquatic community, water and sediment criteria development and fate of contaminants through wastewater treatment processes. Ms. McGrath is a co-author on the original research for the target lipid model (TLM)
Mr. Jay Word	EcoAnalysts Project Manager	Project Manager with EcoAnalysts with over 20 years of practical experience of assessing toxicological impacts to model organisms.

1.3 Peer Advisory Group

The complexity and overall importance of this study has benefited from perspectives and approaches to evaluating the potential environmental concerns associated with the operation of EGCS from a peer advisory group. The research team provided the study plan to the peer advisory group to draw on the expertise, and guidance of external scientists who are experts in the field of ecological risk assessments and/or EGCS water evaluations. This group was composed of three individuals with expertise in aquatic toxicology, analytical chemistry, environmental fate modeling, and risk assessment. The advisory panel provided review and input to the proposed study design, review of intermediate and final project reports, and will review any ensuing publications. Members of the advisory group are:

- Dr. Bjørn Henrik Hansen, SINTEF, Trondheim, Norway
- Mr. Shinichi Hanayama, Class NK, Planning and Design Center for Greener Ships, Japan
- Dr. Wayne Landis, Western Washington University, Bellingham, WA

2. METHODS

2.1 Experimental Design

The goal of this study was to evaluate potential ecological risks associated with the operation of open-loop EGCS in the Puget Sound associated with cruise vessels, in this case represented by CLIA vessels from different cruise lines in Carnival Corporation. A combination of modeling and empirical analysis was used to define EGCS discharge water's constituents and exposure concentrations, provide a modeled assessment of EGCS discharge water toxicity, and validate model predictions using bioassay methods (Figure 2-1). Two load scenarios were assessed that used data collected under a low engine load scenario (~50%) for assessing in-port risk and a high engine load scenario (>70%) for assessing the risk while in transit. There may also be differences in sample chemistry attributable to differences in exhaust gas composition due to differences in engine design and size; therefore, a representative set of samples were collected from six ships, reflecting this variability. The low engine load scenario was designed to emulate discharges that would occur while in port while high loads were more typical of transit operations. However, due to local restrictions that prevented EGCS operation while in Port in Seattle, discharge water samples for low load scenarios were collected while in transit from Victoria BC to Seattle WA. As such, the seawater intake samples would likely contain lower background concentrations of the measured analytes than they would have had the samples been taken while at port, owing to the reduced anthropogenic inputs.

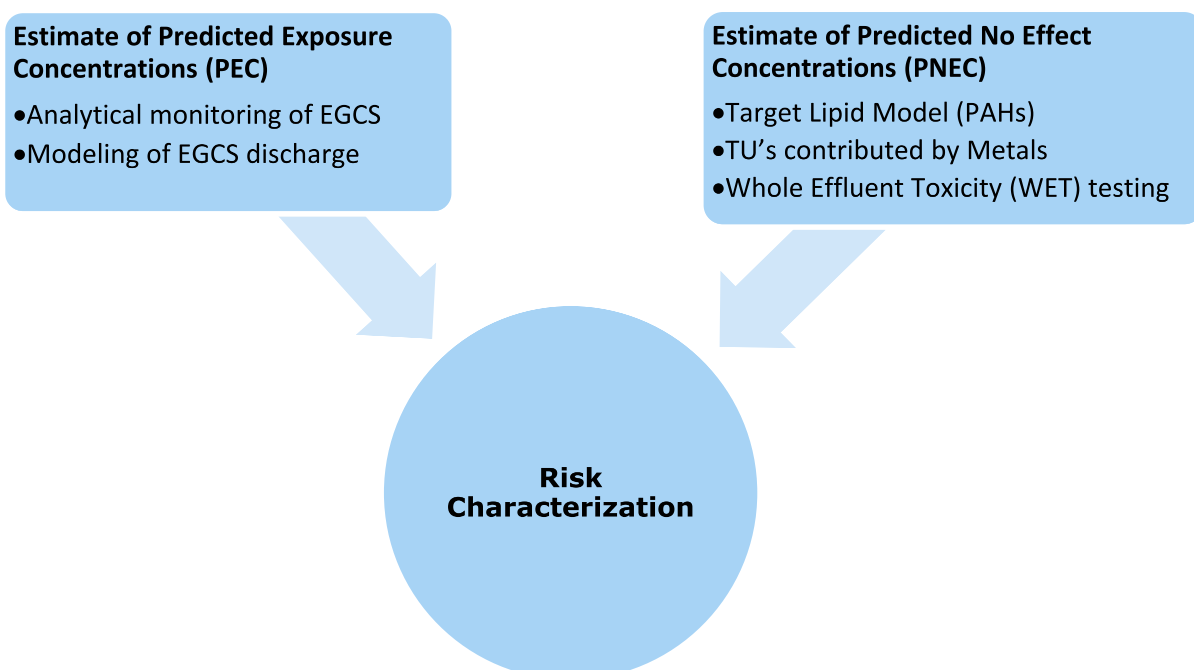

Figure 2-1. Ecological Risk Assessment Components

Although these engines can be fueled by different types of fuels, this research was limited to vessels powered by similar bunker HFO obtained from a common source with a typical range of sulfur content between 1.1 to 1.7%.

The energy demand on these engines could have a direct influence on how efficiently the engines consume fuel and, as a result, this may affect the composition and concentrations of constituents detected in the discharge water samples. The engine load scenarios were included as variables in the experimental design to bracket the range of potential engine load scenarios that are employed during normal Puget Sound operations. Fifty percent was chosen as the upper end of the low load scenario range because operating at a much lower load would not be a common scenario under normal operations while in the Puget Sound. The “hotel” component (i.e., minimum required electrical output while in port) on these vessels draws roughly 50 percent load from a single engine while in port in Seattle climate conditions. It should be noted that in-port operations are moving toward the use of on-shore power supplies rather than on-board sources, and five of the six Carnival Corporation ships visiting the Port of Seattle during the Study period were capable of using shore power. This capability will replace the need to use ship’s engines (and EGCS) for most port visits in the Port of Seattle in the next 2-3 years.

EGCS water samples were collected from cruise vessels operating engines of different sizes (and manufacturers) that are representative of the medium speed marine diesel engines common to the cruise industry worldwide and to cruise ships operating in Puget Sound. Samples were taken while ships were in US waters while transiting overnight from Victoria, British Columbia, Canada, to Seattle, Washington. Each sampling event (sample set) included collecting water samples from the seawater inlet (SI), EGCS desulfurization system (post-DeSOx) outlet (PD) and buffered overboard discharge (OD) from each cruise ship. A sample of the fuel prior to combustion was also collected on each ship. Full analytical chemistry parameters (i.e., PAH and metals) were assessed on the SI, PD, and OD samples. Metals concentrations were also analyzed on the fuel sample (see Section 2.4 for a full description of the analytes). Additionally, the whole effluent toxicity (WET) testing suite was performed for each buffered overboard discharge sample to determine potential toxicity to the organisms in the receiving waters. The SI sample was included in the WET test design as an ambient site water control. Sample nomenclature and abbreviations are included in Table 2-1. A complete list of samples (per sample set) and the analytes measured is included in Table 2-2.

Table 2-1. Sample Nomenclature Abbreviation

Abbreviation	Definition
LOW	Low Load
HIGH	High Load
F	Fuel
SI	Seawater Inlet
PD	Post-DeSOx
OD	Overboard Discharge

Table 2-2. Sampling Design (One Sample Set for Each Load Scenario)

Ship/Engine	Load Scenario	Sample Location/Type	Chemistry			WET tests			
			PAH	Metals	Other	Larval SPP Survival & Development	7-Day Menidia Chronic Survival & Growth	7-Day Mysid Chronic Survival & Growth	96-Hour *Skeletonema costatum* specific growth
Ship XX	NA	Fuel		X	X				
Ship XX	Low	Seawater Inlet	X	X	X	C	C	C	C
Ship XX	Low	DeSOx Outlet	X	X	X				
Ship XX	Low	Overboard Discharge	X	X	X	D	D	D	D
Ship XX	High	Seawater Inlet	X	X	X	C	C	C	C
Ship XX	High	DeSOx Outlet	X	X	X				
Ship XX	High	Overboard Discharge	X	X	X	D	D	D	D

Samples with gray sections were not used for WET testing.
C = For Control Sample only
D = Full Dilution Series

2.2 EGCS Water Sampling

The sampling events occurred on the Holland America Line (HAL) *Eurodam*; Princess Cruise Line (PCL), *Crown Princess* and *Discovery Princess*; and Carnival Cruise Lines (CCL), *Carnival Splendor* and *Carnival Spirit* vessels. A total of 6 rounds of EGCS water samples were collected and analyzed for this study. Table 2-3 provides the identification of the ships that were sampled and the dates of the sampling activities. While the intent was to sample two ships from each cruise line, the planned sampling of the HAL ship, *Westerdam,* could not be performed and, in its place, the PCL *Discovery Princess* was sampled during two sampling rounds. Although this vessel was sampled twice, unique diesel generators sets and their associated EGCS were sampled during each sampling round.

Table 2-3. Ships Sampled and Schedule

Ship	Engine Make and Type/DG with EGCS	Sample Dates (2022) VIC - SEA
HAL Eurodam	MaK - MAK12M43C (12MW) DG 1,2 MaK - MAK8M43C (8MW) DG 5,6	22 July Victoria – 23 July Seattle
Discovery Princess	Wärtsilä - 12V46F (14.4MW) DG 1,4 Wärtsilä - 14V46F (16.8MW) DG 2,3	30 July Victoria – 31 July Seattle 27 Aug Victoria – 28 Aug Seattle
Carnival Splendor	Wärtsilä - 12V46C (12.8MW) DG 1,2,4,5,6	08 Aug Victoria – 09 Aug Seattle
Crown Princess	Wärtsilä - W12V46CR (12.6MW) DG 2,3 Wärtsilä - W8V46CR (8 MW) DG 5,6	19 Aug Victoria – 20 Aug Seattle
Carnival Spirit	Wärtsilä - 9l46D (10.4MW) DG 1,2	05 Sep Victoria – 06 Sep Seattle

2.3 Sampling Procedures

The EGCS water samples were collected following Carnival's internal procedure, "*Exhaust Gas Cleaning System Sampling & Monitoring Compliance Handbook*" (Appendix A), which follows U.S. EPA sampling protocol, except when those procedures differed from those called for in the study plan (EcoAnalysts, 2022). For example, the handbook recommends collecting samples when the ship's generator is operating near full power, i.e., approx. 85% load, while the study plan describes collecting samples under the two selected power load scenarios (i.e., ≤50 and ≥70%).

In brief, EGCS water samples were collected by the study team, including third party observers, with assistance from ship's staff including the EGCS Engineer and Environmental Officer, all of whom have been properly trained in the method for collecting EGCS water samples. Oversight observers, including those from the Port of Seattle and Lloyd's Register, were present during each of the water collection deployments to observe and participate in the sampling process. Each ship's EGCS Engineer or Environmental Officer organized each of the sampling events with all parties and documented the EGCS system configuration and operation. The designated Engineer or Environmental Officer was also assisted by members of the ship's engineering staff during the sample collection process. Sampling kits were provided by the analytical and WET testing laboratories prior to the sampling date, each contained pre-cleaned sample bottles or Cubitainers® necessary for sample collection. Sample bottles contained appropriate volumes of preservative solution where necessary. Each sampling container was supplied with a label that was filled out completely by the sampling team. Each bottle was filled with the appropriate amount of sample and held under appropriate temperature conditions until packaged for offload and delivery to the laboratory.

The shipboard EGCS Compliance Computer was used to create a summary report for each sampling event. This report documented a summary of measured values for the continuously monitored parameters (pH, PAH, turbidity, seawater intake and buffering flow, and sulfur dioxide/carbon dioxide ratio) during the sampling efforts.

When the ships arrived in Seattle, the samples were couriered to the two laboratories where analysis or testing occurred. All samples were preserved with wet ice during transport. A temperature blank was included with each cooler so that the laboratories could measure temperature at time of receipt. A chain-of-custody (COC) form was attached to each shipping cooler and the cooler was sealed with a custody seal over the opening. Holding time recommendations for the analytical chemical parameters are provided in Table 2-4. WET samples were logged into the laboratory and water quality parameters were measured on the bulk samples.

2.4 Chemical Analysis

Analytical methods used in the analysis of water samples are provided in Table 2-4. PAH constituents were measured in the SI, PD, and OD samples. Historically, regulatory monitoring of EGCS discharged waters required analysis of the 16 parent PAHs identified by the U.S. EPA as primary pollutants (Table 2-5). However, following the Deepwater Horizon incident in 2010, the recommended approach for monitoring of PAHs was expanded to include measuring a number of alkylated PAH compounds and decalins, as well as other daughter PAH compounds. With this in mind, the current study measured an expanded group of sixty (60) individual PAH analytes.

Selected metal concentrations were also measured in SI, PD, and OD samples, in addition to metals concentrations measured in the fuel samples. Dissolved metals samples were collected without preservation and filtered at the analytical chemistry laboratory prior to analysis. This study analyzed the metal elements recommended by the Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP) Task Team; arsenic, cadmium, copper, chromium, nickel, lead, selenium, vanadium, and zinc (Linders, et al., 2019). In addition to the GESAMP recommended metals, the samples were also analyzed for mercury and thallium.

Other analytes that could affect the bioavailability and influence the toxicity (i.e., dissolved organic carbon, particulate organic carbon, chemical oxygen demand, nitrate, nitrite, pH, and total suspended solids) of the water samples were also measured. ALS Global (Kelso, WA) conducted the chemical analysis of the water samples for this study. ALS Global holds laboratory accreditations from NELAP (ORELAP ID WA100010), WDOE (WDOE – C544), and the Department of Defense (DOD – L22-478). See Appendix B for accreditation certificates.

Table 2-4. Analytical Methods

Analyte		Method	Preservation	Recommended Holding Time	Method Detection Limit	Method Reporting Limit	Units
PAH and Alkylated Homologues		8270D	NA	7-D	*	*	*
Total and Dissolved Metals	As	6020A	HNO_3 (total) / NA (Dissolved)[1]	180-D	0.2	0.5	µg/L
	Cd				0.008	0.02	µg/L
	Cr				0.03	0.2	µg/L
	Cu				0.05	0.1	µg/L
	Ni				0.04	0.2	µg/L
	Pb				0.006	0.02	µg/L
	Se				0.2	2	µg/L
	Tl				0.008	0.02	µg/L
	V				0.05	0.2	µg/L
	Zn				0.5	2	µg/L
Total and Dissolved Hg		7470A		28-D	0.02	0.2	µg/L
Total and Dissolved Cr (VI)		SM 3500-Cr	NA	24-H	0.004	0.05	µg/L
Nitrate and Nitrite		353.2	H_2SO_4	48-H	0.002	0.05	mg/L
pH		SM 4500-H+	NA	0-H	NA	NA	pH units
Total Chemical Oxygen Demand		SM 5220 C	H_2SO_4	28-D	20	50	mg/L
Dissolved Organic Carbon		SM 5310 C	H_2SO_4	28-D	0.07	0.5	mg/L
Total Suspended Solids		SM 2540 D	NA	7-D	5	5	mg/L
Particulate Organic Carbon		440.0	NA	100-D	NA	NA	NA

* See Appendix D for details on PAHs
NA = Not applicable
[1] Dissolved metal samples were collected unpreserved in the field and filtered at the analytical laboratory.

Table 2-5. Puget Sound EGCS Water Study PAHs

Puget Sound EGCS PAHs			
Acenaphthene	C1 - Decalins	C2-Naphthobenzothiophenes	C4 - Naphthalenes
Acenaphthylene	C1 - Dibenzothiophenes	C3 - Benzothiophenes	C4 - Naphthobenzothiophenes
Anthracene	C1 - Fluoranthenes/Pyrenes	C3 - Chrysenes	C4 - Phenanthrenes/Anthracenes
Benz(a)anthracene	C1 - Fluorenes	C3 - Decalins	Chrysene
Benzo(a)fluoranthene	C1 - Naphthalenes	C3 - Dibenzothiophenes	Decahydronaphthalene, Total
Benzo(a)pyrene	C1 - Naphthobenzothiophenes	C3 - Fluoranthenes/Pyrenes	Dibenz(a,h)anthracene
Benzo(b)fluoranthene	C1 - Phenanthrenes/Anthracenes	C3 - Fluorenes	Dibenzothiophene
Benzo(b)fluorene	C2 - Benzothiophenes	C3 - Naphthalenes	Fluoranthene
Benzo(b)thiophene	C2 - Chrysenes	C3 - Naphthobenzothiophenes	Fluorene
Benzo(e)pyrene	C2 - Decalins	C3 - Phenanthrenes/Anthracenes	Indeno(1,2,3-cd) pyrene
Benzo(g,h,i)perylene	C2 - Dibenzothiophenes	C4 - Benzothiophenes	Naphthalene
Benzo(k)fluoranthene	C2 - Fluoranthenes/Pyrenes	C4 - Chrysenes	Naphthobenzothiophene
Biphenyl	C2 - Fluorenes	C4 - Decalins	Perylene
C1 - Benzothiophenes	C2 - Naphthalenes	C4 - Dibenzothiophenes	Phenanthrene
C1 - Chrysenes	C2 - Phenanthrenes/Anthracenes	C4 - Fluoranthenes/Pyrenes	Pyrene

2.5 Metals Effects Estimate

Metals under most natural environmental conditions occur as a mixture; however, the current state-of-the-science for evaluating potential risks attributable to metals in the environment are based on a metal-by-metal approach and do not consider possible interactive effects among metals. Although in some instances metal mixtures have been shown to result in effects greater than would be predicted if the metals acted independently, these "greater than additive" effects are atypical and the "additive" or "less-than-additive" interaction accounted for >70% of the reported studies (Norwood, Borgmann, Dixon, & Wallace, 2003; Vijver, t, Peijnenburg, & De Snoo, 2011).

Norwood et al. (2003) provides the following recommendation for the assessment of metals mixtures:

"Unfortunately, the BLM [Biotic Ligand Model] approach is not yet available to assess metal mixtures. So, what is the best approach currently available for a site-specific risk assessment of metal mixtures? The best approach is probably to do an assessment of the toxicity of the mixture as present in situ, including a combination of in situ community composition and laboratory toxicity testing of field collected water and sediment samples."

This is the approach adopted for the EGCS risk assessment effort. As described, in addition to an evaluation of toxicity using the state-of-the-science models for predicting toxicity, water samples of the overboard discharge waters were collected and evaluated using a combination of empirical toxicity tests with a variety of sensitive marine species.

2.6 Target Lipid Model (TLM) PAH Effects Estimate

Post DeSOx waters are buffered with ambient seawater to produce a solution where the pH is greater than the VGP regulatorily mandated 6.0 pH and this water is discharged into the waters of Puget Sound. The composition of the chemical constituents in these waters may be related to the vessel's engine type, operating efficiency, load, and fuel characteristics. The results from the analysis of selected metals and sixty (60) PAHs which included parent and alkylated homologs[2] were used to estimate toxicity using the TLM.

2.6.1 Target Lipid Model (TLM) for Chemical Mixtures

The EGCS water monitoring data consisted of individual measurements of PAHs in the water phase. The TLM combined with the TU concept was used to estimate the toxicity of the mixture of these compounds in the water.

The TLM is applicable to Type I narcotics that act via a similar mode of action, (i.e., narcosis). Although the exact mechanism of toxicity by narcosis is not known, the lipid of an organism is believed to be reflective of the site of action (Di Toro, McGrath, & Hansen, 2000). This assumption is based on the well-known correlation between the exposure concentration of a chemical needed to induce an effect (e.g., LC_{50} -concentration causing mortality to 50% of the test organisms) and the chemical's log octanol-water partition coefficient (log K_{OW}), an indicator of the degree of a chemical's hydrophobicity. The TLM describes this relationship and provides a method for predicting the toxicity of Type I narcotics based on the chemical's log K_{OW} value. It is assumed that on an acute basis, PAHs exert a narcotic effect. The TLM has been validated for predicting the toxicity of petroleum components and was adopted by United States Environmental Protection Agency (U.S. EPA) as the technical foundation for developing criteria for PAHs and other non-ionic organic chemicals that act by a non-specific mode of action (Di Toro, McGrath, & Hansen, 2000; McGrath, Parkerton, Hellweger, & Di Toro, 2005; Burgess, et al., 2008; McGrath & Di Toro, 2009).

Equation 1 - TLM equation that predicts the acute water effect concentration.

$$\log(C_W^*) \ = m\log(K_{OW}) \ + log(C_L^*) + \ \Delta c$$

The TLM predicts the toxicity of PAHs in a sample based on Equation 1, where C_W^* is the critical aqueous concentration (mmol/L) (i.e., LC_{50} for a mortality endpoint), m is the narcosis slope, C_L^* is the Critical Target Lipid Body Burden (CTLBB) (µmol/$g_{octanol}$), and Δ*c* ($\log_{10}$ mmol/L) is a correction factor that is needed for some chemical classes (e.g., ketones, aromatic hydrocarbons, halogenated chemicals) to account for class-specific differences in partitioning behavior. The CTLBB is the chemical concentration in the organism lipid needed to achieve the observed effect and varies for each species. Since its development in 2000, the TLM chemical toxicity database has been updated to include aromatic heterocyclic compounds and additional sensitive species, such as sea urchins. The more recent validation efforts of the TLM in 2018 and 2021 include those additional data (McGrath, et al., 2018; McGrath, et al., 2021). The current TLM database includes over 240 chemicals and 79 species across eight taxonomic groups.

PAHs are usually found in the environment as mixtures of chemicals (e.g., EGCS water) rather than single chemicals. Since these chemicals have an assumed similar mode of toxic action (i.e., narcosis), it is appropriate to use the toxic unit theory of additivity and simply sum the toxic contribution of

[2] Biphenyl and the decalin series are structurally not polycyclic aromatic compounds. Biphenyl is a monocyclic aromatic hydrocarbon and the decalins are saturated hydrocarbons. They are petroleum compounds and are often measured as part of the PAHs. For the purposes of this report, all measured compounds are referred to as PAHs.

each chemical in the mixture to compute the overall toxicity of the mixture (Di Toro, McGrath, & Hansen, 2000). For each PAH in the mixture, the PAH-specific toxic unit is computed by dividing the PAH-specific measured chemical concentration in the water by the PAH-specific TLM predicted threshold as follows:

Equation 2 - TLM predicted threshold.

$$TU(individual) = Cmeasured/(TLM\ threshold)$$

The toxic units of the individual PAHs in the mixture are then summed to compute the total toxic units of the mixture as follows:

Equation 3 – TLM total toxic units of the mixture

$$TU(total) = \sum TU$$

Theoretically, if the total TUs of the mixture are greater than or equal to 1, the mixture is predicted to be toxic according to the endpoint as defined in the threshold. This approach assumes independent action from each of the PAH components; no interactive effects are considered. Available data do not suggest any type of enhanced interactive effects (i.e., synergism) occurring among PAHs, in fact, some degree of conservatism is likely in model predictions due to the conservative method used in calculating the acute and chronic CTLBB (see Section 2.6.2).

2.6.2 TLM Coefficients

TLM coefficients were taken from the most recent calibrations of the model (McGrath, et al., 2018; McGrath, et al., 2021). Required parameters include the universal slope, chemical class corrections Δc, and the CTLBB. The universal slope is -0.94. The chemical class corrections for monocyclic aromatic hydrocarbons, polycyclic aromatic hydrocarbons and heterocyclic aromatic compounds are -0.025, -0.364 and -0.472, respectively. Decalins are structurally baseline narcotic chemicals and do not require a chemical class correction. The CTLBB selected was the 5th percentile acute CTLBB of 18.6 µmol/g octanol calculated following the U.S. EPA method for computing final chronic value (Stephen, et al., 1985). The acute CTLBB was converted to a chronic CTLBB of 3.6 µmol/g octanol by applying an acute-to-chronic ratio (ACR) of 5.2. These 5th percentile CTLBB are protective of 95% of species. When the 5th CTLBB is used in TLM equation, the computed water concentration represents the concentration of a chemical below which effects are expected to only five percent of species.

The TLM acute and chronic aqueous thresholds are presented in Table 1 in Appendix E, along with physical chemical properties needed for the analysis. One caveat with some of the heavier molecular weight PAHs is that their bioavailability is limited due to their reduced water solubility. The assumption that nonpolar organic chemicals have reduced bioavailability led to an empirical modification in the TLM that limited the lipid-water partition ratio at a log K_{OW} value of 6 (Redman, Parkerton, MacGrath, & Di Toro, 2012). Redman et al. (2014) demonstrated a lack of bioavailability for chemicals with increasing log K_{OW} values such that no toxicity was observed for chemicals with a log K_{OW} value greater than ~6. PAHs with log K_{OW} values ≥ 6 are retained in the TLM analysis, but the toxicity estimate is set to an (de Vries, Jak, & Frost, 2022) equal value for all PAHs with a log K_{OW} greater than or equal to 6. This approach leads to conservatism in the TLM predictions of toxicity, often resulting in an overestimate of predicted toxicity as described by Parkerton et al (2021) and deVries et al (2022).

2.7 Whole Effluent Toxicity Testing

WET test methods used for this study included species and endpoints that are recognized both federally by the U.S. EPA, and locally by the Washington State Department of Ecology (WDOE). Test species were chosen to include a range of trophic levels (plant, invertebrate, fish) and potential toxicant sensitivities. These tests included tests conducted with the Inland Silverside (fish; *Menidia beryllina*), mysid shrimp (crustacean; *Americamysis bahia*), a larval sand dollar (echinoderm; *Dendraster excentricus*), and a marine diatom (algae; *Skeletonema costatum*), all of which were tested at sensitive life stages. Except for the marine diatom test, these test methods are commonly used by WDOE to monitor compliance with other marine discharges to the Puget Sound (Table 2-6); These tests have been shown to be good predictors of potential toxic responses among marine species and they span the sensitivity range of species of the ecosystems that have been studied (U.S. EPA, 1991 and ASTM, 1998). The WET tests were used to verify the estimated toxicity from the TLM and metals modeling and provide empirical test data on the toxicity of the EGCS discharge waters. EcoAnalysts' toxicity laboratory in Port Gamble, Washington performed all toxicity testing, except for the marine diatom test, which was performed by Pacific EcoRisk in Fairfield, CA. EcoAnalysts and Pacific EcoRisk hold federal accreditation through NELAP (EcoAnalysts – ORELAP ID 4165; Pacific EcoRisk – ORELAP ID 4043) as well as Washington State accreditation for conducting toxicity testing (EcoAnalysts – WDOE C897; Pacific EcoRisk – WDOE C848). See Appendix B for full accreditation details.

Table 2-6. WET Test Methods and Guidance References

WET Test Methods		
Test	**Test Method**	**Test Acceptability Criteria**
Inland Silverside (*Menidia beryllina*) 7-Day Larval Survival and Growth Test	WQ-R-95-80 (WDOE, 2016); EPA-821-R-02-014 (U.S. EPA, 2002)	Control: ≥ 80% survival; ≥ 0.50 mg growth
Mysid (*Americamysis bahia*) 7-Day Larval Survival and Growth Test	WQ-R-95-80 (WDOE, 2016); EPA-821-R-02-014 (U.S. EPA, 2002)	Control: ≥ 80% survival; ≥ 0.20 mg growth
Sand dollar (*Dendraster excentricus*) Survival and Development Test	WQ-R-95-80 (WDOE, 2016); EPA/600/R-95/136 (U.S. EPA, 1995)	Control: ≥ 80% Normal development; MSD <25%
Microalgae, (*Skeletonema costatum*) 96-hour Growth Inhibition Test	ASTM E1218-97a	Control: Specific growth rate ≥0.9/day. CV specific growth rate ≤ 7%

2.7.1 WET Test Statistical Methods

Determination of study point estimates of toxicant effects concentrations shown to reduce survival, development or growth (depending upon species and test design) by 10%, 25%, and 50% relative to background (Seawater Intake) sample performance (LC_{10}/LC_{25}/LC_{50} and EC_{10}/EC_{25}/EC_{50}) were made according to the EPA guidance (U.S. EPA, 2002) for each bioassay endpoint used during this study. The LC_{xx}/EC_{xx} values were chosen to provide both national and international perspectives for the WET test results. EC_{10} values are typically used by European regulators to estimate chronic impacts using chronic biological endpoints (growth, development, etc. (ECHA, 2008)). EC_{25} values are used in the U.S. to estimate chronic impacts from WET testing while EC_{50} values are used for estimating acute impacts (i.e., survival) in both jurisdictions.

Point estimates were calculated using CETIS™ statistical software (Comprehensive Environmental Toxicity Information System [CETIS], version 1.8.7.4, Tidepool Scientific Software, McKinleyville, CA, USA). For quantal or "all or nothing" responses (such as mortality), the point estimation tools used either the Linear Regression (Probit), Trimmed Spearman-Karber, or Linear Interpolation method, depending upon the outcomes of the respective stepwise statistical assumptions. For non-quantal data (such as biomass) the Linear Interpolation method was used for point estimation. Transformations of the data, (e.g., arc sine square root and logs), are used where necessary to meet assumptions of the proposed analyses, such as the requirement for normally distributed data. The point estimate values were selected for data analysis from amongst the possible statistical endpoints because the use of hypothesis testing endpoints such as No Observed Effect Concentration (NOEC) and Lowest Observable Effect Concentration (LOEC) values rely entirely on the selection of test concentrations and numbers of replicates (Warne & van Dam, 2008; Isnard, 2001).

2.7.2 WET Test Endpoints

The endpoints used to evaluate the toxicity of the overboard discharge (OD) waters are described in Table 2-7.

Table 2-7. Test endpoints used to evaluate the toxicity of Overboard Discharge (OD) waters.

Test Species	Endpoint	Description
Silverside, Mysid	Survival	Endpoint is based on the organisms that are alive at the end of the method-defined test period. To determine survival, the count of the organisms that remain alive at the end of testing is divided by the initial count of organisms exposed at the start of the test in each replicate.
Silverside, Mysid	Biomass	Endpoint combines survival and growth endpoints. Biomass is calculated by dividing the total (dry) weight of fish surviving to test termination in each chamber, by the total number of larvae that were placed in that chamber at the start of the test.
Sand dollar	Combined Proportion Normal	Endpoint is calculated by dividing the number of normally developed larvae by the average stocking density. Average stocking density is the mean number of fertilized eggs that are added to chambers that are preserved at the beginning of the experiment. These chambers are randomly allocated eggs throughout initiation of the test.
Algae	Specific Growth Rate	Endpoint is a measure of the increase in biomass per day. It is calculated by subtracting the natural log of the original cell density from the natural log of the final density and dividing the result by the number of days.

2.7.3 WET Test Toxic Unit Calculation

Toxic units were calculated for each of the WET test endpoints. Assessment of chronic toxic units were calculated using either the EC_{10} or EC_{25} while acute toxic units used the EC_{50} results (EC_{25} results are also provided). The EC_{xx} results were calculated based on the percentage of overboard discharge water that was used for the WET test dilutions. This dilution series included 6.25, 12.5, 25, 50 and

100% (overboard discharge sample water) concentrations. Toxic units were calculated by dividing 100 by the EC_{xx} values.

2.8 Predicted Environmental Concentrations (PECs)

"The exposure assessment was considered an essential part of a risk assessment", as pointed out by GESAMP (Linders, et al., 2019). Exposure assessment utilizes various modeling techniques to identify the PEC of the relevant chemicals of concern (e.g., pH, trace metals or a PAHs) in effluent discharge for which the risk assessment is carried out. The models can also be used to predict the dilution of the discharge, which could be used in connection with whole effluent toxicity testing.

GESAMP recommends using the MAMPEC model "as a Tier 1 tool to estimate the PEC of the chemical under consideration." The MAMPEC model is an integrated hydrodynamic and chemical fate model that has been used by the IMO to simulate PECs in harbors, estuaries, rivers, and open waters. GESAMP also recommends doing "a side-by-side comparison" of MAMPEC and other models to validate modeled PEC concentrations. This study uses the CORMIX model for comparison. CORMIX is a U.S. EPA-supported mixing zone model and decision support system for environmental impact assessment of regulatory mixing zones resulting from continuous point source discharges.

All estimations of chemical constituent background concentrations were made based on the seawater intake samples collected during the sampling efforts for each vessel. These samples were collected while transiting from Victoria to Seattle while in Puget Sound waters. Seawater intake samples collected during the low engine load scenario were used for estimating background conditions in the Port, however, these results are likely lower than background conditions in Port waters and are likely more reflective of regional background concentrations for the measured compounds.

2.8.1 MAMPEC Model

Estimates of water quality impacts due to the EGCS water discharges by cruise ships while the ships are at berth in the Port of Seattle, WA, and moving in the shipping lanes of Puget Sound were developed. The Predicted Environmental Concentration (PEC) of key constituents of concern in discharged waters was computed using the MAMPEC 3.1 model, an integrated hydrodynamic and chemical fate model that has been used to simulate PECs in harbors, estuaries, rivers, and open waters (van Hattum, van Gils, Elzinga, & Baart, 2014). The key constituents were selected based on the TLM and the metals effects estimate results. Additional details on the model setup and input values are provided in Appendix G.

2.8.1.1 Port of Seattle - Model Setup

MAMPEC's Estuarine Harbour module was used to model the PECs associated with discharge from a single ship at berth in the Port of Seattle and the PECs in waters flushed out to the Puget Sound. Two renderings of the MAMPEC Estuarine Harbour model were evaluated with different inputs for the port dimensions. In the first analysis (large domain) , the MAMPEC model was utilized in a typical and standard way however, as the model employs a fixed grid of 10 cells by 10 cells, this resulted in a relatively large cell size of 390 m by 300 m, which is useful for understanding the distribution and accumulation in the modeled area but is insufficient resolution for describing the dynamics immediately adjacent to the discharge point. Therefore, a second model setup (small domain) was employed limiting the Port Area and thereby decreasing the cell size to 69.5 m by 63.8 m.

As shown in Figure 2-2, the model simulations were conducted using a simplified geometry as the model domain that included:

- The first Estuarine Harbour model (for a large domain) used inputs of the port geometry based upon the following assumptions – The Port of Seattle area dimensions 3,900 m wide (X2) 3,000 m (Y1) in length, opening width of 3900 m (X3) with a depth of 62 m. The Puget Sound, which is connected to the Port of Seattle was modeled as a straight channel with a length of 6,500 m (X1) and width of 6,500 m (Y2).
- The second Estuarine Harbour model (small domain) focused on the Smith Cove Cruise Terminal area in the Port of Seattle. Which is represented in the model as rectangular harbor with a width of 695 (X2) length of 638 m (Y1), opening width of 695 m (X3), and depth of 10 m; and the Puget Sound (considered to be the confines of Elliott Bay), connected to the Port on one side, as a straight channel with a length of 700 m (X1) and width of 3,900 m (Y2).

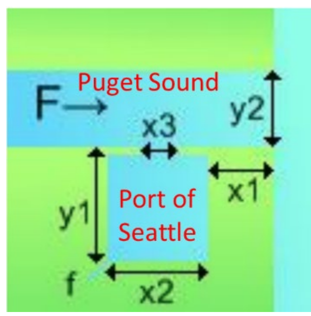

Figure 2-2. Schematic map showing the model domain of the Port of Seattle simulation using MAMPEC Estuarine Harbour module.

Typical hydrodynamic and atmospheric data in the Puget Sound and Port of Seattle, including flow velocity, salinity, temperature, tides, wind, and cloud cover, were obtained through a literature review, and applied as model input parameters in the MAMPEC Estuarine Harbor module (see Appendix G for more details).

The ship was assumed to operate continuously under low engine load while in port in Seattle. While a single ship will typically only spend a fraction of its time in the Port (9-10 hours per week), this assumption of a continuous presence was applied to obtain a conservative estimate of the maximum discharge loading from a single ship. The ship arrival and departure pattern in Port of Seattle also means that there are substantial periods of time when no ships are present in the Port, or most ships in port are on shore power therefore, the assumption of a single continuous discharge is still conservative for simulating discharge loading of multiple ships in Port. The measured discharge flow rate for low load conditions was taken as the mean value from six rounds of ship sampling. The discharge rate for each sampling round was assumed to be continuous and constant in time, i.e., the rate does not vary over the course of each sampling round and is taken as the average of the samples collected at the start (or near the start) and end (or right after completion) of each sampling period. Background and discharge concentrations were estimated from samples analyzed for the constituents of interest which were defined by the TLM and metals effects estimate. The background concentration measurements were assumed to be also representative of ambient conditions in the Port of Seattle. Similarly, the effluent discharge rates and concentrations measured from the ships were assumed to be also representative of long-term loading conditions in the Port.

The mean values from the six sampling campaigns for the measured background and discharge concentrations of the constituents, as well as the mean discharge rate under low engine load, were used as constant inputs in the model to conduct steady-state simulations of the long-term loading due to a single ship at berth. To maximize the simulated total concentrations[3], the model was configured conservatively to assume no removal of discharged constituents from the water environment through sedimentation, photolysis, hydrolysis, or biodegradation.

The model was run with background concentration considered separately for the volumetric fraction of both the receiving water and the overboard discharge, to determine: 1) the concentration in the receiving water due solely to the discharge (the discharge concentration includes the seawater inlet background concentration plus any increase due to the scrubbing process). 1) the concentration in the receiving water due solely to the discharge, and 2) the volumetric fraction of discharge water in the receiving water. Then, the ambient receiving water concentrations were computed using the following equation.

Equation 4 – MAMPEC ambient receiving water concentrations

$$C = f_D C_D + (1 - f_D) C_B$$

where C is the calculated receiving water concentration, C_D is the discharge concentration, C_B is the background concentration, and f_D is the volumetric fraction of discharge water in the receiving water. The MAMPEC model assumes equal dispersion throughout each cell and provides an average concentration in each cell. For this study, most of the dilution was observed in the cell that was nearest the discharge port. To refine the MAMPEC results, additional modeling was performed using the CORMIX model.

2.8.1.2 Shipping Lanes in Puget Sound – Model Setup

MAMPEC's Shipping Lane/Open Sea module was used to model the maximum (or acute) PEC in the vicinity of the discharge from ships while moving in Puget Sound. As shown in Figure 2-3, the model domain included a segment of the Puget Sound shipping lane with a length of 20 m along-channel (x), a width of 10 m across-channel (y), and a depth of 4 m. As the Shipping Lane module assumes a grid with fixed number of cells (20 by 10) in the domain, each cell has a size of 1 m by 1 m. In the MAMPEC model, the discharge from moving ships is distributed along the center line of the shipping lane segment in the direction of the ship movement and flow velocity (assuming both move along x though direction may be different) (Figure 2-3).

[3] Total concentration in water (Cw-t) expressed in µg/L is the sum of the concentration bound to suspended particulate matter in µg/L (Cw-SPM), the concentration bound to dissolved organic carbon in µg/L (Cw-DOC) and the freely dissolved concentration in µg/L (Cw-fd): Cw-t = Cw-SPM + Cw-DOC + Cw-fd. The calculation of the partitioning between the different phases is based on individual constituent properties.

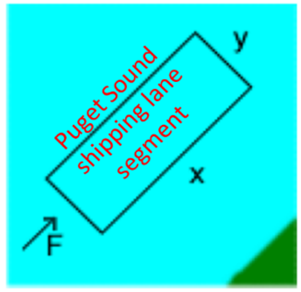

Figure 2-3. Schematic map showing the model domain of the Puget Sound simulation using MAMPEC Shipping Lane/Open Sea module.

Typical hydrodynamic and atmospheric data in Puget Sound, including flow velocity, salinity, temperature, tides, and cloud cover, and applied as model input parameters in the MAMPEC Shipping Lane/Open Sea module (see Appendix G for more details). To replicate local flow conditions relative to the ship, the flow velocity (F) was set equal to the relative speed between the ship and the ambient water currents, which were assumed to be in the same (x) direction along the channel of Puget Sound.

The ship was assumed to operate under high engine loads while moving. The simulated discharge flow rate under high load conditions was taken as the mean value from six rounds of field measurements on moving ships in Puget Sound. Sampled background and discharge concentrations were analyzed and averaged over six sampling rounds for the constituents of interest that were identified by the TLM and the metals risk assessment. The background concentration measurements were assumed to be also representative of ambient conditions in the Puget Sound. Similarly, the effluent discharge rates and concentrations measured in effluent from the ships were assumed to be also representative of the loading from the ship while moving.

The mean values from the six sampling campaigns for the measured background and discharge concentrations of the constituents, as well as the mean measured discharge rate under high engine load, were used as constant inputs in the model to conduct simulations due to a single ship moving through Puget Sound. To maximize the simulated total concentrations, the model was configured conservatively to assume no removal of discharged constituents from the open sea environment through sedimentation, photolysis, hydrolysis, or biodegradation.

The model was run with background concentration considered separately for the volumetric fraction of both the receiving water and the overboard discharge, to determine: 1) the concentration in the receiving water due solely to the discharge (the discharge concentration includes the seawater inlet background concentration plus any increase due to the scrubbing process), and 2) the volumetric fraction of discharge water throughout the model domain. Then, the ambient receiving water concentrations were computed using Equation 4 in Section 2.8.1.1.

2.8.2 CORMIX Model – Smith Cove Cruise Terminal in Port of Seattle

Estimates of dispersion modeling to evaluate and visualize the spatial distribution of EGCS water concentration in the vicinity of ship discharge at the Smith Cove Cruise Terminal in the Port of Seattle were developed using the CORMIX (v12.0GT). The model domain included Port waters at the Smith Cove Cruise Terminal at Pier 91 to assess initial dilution near the source (Figure 2-4). For the Smith Cove Cruise Terminal application, the horizontal domain was set to unbounded because the nearfield region was smaller (<100 m) than the distance to the opposite bank (>1,000 m). The vertical extent of the model domain for the surface discharge was limited to the top 10 m of the water column, which is equal to the depth at the terminal.

EGCS water concentrations in the Port of Seattle were simulated for one ship at berth discharging continuously using analytical data collected for the low engine load scenario. A summary of model input parameters and values used in simulating the PECs in the Port of Seattle is included in Appendix H. Additionally, schematics for the geometry of the discharge configuration in the CORMIX model are shown in Figure 2-5.

EGCS discharged water was modeled as a conservative surface buoyant discharge at the Smith Cove Cruise Terminal using the CORMIX3 core hydrodynamic simulation model. This is conservative because a surface discharge experiences less dilution and dispersal potential compared to a near-surface discharge. The discharge flow rate, temperature, and salinity were based on the MAMPEC assessment of PEC of key chemicals of concern in the Port of Seattle (Elliott Bay), and the discharge outlet configuration was considered in the modeling analysis using CORMIX.

This model was performed using two tidal scenarios. The first modeled scenario was of a slack tide assessment where the ambient currents were 0 m/s which provided an assessment of the dilution of the discharge when local currents, flushing, and mixing are weakest. Additionally, the model was run under at time of one hour after slack tide, with a weak ambient flow of 0.1 m/s.

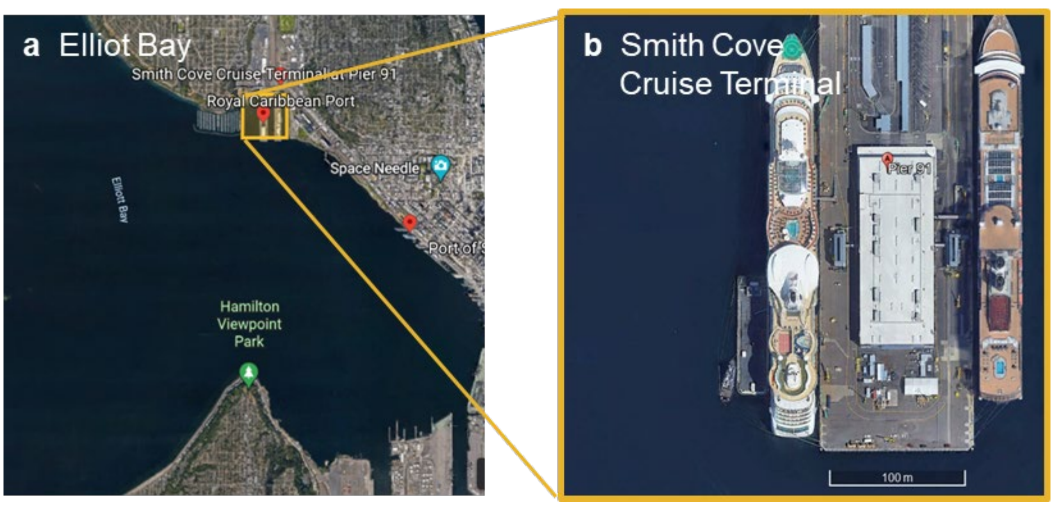

Figure 2-4. Map of Elliott Bay (left) and Smith Cove Cruise Terminal (right), outlined in yellow.

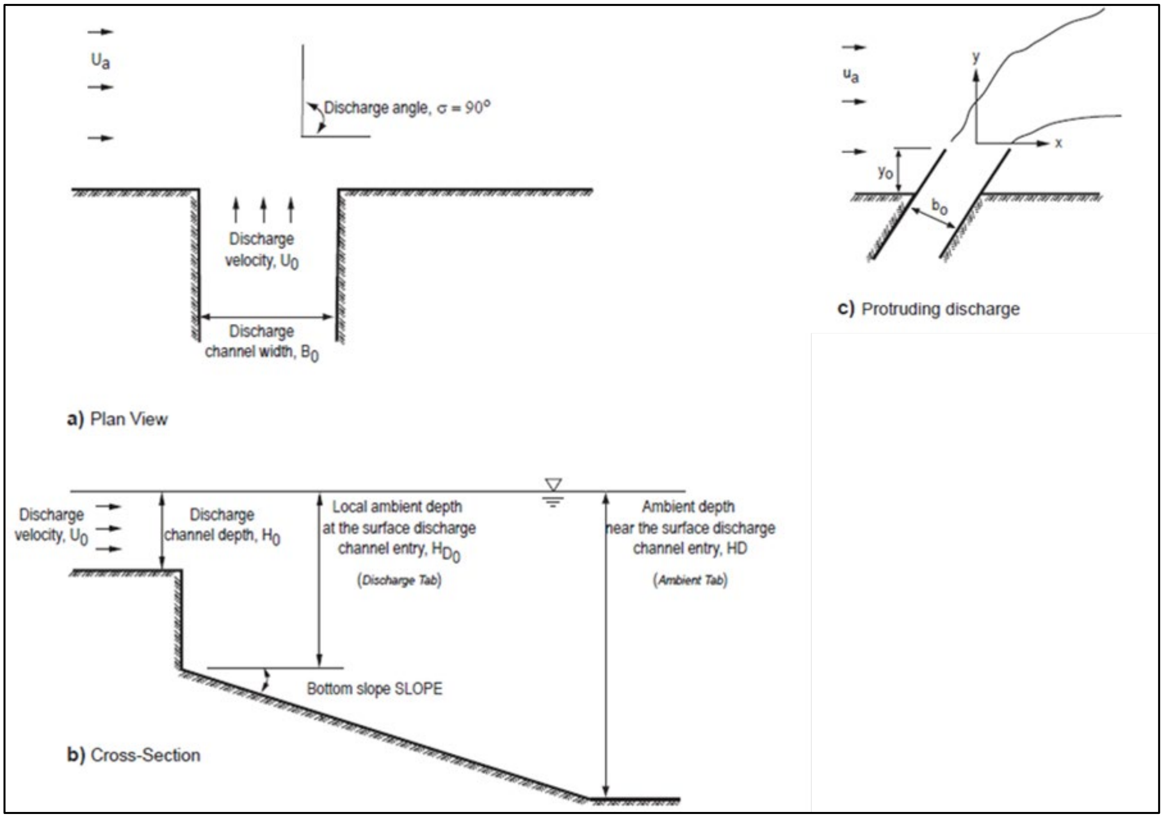

Figure 2-5. CORMIX3 discharge channel geometry and discharge configurations (adapted from the CORMIX User Manual).

3. RESULTS

This section contains the results of the analytical characterizations and empirical testing that was performed on the SI, PD, and OD waters. These sections discuss the sampling schedule and procedures (Section 3.1), the results of the chemical analysis (Section 3.2) and the results of the WET testing (Section 3.4).

3.1 Sampling Schedule and Procedures

Water samples were collected from cruise ships from July 23 to September 6, 2022. The sampling team boarded the cruise ships in Victoria, Canada the evening prior to sampling and debarked in Seattle, Washington (USA) the following morning. Water samples were collected in the early morning after the cruise ships had entered the Puget Sound in United States territorial water. Table 3-1 provides a summary of some of the details associated with each collection including the engine type, diesel generator sampled, fuel type and reported percent sulfur as well as the sampling team. Each water collection was assisted and observed by at least one third party representative from the Port of Seattle and/or Lloyd’s Register. In addition to the sampling team, the water collection effort was assisted by the vessel staff including the EGCS Engineer.

Table 3-1. EGCS Water Collection Details

Ship	Engine type	DG #	Sample Date	Sampling Team
Holland American Line Eurodam	MAK8M43C (8 MW)	#6	July 23, 2022	EcoAnalysts - Jay Word Port of Seattle – Randy Edwards and Danielle Butsick Lloyd's Register – Sergiy Savelyev Carnival Corporation - Valerie Chatterley
Discovery Princess	14V46F (16.8 MW)	#3	July 31, 2022	EcoAnalysts – Mary Ann Rempel-Hester Port of Seattle – Jane Dewell Lloyd's Register – Weitze Gjaltema Carnival Corporation – Valerie Chatterley
Carnival Splendor	12V46C (12.8 MW)	#6	August 9, 2022	EcoAnalysts – Jay Word Port of Seattle – Ryan Child Lloyd's Register – Weitze Gjaltema Carnival Corporation- Valerie Chatterley
Crown Princess	W8V46CR (8 MW)	#6	August 20, 2022	EcoAnalysts – Mary Ann Rempel-Hester Port of Seattle – Randy Edwards Carnival Corporation - Valerie Chatterley
Discovery Princess	12V46F (14.4 MW)	#4	August 28, 2022	EcoAnalysts – Mary Ann Rempel-Hester Port of Seattle – Rosalba Courtney Lloyd's Register – Weitze Gjaltema Carnival Corporation- Valerie Chatterley
Carnival Spirit	9l46D (10.4 MW)	#3	September 6, 2022	EcoAnalysts – Jay Word Port of Seattle – Detta Dayton Oregon State University – William Stubblefield Carnival Corporation – Valerie Chatterley

MW = Megawatts
DG = Diesel Generator

3.2 EGCS Flow Measurements During Sampling

Measured seawater intake and buffering pump flows were relatively consistent across all ships with flows under high engine loads typically exceeding those during low engine loads (Table 3-2). Buffering flow waters represented ⅔ to ¾ of the total overboard discharge volume. Buffering water flows are required to achieve federal regulatory pH standards, i.e., $\geq$ pH 6.0, for overboard discharges.

Table 3-2. EGCS Flow Measurements During Sampling

Ship	Engine Load	Seawater Intake Pump Flow (m^3/h)		Buffering Pump Flow (m^3/h)		Overboard discharge Flow (m^3/h)[1]		Avg % dilution buffering flow[1]
		Start	End	Start	End	Start	End	
Holland American Line Eurodam	High	618	605	1202	1142	1820	1747	65.7
	Low	487	470	1145	1166	1632	1636	70.7
Discovery Princess	High	632	687	1420	1407	2052	2094	68.2
	Low	640	709	1436	1445	2076	2154	68.1
Carnival Splendor	High	475	521	1460	1401	1935	1922	74.2
	Low	333	319	1040	764	1373	1083	73.1
Crown Princess	High	535	529	847	982	1382	1511	63.1
	Low	536	531	1023	868	1559	1399	63.8
Discovery Princess	High	709	708	1437	1424	2146	2132	66.9
	Low	529	508	1173	649	1702	1157	62.5
Carnival Spirit	High	449	385	1248	1229	1697	1614	74.8
	Low	263	394	1325	671	1588	1065	73.2

[1] Overboard discharge flow and average dilution calculated based on measured Seawater Intake and Buffering pump flow

3.3 Chemical Analysis

Results from the chemical analyses performed on EGCS waters and associated samples are presented in this section. These results are provided by fuel (Section 3.3.1), PAHs (Section 3.3.2), metals (Section 3.3.2), and other parameters (Section 3.3.4). All analytical chemical reports are included in Appendix D.

3.3.1 Fuel Chemistry Results

The sulfur content of the fuel was analyzed to verify the reported percent sulfur concentrations. Fuel oil was collected from the diesel generator fuel supply line while the EGCS water sampling was occurring. In every case, the measured sulfur concentration was lower than the value reported in the bunkering notes (Table 3-3). Table 3-4 presents the mean metal concentrations that were measured in the fuel samples.

Table 3-3. Fuel Chemistry Results (Sulfur)

Ship	Fuel Type	Bunker Receipt Reported Sulfur (%)	Measured Sulfur (%)	Measured Sulfur (mg/Kg)
Holland American Line Eurodam	RMG 380[1]	1.67	0.93	9300
Discovery Princess	IFO 380	1.31	0.76	7600
Carnival Splendor	IFO 380	1.26	0.65	6450
Crown Princess	IFO 380	1.19	0.78	7750
Discovery Princess	IFO 380	1.13	0.82	8230
Carnival Spirit	IFO 380	1.11	0.83	8310
Mean ± S.D.			**0.794 ± 0.094**	**7,940 ± 942.6**

[1] – Residual Marine Fuel Oil (RMG 380) and intermediate fuel oil (IFO) 380 are synonyms

Table 3-4. Mean Fuel Chemistry Results (n=6)

Total Metal	Holland American Line Eurodam	Discovery Princess	Carnival Splendor	Crown Princess	Discovery Princess	Carnival Spirit	Mean ± S.D. (mg/Kg)
As	0.12	0.1	0.08	0.1	0.08	0.1	0.10 ± 0.02
Cd	ND	0.005	0.005	0.005	0.006	0.01	0.006 ± 0.002
Cr	0.16	0.3	0.22	0.23	0.22	0.12	0.22 ± 0.05
Cu	0.174	0.631	0.338	0.206	0.364	0.232	0.35 ± 0.16
Pb	0.108	0.381	0.527	0.406	0.407	0.188	0.37 ± 0.14
Hg	ND	ND	0.002	ND	ND	0.003	0.0025 ± 0.0007[1]
Ni	22.7	20.5	17.1	17.8	17.2	8.81	18.8 ± 2.3
Se	0.11	0.09	0.09	0.08	0.06	0.24	0.13 ± 0.11
Tl	0.005	0.009	0.01	0.007	0.006	0.0095	0.009 ± 0.004
V	52.1	42.6	34.2	36.3	35.5	18.36	39.6 ± 6.8
Zn	2.49	4.47	1.96	1.82	3.34	1.56	2.61 ± 1.11

[1] – Mean and SD based on detected concentrations

3.3.2 Summary of Metals Analyses

Water metals concentrations are summarized in Table 3-5. The mean value for each metal is provided with the standard deviation for each sample set. Dissolved and total copper in the SI samples, which represents ambient conditions, collected during the high engine load in one ship for Round 3 were unusually high, 26.0 (total copper) and 20.5 µg/L (dissolved copper). Seawater Intake samples collected during the low engine load for Round 3 were also elevated, 7.7 (total copper) and 6.4 µg/L (dissolved). The Seawater Intake samples were collected from the EGCS system near the inlet point. This system has no source of copper associated with the seachest or water lines and the source of the elevated copper values in this one ship's sample set was not identified. The mean data for copper concentrations in the Seawater Intake samples are provided including these values as well as without. A discussion of the potential risk of metals associated with the waters is included in Section 4.1.

Table 3-5. Mean results for Water Metal Analyses by Engine Load

Metal (µg/L)	High Load						Low Load					
	SI		PD		OD		SI		PD		OD	
	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.
As (total)	1.66	0.10	1.82	0.11	1.75	0.14	1.65	0.17	1.68	0.26	1.65	0.19
As (dissolved)	1.67	0.14	1.74	0.14	1.66	0.12	1.49	0.21	1.67	0.18	1.67	0.21
Cd (total)	0.07	0.01	0.07	0.01	0.06	0.00	0.06	0.01	0.06	0.01	0.06	0.01
Cd (dissolved)	0.06	0.01	0.06	0.00	0.07	0.01	0.06	0.01	0.06	0.01	0.06	0.01
Cr (total)	0.48	0.20	1.09	0.76	0.49	0.21	0.35	0.05	1.03	1.16	0.43	0.27
Cr (dissolved)	0.13	0.01	0.69	0.58	0.22	0.07	0.13	0.01	0.88	0.84	0.23	0.20
Cr (VI) (total)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Cr (VI) (dissolved)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Cu (total)	5.44[a] (1.33)	10.15[a] (1.40)	3.63	3.93	1.33	1.25	2.29[a] (1.21)	2.79[a] (0.97)	3.09	3.55	1.02	0.49
Cu (dissolved)	4.04 (0.75)	8.09 (0.69)	2.01	3.03	1.38	1.24	1.69 (0.74)	2.37 (0.47)	2.32	2.37	0.84	0.38
Pb (total)	0.04	0.03	0.29	0.14	0.08	0.03	0.03	0.02	0.12	0.05	0.05	0.01
Pb (dissolved)	ND	ND	0.22	0.13	0.06	0.02	ND	ND	0.14	0.08	0.03	0.00
Hg (total)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Hg (dissolved)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Ni (total)	0.85	0.35	14.29	5.56	4.19	1.47	0.78	0.41	8.11	7.08	2.66	1.44
Ni (dissolved)	0.71	0.27	9.35	7.03	4.08	1.85	0.69	0.39	7.73	5.40	1.81	0.50
Tl (total)	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00
Tl (dissolved)	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00
V (total)	1.70	0.47	38.85	7.47	14.93	5.81	1.54	0.22	21.03	3.75	10.42	6.53
V (dissolved)	1.62	0.31	34.90	6.52	13.27	4.88	1.65	0.42	19.60	1.84	8.77	3.59
Zn (total)	0.86	0.65	2.32	0.89	1.29	0.93	0.87	0.68	1.15	0.43	0.79	0.33
Zn (dissolved)	0.68	0.75	1.60	0.83	1.62	1.21	0.42	0.14	1.16	0.55	0.94	0.90

[a] Seawater intake copper concentrations were unusually high for one sample. Ranges are reported which both include (upper) and exclude (lower) results shown to be statistical outlier.

S.D. – Standard deviation

3.3.3 Water PAHs

In an effort to simplify the presentation of the analytical results for the various waters, PAH data are presented as a sum of the measured values for each constituent based on the constituent's ring count (Figure 3-1and Figure 3-2). Individual data are available in Appendix D. Boxplots in these figures provide the median and 1st and 3rd quartiles. Non-detected analytes were given a value of half the detection limit for that constituent (Newman, Dixon, Looney, & Pinder, 1989). Results reported between the detection limit and the reporting limit (J-flagged) were included at reported values. Mean results for individual parameters are provided by engine load (Table 3-7) which also included non-detected values at 50 % of the detection limit. ANOVA testing was performed to compare the results between the high and low engine loads for each PAH category. All p-values were greater than 0.05 which indicates that there was not a statistically significant difference in the PAH profile between the low and high engine loads (Table 3-6).

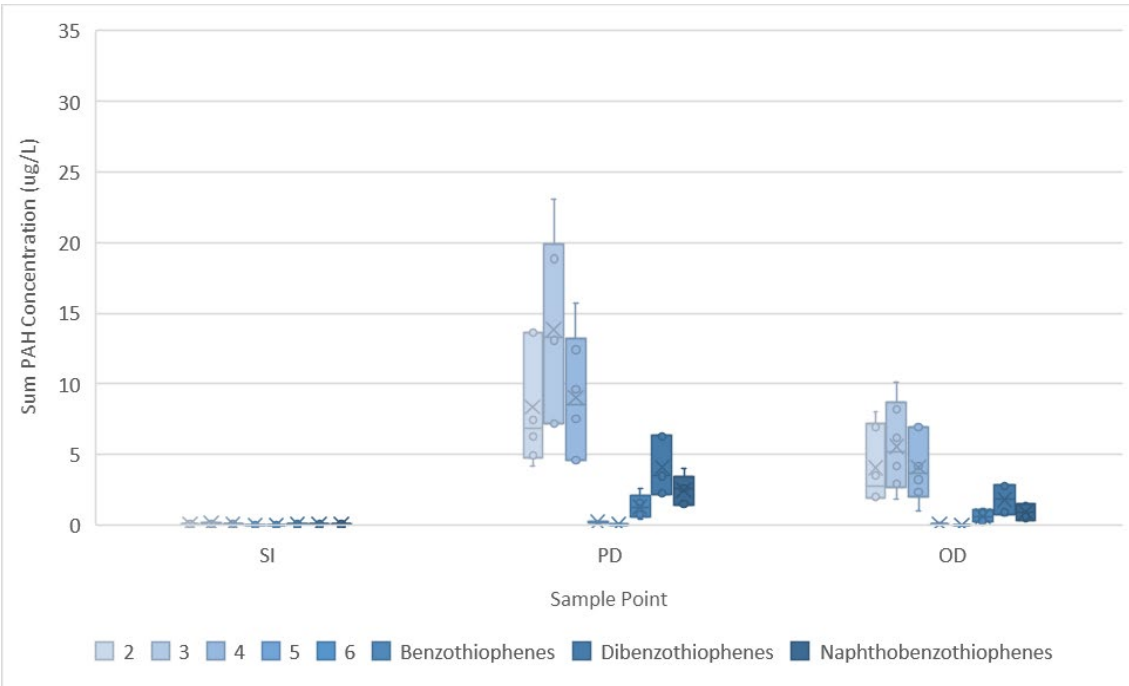

Figure 3-1. Sum PAHs by Benzene Ring Count, seawater intake (SI), post DeSOx (PD), overboard discharge (OD) (High Engine Load, n=6)

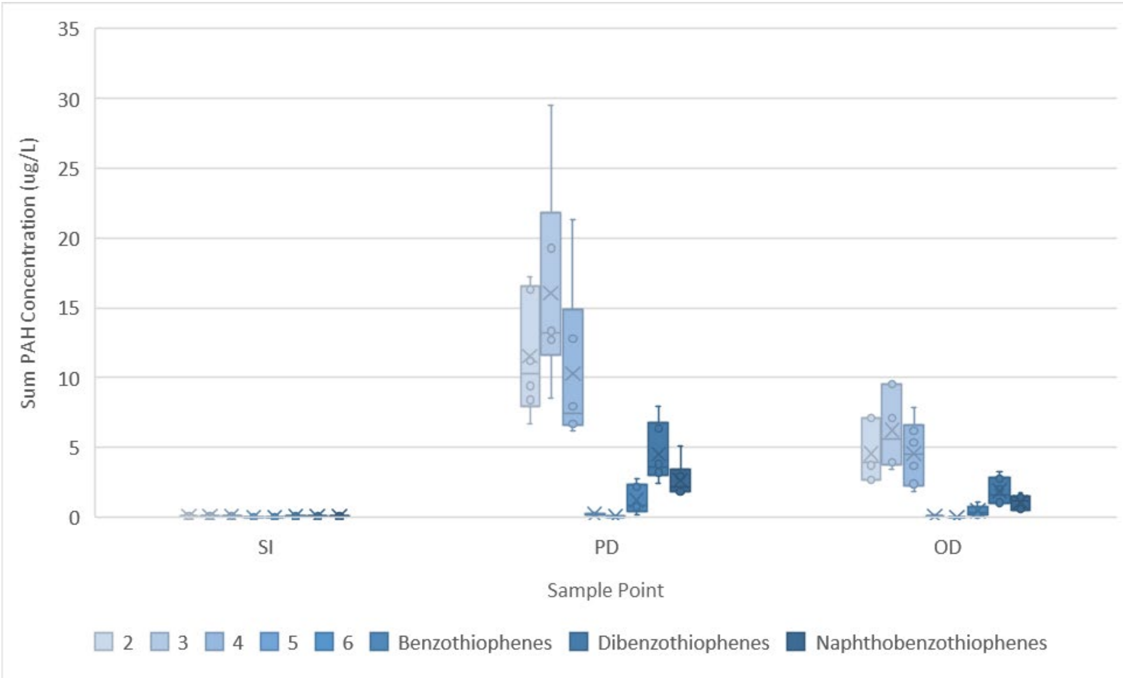

Figure 3-2. Sum PAHs by Benzene Ring Count, seawater intake (SI), post DeSOx (PD), overboard discharge (OD) (Low Engine Load, n=6)

Table 3-6. ANOVA Test Results (p-values)

PAH Comparison (High vs. low engine load)	SI	PD	OD
2-Ring	0.793	0.196	0.649
3-Ring	0.101	0.390	0.531
4-Ring	0.256	0.405	0.505
5-Ring	0.794	0.979	0.807
6-Ring	0.152	0.918	0.915
Benzothiophenes	0.945	0.823	0.750
Dibenzothiophenes	0.326	0.540	0.677
Naphthobenzothiophenes	0.942	0.863	0.824

Table 3-7. Mean results for Water PAH Analyses by Engine Load. Results presented as mean of six sample events (ND included at ½ MDL).

Mean PAH (µg/L) ± S.D.	High Load						Low Load					
	SI		PD		OD		SI		PD		OD	
	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.
Acenaphthene	0.001	0.000	0.102	0.062	0.057	0.050	0.001	0.000	0.145	0.076	0.058	0.040
Acenaphthylene	0.001	0.000	0.030	0.016	0.028	0.019	0.001	0.000	0.039	0.017	0.069	0.022
Anthracene	0.001	0.000	0.003	0.006	0.001	0.000	0.001	0.000	0.001	0.000	0.001	0.000
Benz(a)anthracene	0.002	0.001	0.090	0.060	0.061	0.036	0.001	0.000	0.114	0.082	0.076	0.034
Benzo(a)fluoranthene	0.001	0.000	0.001	0.000	0.001	0.000	0.001	0.000	0.001	0.000	0.001	0.000
Benzo(a)pyrene	0.001	0.000	0.020	0.014	0.010	0.007	0.001	0.000	0.020	0.015	0.011	0.008
Benzo(b)fluoranthene	0.001	0.000	0.049	0.016	0.019	0.011	0.001	0.000	0.051	0.018	0.021	0.009
Benzo(b)fluorene	0.001	0.000	0.086	0.048	0.046	0.026	0.001	0.000	0.111	0.068	0.049	0.024
Benzo(b)thiophene	0.001	0.000	0.138	0.097	0.044	0.034	0.001	0.000	0.155	0.076	0.046	0.023
Benzo(e)pyrene	0.001	0.000	0.103	0.037	0.039	0.024	0.001	0.000	0.101	0.043	0.042	0.019
Benzo(g,h,i)perylene	0.001	0.000	0.025	0.011	0.010	0.006	0.001	0.000	0.023	0.011	0.009	0.005
Benzo(k)fluoranthene	0.001	0.000	0.002	0.003	0.001	0.001	0.001	0.000	0.005	0.005	0.002	0.002
Biphenyl	0.001	0.000	0.248	0.084	0.077	0.036	0.001	0.000	0.283	0.090	0.086	0.022
C1 - Benzothiophenes	0.010	0.000	0.560	0.403	0.227	0.143	0.010	0.000	0.523	0.631	0.173	0.109
C1 - Chrysenes	0.010	0.000	1.090	0.505	0.465	0.263	0.010	0.000	1.167	0.630	0.505	0.249
C1 - Decalins	0.010	0.000	0.010	0.000	0.010	0.000	0.010	0.000	0.010	0.000	0.010	0.000
C1 - Dibenzothiophenes	0.010	0.000	0.695	0.404	0.275	0.189	0.010	0.000	0.807	0.447	0.305	0.190
C1 - Fluoranthenes/Pyrenes	0.012	0.005	0.962	0.564	0.553	0.321	0.010	0.000	1.298	0.879	0.623	0.318
C1 - Fluorenes	0.010	0.000	0.740	0.370	0.257	0.180	0.010	0.000	0.800	0.412	0.292	0.159
C1 - Naphthalenes	0.010	0.000	1.952	1.539	0.852	0.597	0.010	0.000	2.683	1.057	0.893	0.406
C1 - Naphthobenzothiophenes	0.010	0.000	0.717	0.277	0.292	0.140	0.010	0.000	0.747	0.336	0.297	0.136
C1 - Phenanthrenes/Anthracenes	0.010	0.000	2.150	0.954	0.878	0.483	0.010	0.000	2.533	1.058	0.978	0.428
C2 - Benzothiophenes	0.010	0.000	0.292	0.334	0.130	0.162	0.010	0.000	0.357	0.363	0.131	0.142
C2 - Chrysenes	0.010	0.000	1.138	0.526	0.477	0.292	0.010	0.000	1.210	0.647	0.508	0.279
C2 - Decalins	0.010	0.000	0.010	0.000	0.010	0.000	0.010	0.000	0.010	0.000	0.010	0.000
C2 - Dibenzothiophenes	0.010	0.000	1.043	0.506	0.387	0.299	0.010	0.000	1.168	0.567	0.423	0.308
C2 - Fluoranthenes/Pyrenes	0.015	0.012	1.602	0.784	0.765	0.459	0.010	0.000	1.933	1.140	0.860	0.422

Mean PAH (µg/L) ± S.D.	High Load						Low Load					
	SI		PD		OD		SI		PD		OD	
	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.
C2 - Fluorenes	0.010	0.000	0.995	0.522	0.392	0.248	0.010	0.000	1.175	0.559	0.447	0.236
C2 - Naphthalenes	0.010	0.000	2.750	1.461	1.258	0.904	0.010	0.000	3.583	1.551	1.393	0.722
C2 - Phenanthrenes/Anthracenes	0.023	0.032	2.450	1.099	1.105	0.572	0.010	0.000	2.867	1.288	1.225	0.494
C2-Naphthobenzothiophenes	0.010	0.000	0.817	0.306	0.337	0.171	0.010	0.000	0.833	0.402	0.352	0.168
C3 - Benzothiophenes	0.010	0.000	0.064	0.067	0.047	0.060	0.010	0.000	0.105	0.130	0.049	0.066
C3 - Chrysenes	0.010	0.000	0.727	0.392	0.304	0.189	0.010	0.000	0.747	0.462	0.317	0.170
C3 - Decalins	0.010	0.000	0.010	0.000	0.010	0.000	0.010	0.000	0.010	0.000	0.010	0.000
C3 - Dibenzothiophenes	0.010	0.000	1.090	0.459	0.477	0.227	0.010	0.000	1.150	0.641	0.507	0.250
C3 - Fluoranthenes/Pyrenes	0.010	0.000	1.418	0.718	0.587	0.355	0.010	0.000	1.537	0.920	0.663	0.363
C3 - Fluorenes	0.010	0.000	0.952	0.485	0.353	0.252	0.010	0.000	1.130	0.531	0.435	0.204
C3 - Naphthalenes	0.010	0.000	1.595	0.746	0.983	0.660	0.010	0.000	2.383	0.947	1.160	0.582
C3 - Naphthobenzothiophenes	0.010	0.000	0.530	0.217	0.218	0.140	0.010	0.000	0.557	0.284	0.236	0.128
C3 - Phenanthrenes/Anthracenes	0.022	0.030	2.617	1.301	1.097	0.662	0.010	0.000	3.000	1.905	1.203	0.641
C4 - Benzothiophenes	0.010	0.000	0.063	0.093	0.037	0.044	0.010	0.000	0.071	0.077	0.039	0.050
C4 - Chrysenes	0.010	0.000	0.415	0.292	0.162	0.171	0.010	0.000	0.422	0.342	0.158	0.165
C4 - Decalins	0.010	0.000	0.010	0.000	0.010	0.000	0.010	0.000	0.010	0.000	0.010	0.000
C4 - Dibenzothiophenes	0.032	0.053	0.862	0.312	0.457	0.150	0.010	0.000	0.897	0.410	0.485	0.157
C4 - Fluoranthenes/Pyrenes	0.010	0.000	0.750	0.374	0.301	0.178	0.010	0.000	0.812	0.462	0.352	0.186
C4 - Naphthalenes	0.010	0.000	0.692	0.385	0.480	0.373	0.010	0.000	1.228	0.644	0.595	0.312
C4 - Naphthobenzothiophenes	0.010	0.000	0.270	0.128	0.095	0.096	0.010	0.000	0.265	0.169	0.098	0.098
C4 - Phenanthrenes/Anthracenes	0.020	0.024	1.998	1.083	0.778	0.487	0.010	0.000	2.317	1.426	0.828	0.451
Chrysene	0.002	0.002	0.448	0.184	0.186	0.088	0.001	0.001	0.470	0.167	0.190	0.075
Decahydronaphthalene, Total	0.003	0.000	0.003	0.000	0.003	0.000	0.003	0.000	0.003	0.000	0.003	0.000
Dibenz(a,h)anthracene	0.001	0.000	0.017	0.007	0.005	0.005	0.001	0.000	0.015	0.007	0.007	0.005
Dibenzothiophene	0.001	0.001	0.448	0.306	0.153	0.116	0.001	0.000	0.502	0.322	0.153	0.099
Fluoranthene	0.002	0.001	0.109	0.036	0.037	0.025	0.001	0.000	0.143	0.030	0.058	0.016
Fluorene	0.002	0.002	0.403	0.231	0.130	0.091	0.001	0.000	0.440	0.209	0.148	0.075
Indeno(1,2,3-cd)pyrene	0.001	0.000	0.005	0.003	0.001	0.002	0.001	0.000	0.005	0.005	0.002	0.003

Mean PAH (µg/L) ± S.D.	**High Load**						**Low Load**					
	SI		**PD**		**OD**		**SI**		**PD**		**OD**	
	Mean	**S.D.**	**Mean**	**S.D.**	**Mean**	**S.D.**	**Mean**	**S.D.**	**Mean**	**S.D.**	**Mean**	**S.D.**
Naphthalene	0.003	0.003	1.083	0.455	0.345	0.207	0.002	0.000	1.317	0.248	0.375	0.085
Naphthobenzothiophene	0.001	0.000	0.232	0.079	0.093	0.041	0.001	0.000	0.238	0.086	0.097	0.036
Perylene	0.001	0.000	0.004	0.007	0.001	0.000	0.001	0.000	0.002	0.003	0.001	0.000
Phenanthrene	0.004	0.002	1.283	0.658	0.445	0.274	0.002	0.000	1.483	0.664	0.500	0.242
Pyrene	0.002	0.002	0.278	0.168	0.188	0.098	0.001	0.000	0.440	0.229	0.227	0.084
Average Sum of 60 PAHs (ND = ½ MDL)	0.454	0.109	39.245	17.213	17.044	9.682	0.378	0.001	46.453	19.707	18.801	8.377
Average Sum of 60 PAHs (ND = 0)	0.086	0.130	39.180	17.224	16.962	9.712	0.000	0.001	46.387	19.717	18.723	8.404

3.3.4 Summary of other Parameters

Additional parameters were analyzed, and the results are presented in Table 3-8. These parameters were used during the modeling activities as discussed in more detail in Section 4.3.

Table 3-8. Mean results for EGCS Associated-Water Parameters

Parameter (mg/L)	High Load						Low Load					
	SI		PD		OD		SI		PD		OD	
	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.
Dissolved Organic Carbon	0.18	0.06	0.15	0.05	0.15	0.08	0.26	0.01	0.17	0.11	0.18	NA
Particulate Organic Carbon	0.89	0.50	0.95	0.14	1.01	0.25	0.75	0.34	0.87	0.38	0.81	0.11
Chemical Oxygen Demand	451	337	527	325	531	333	531	385	490	310	555	374
Nitrate as Nitrogen	0.26	0.03	0.27	0.05	0.24	0.02	0.22	0.10	0.10	0.09	0.18	0.11
Nitrate + Nitrite as Nitrogen	0.26	0.03	0.28	0.03	0.27	0.03	0.22	0.10	0.18	0.13	0.20	0.12
Nitrite as Nitrogen	0.01	NA	0.03	0.03	0.04	0.01	ND	NA	0.11	0.03	0.03	0.01
pH*	7.8	0.1	5.1	0.74	6.3	0.18	7.9	0.2	5.1	0.27	6.4	0.08
Total Suspended Solids	6.0	1.4	ND	ND	ND	ND	6.5	0.7	ND	ND	ND	ND

* pH units; measured by compliance computer during sampling

3.4 Whole Effluent Toxicity Testing

WET tests were conducted for each round of EGCS water analysis. All individual tests passed control performance criteria (Table 3-9). Minimal toxicity was noted among the WET tests. Detailed results are included in individual laboratory reports in Appendix F. The evaluations of the WET test results were conducted by initially comparing the 10, 25, and 50 percent Effect Concentration (EC_{10}, EC_{25} and EC_{50}) and 95% Confidence Intervals obtained from tests conducted concurrently with individual EGCS discharge water samples (Table 3-10, Table 3-11, and Table 3-12). EC_{10}, EC_{25} and EC_{50} were calculated using the seawater intake water samples as the control. The *Skeletonema costatum* test was added to the sampling design on the advice of the peer-advisory group just prior to the collection effort in Round 1 and the sample was not received at the testing laboratory before the holding time expired thus, a *Skeletonema costatum* test was not conducted for Round 1.

Table 3-9. Control Acceptability Criteria Results

WET Species	Engine Load	Round 1	Round 2	Round 3	Round 4	Round 5	Round 6
Inland Silverside *(Menidia beryllina)*	High	Pass	Pass	Pass	Pass	Pass	Pass
	Low	Pass	Pass	Pass	Pass	Pass	Pass
Mysid (*Americamysis bahia*)	High	Pass	Pass	Pass	Pass	Pass	Pass
	Low	Pass	Pass	Pass	Pass	Pass	Pass
Sand dollar (*Dendraster excentricus*)	High	Pass	Pass	Pass	Pass	Pass	Pass
	Low	Pass	Pass	Pass	Pass	Pass	Pass
Algae *(Skeletonema costatum*)	High		Pass	Pass	Pass	Pass	Pass
	Low		Pass	Pass	Pass	Pass	Pass

Menidia beryllina – Control: ≥ 80% survival; ≥ 0.50 mg growth
Americamysis bahia – Control: ≥ 80% survival; ≥ 0.20 mg growth
Dendraster excentricus – Control: ≥ 80% Normal development; MSD <25%
Skeletonema costatum – Control: Cell density Increased by factor of more than 16 in 72 hours; CV specific growth rate < 7%; pH change is < 1.0

When the EC_{10} results were compared across concurrent tests with a common sample, the most sensitive endpoint was consistently the biomass endpoint for inland silverside and/or the combined proportion normal endpoint for the sand dollar. The survival EC_{10} and confidence interval for inland silverside during Round 1 also indicated that it was a sensitive endpoint; however, the mean control survival for this test was 100 percent, while the lowest mean survival of any test treatment concentration was 87.5 % (25 and 100 percent treatments). The lack of mortality in the control resulted in a minor reduction from the treatments being identified as significantly different which resulted in the lower EC_{10}.

Biomass for the mysid was the most sensitive endpoint for the high engine load in Round 6 and the low engine load in Round 1. In both cases, the mean biomass in the seawater intake sample was statistically significantly higher than the laboratory control waters, therefore controls could not be combined for subsequent statistical analyses. When the test treatments were compared to the laboratory control, no statistically significant effect was observed.

EC_{25} results were all >100% concentration for bioassays conducted on samples collected during the high engine load scenario except for the combined proportion normal endpoint of the sand dollar test. EC_{25}s calculated for this endpoint ranged from 52.9 to 59.6 for rounds 1, 2, 4, and 5. Rounds 3 and Round 6 resulted in all EC_{25} >100% including the combined proportion normal endpoint. EC_{25} for the low engine load samples were all >100% except for inland silverside biomass during Round 2 and inland silverside biomass and sand dollar combined proportion normal for Round 5.

Only three tests resulted in responses where an EC_{50} could be calculated. These results were for the combined proportion normal endpoint for the sand dollar test for rounds 1, 2 and 4 from the high engine load scenario.

Table 3-10. WET Point Estimate Results (EC10) – Using Seawater Intake results.

EGCS Water Collection Round	Inland Silverside				Mysid				Sand dollar				Algae	
	Survival	CI	Biomass	CI	Survival	CI	Biomass	CI	Proportion Survived	CI	Combined Proportion Normal	CI	Growth Rate	CI
EC_{10} (% effluent) – High Engine Load														
1	**21.8**	10.2 – 146.9	**71.4**	39.7 - NA	>100	NA	>100	NA	>100	NA	**52.9**	NA – 56.4		
2	>100	NA	**29.3**	14.1 - 66.4	>100	NA	>100	NA	>100	NA	**53.8**	51.5 – 55.4	>100[a]	NA
3	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100[a]	NA
4	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	**54.3**	49.0 – 57.2	>100[a]	NA
5	>100	NA	**55.9**	NA – 97.4	>100	NA	>100	NA	>100	NA	**59.6**	52.4 – 67.0	>100[a]	NA
6	>100	NA	>100	NA	>100	NA	**3.0**[b]	1.6 - NA	>100	NA	>100	NA	>100[a]	NA
EC_{10} (% effluent) – Low Engine Load														
1	>100	NA	87.8	14.1 - NA	>100	NA	**1.6**[b]	1.1 – 3.2	>100	NA	66.0	50.3 – 101.6		
2	>100	NA	**24.4**	14.4 - 47.5	>100	NA	>100	NA	>100	NA	89.0	74.9 - NA	>100[a]	NA
3	>100	NA	**62.6**	NA – 82.1	>100	NA	>100	NA	>100	NA	>100	NA	>100[a]	NA
4	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100[a]	NA
5	>100	NA	**19.0**	12.7 – 27.3	>100	NA	>100	NA	>100	NA	**55.3**	2.1 – 65.8	>100[a]	NA
6	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100[a]	NA

CI – Confidence Interval

BOLD – Most sensitive endpoint. Determined by 95% confidence intervals.

[a] – EC10s were not calculated, EC 15s provided.

[b] – laboratory control was significantly less than Seawater intake sample – Comparison testing indicated no significant differences between any of the EGCS water concentrations and the laboratory control

Table 3-11. WET Point Estimate Results (EC_{25}) – Using Seawater Intake results.

EGCS Water Collection Round	Inland Silverside				Mysid				Sand dollar				Algae	
	Survival	CI	Biomass	CI	Survival	CI	Biomass	CI	Proportion Survived	CI	Combined Proportion Normal	CI	Growth Rate	CI
EC_{25} (% effluent) – High Engine Load														
1	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	**61.3**	55.0 – 65.1		
2	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	**61.6**	59.2 – 64.0	>100	NA
3	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA
4	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	**64.8**	58.3 – 68.7	>100	NA
5	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	**81.0**	69.5 – 107.5	>100	NA
6	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA
EC_{25} (% effluent) – Low Engine Load														
1	>100	NA	>100	NA	>100	NA	**78.5**[b]	5.2 - NA	>100	NA	>100	NA		
2	>100	NA	**70.0**	54.1 – 85.2	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA
3	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA
4	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA
5	>100	NA	**95.8**	41.4 - NA	>100	NA	>100	NA	>100	NA	**84.9**	65.7 - NA	>100	NA
6	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA

CI – Confidence Interval

BOLD – Most sensitive endpoint. Determined by 95% confidence intervals.

b – laboratory control was significantly less than Seawater intake sample – Comparison testing indicated no significant differences between any of the EGCS water concentrations and the laboratory control

Table 3-12. WET Point Estimate Results (EC_{50}) – Using Seawater Intake results.

EGCS Water Collection Round	Inland Silverside				Mysid				Sand dollar				Algae	
	Survival	CI	Biomass	CI	Survival	CI	Biomass	CI	Proportion Survived	CI	Combined Proportion Normal	CI	Growth Rate	CI
EC_{50} (% effluent) – High Engine Load														
1	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	**78.4**	72.6 – 83.3		
2	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	**77.2**	72.4 – 83.1	>100	NA
3	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA
4	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	**86.7**	76.6 – 94.8	>100	NA
5	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA
6	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA
EC_{50} (% effluent) – Low Engine Load														
1	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA		
2	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA
3	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA
4	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA
5	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA
6	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA	>100	NA

CI – Confidence Interval

BOLD – Most sensitive endpoint. Determined by 95% confidence intervals.

b – laboratory control was significantly less than Seawater intake sample – Comparison testing indicated no significant differences between any of the EGCS water concentrations and the laboratory control

The highest toxic unit (defined as 100/EC_{xx} concentration) results for each round of WET testing are provided (Table 3-13 and Table 3-14). Chronic toxic units ranged from <1.0 to 3.4 (based on EC_{10}) or <1.0 to 1.6 (based on EC_{25}).

Two results were removed from the toxic unit calculations for the high engine load and one from the low engine load based on concerns for the validity of the calculated EC_{10}. The EC_{10} for Inland Silverside survival from Round 1 was calculated at 21.8% which would result in 4.6 toxic units. This result was removed; however, because the Seawater Intake control survival was 100% and the lowest test concentration survival was 87.5% in the 25% and 100% discharged water treatments. Even though an EC_{10} was calculable for this test, the result was deemed a statistical anomaly and not representative of the toxicity of the sample.

The toxic units for mysid biomass from Round 1 (low load) and Round 6 (high load) were also removed. These tests resulted in an EC_{10} of 1.6% (low load) and 3.0% (high load) which resulted in toxic unit calculations of 64.3 and 33.6, respectfully. The Seawater Intake biomass results for these rounds were significantly better than lab controls for these tests. Seawater intake biomass results were also significantly higher than all PD and OD treatments, however, the lab control was not significantly different than any dilution and resulted in NOECs of 100. The results suggest the potential for beneficial constituents in these 2 seawater intake samples that may not be present throughout the EGCS process.   Since the laboratory controls were not significantly different from the PD and OD treatments, we do not attribute this to a toxic response.

Table 3-13. Highest Toxic Unit Calculation for WET Tests (High Engine load)

EGCS Water Collection Round	High Engine Load					
	Toxic Units (100/EC_{10})	WET Species/Endpoint	Toxic Units (100/EC_{25})	WET Species/Endpoint	Toxic Units (100/EC_{50})	WET Species/Endpoint
1	1.4[a]	Inland Silverside / Biomass	1.6	Sand Dollar / Combined proportion normal	1.3	Sand Dollar / Combined proportion normal
2	3.4	Inland Silverside / Biomass	1.6	Sand Dollar / Combined proportion normal	1.3	Sand Dollar / Combined proportion normal
3	1.8	Marine Diatom / Growth Rate	<1.0	All species	<1.0	All species
4	1.8	Sand Dollar / Combined proportion normal	1.5	Sand Dollar / Combined proportion normal	1.2	Sand Dollar / Combined proportion normal
5	1.8	Inland Silverside / Biomass	1.2	Sand Dollar / Combined proportion normal	<1.0	All species
6	<1.0[a]	All species	<1.0	All species	<1.0	All species

[a] – TU (4.6) for Inland Silverside survival was removed. Mean survival in Seawater Intake treatment was 100% while lowest mean survival of test treatment was 87.5% in 25% and 100% test concentrations.
[b] - TU (33.6) for Mysid biomass removed. Seawater intake biomass result was significantly better than lab control. Seawater intake biomass results were significantly different than all overboard discharge treatments. The lab control was not significantly different than any dilution and resulted in NOEC of 100, LOEC >100

Table 3-14. Highest Toxic Unit Calculation for WET Tests (Low Engine load)

EGCS Water Collection Round	Low Engine Load					
	Toxic Units ($100/EC_{10}$)	WET Species/Endpoint	Toxic Units ($100/EC_{25}$)	WET Species/Endpoint	Toxic Units ($100/EC_{50}$)	WET Species/Endpoint
1	1.5[a]	Sand Dollar / Combined proportion normal	1.3	Mysid / Biomass	<1.0	All species
2	4.1	Inland Silverside / Biomass	1.4	Inland Silverside / Biomass	<1.0	All species
3	2.4	Marine Diatom / Growth Rate	<1.0	All species	<1.0	All species
4	<1.0	All species	<1.0	All species	<1.0	All species
5	5.3	Inland Silverside / Biomass	1.2	Sand Dollar / Combined proportion normal	<1.0	All species
6	<1.0	All species	<1.0	All species	<1.0	All species

[a] TU (64.3) for Mysid biomass removed. Seawater intake biomass result was significantly better than lab control. Seawater intake biomass results were significantly different than all the overboard discharge treatments. The lab control was not significantly different than any dilution and resulted in NOEC of 100, LOEC >100

4. RISK CHARACTERIZATION

The goal of this study was to predict the potential impacts of EGCS waters discharged to the Puget Sound. A tiered assessment strategy was used to assess both PEC and PNEC concentrations for possible contaminants of concern (COCs). PECs were developed first through the analytical determination of COCs (PAHs and metals) concentrations in EGCS waters, followed by stochastic modelling predictions of post-discharge concentrations. PNECs were estimated using a combination of modelling (metals assessment and TLM) to first predict potential environmental impacts, followed by empirical determinations of toxicity (WET testing) to validate those predictions. The characterization of potential risks of overboard discharged waters will be discussed here (Section 4.1 for metals and Sections 4.2 for PAHs) as well as a characterization of EGCS waters once discharged using dilution models such as MAMPEC and CORMIX (Section 4.3).

4.1 Metals Risk Assessment

A three-step approach was used to evaluate the effects of metals contained in overboard discharge waters from EGCS (Figure 4-1). The first step was to compare the metals concentrations observed in the seawater intake sample to the overboard discharge samples. If ambient concentrations were not significantly different from the concentrations in the overboard discharge, the increased risk from the metals contained in the discharge of buffered EGCS waters is expected to be negligible. The dissolved metals fraction was used for this comparison and metals that were not detected were evaluated at half the detection limit.

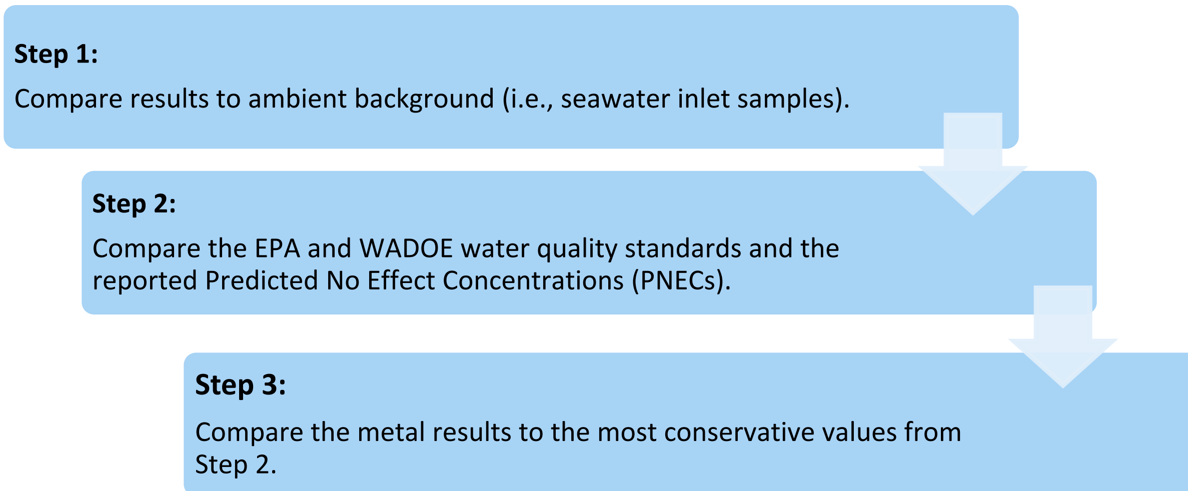

Figure 4-1. Metals assessment approach used in the evaluation of overboard discharge waters.

Figure 4-2 graphically presents the mean data for the metals that were eliminated from further analysis, i.e., SI water was not statistically significantly elevated over OD waters. For this comparison, significance was determined by comparing the mean values from the SI sample to the OD sample results using a standard Student's T-test. Dissolved metal concentrations that were unchanged across all three sampling points were arsenic, cadmium, mercury, selenium, thallium, and zinc. Copper and chromium spike in the Post DeSOx sample but return to ambient concentrations in the buffered Overboard Discharge samples. One copper result was elevated in the Seawater Intake sample during the high engine load for Round 3. A Grubbs test (Grubbs, 1969) was performed on the dissolved copper results for the high and low engine load using the online calculator by Graphpad (Graphpad, 2023). Results from this test are provided in Table 4-1. Outliers from round 3 were removed from the mean for the SI samples. No dissolved copper results associated with the PD and OD samples were removed from the mean data.

Table 4-1. Grubbs Test for Outliers (Dissolved Copper)

Sample Collection Round	High Engine Load			Low Engine Load		
	Dissolved Copper (µg/L)	Z	Outlier (Y/N)	Dissolved Copper (µg/L)	Z	Outlier (Y/N)
1	1.94	0.260	N	1.55	0.058	N
2	0.31	0.462	N	0.61	0.455	N
3	20.50	2.04	Y	6.44	2.01	Y
4	0.47	0.442	N	0.38	0.552	N
5	0.30	0.463	N	0.49	0.506	N
6	0.73	0.410	N	0.65	0.438	N

ANOVA tests were performed on dissolved metal concentrations for each metal (Table 4-2) using "R" software (R Core Team, 2021).When a significant difference was noted in the ANOVA test, an additional post-hoc test was conducted using TukeyHSD. Significant differences were noted between the SI and PD samples for chromium, lead, nickel, thallium, and vanadium (high load and low load) which indicated that these metals are present in the PD samples at greater concentrations than in the ambient waters (i.e., they were increased in the EGCS process). Additionally, significant differences were noted between the PD and OD samples for lead, vanadium (high and low engine load) and nickel (low engine load) samples, however, the comparison between the SI and OD results indicated no significant differences for lead and nickel. This indicates that even though lead and nickel increase after the DeSOx process, dissolved metal concentrations for these metals are at background concentrations in the discharged water. Vanadium was the only metal that showed a statistically significant increase between the SI and OD samples.

Table 4-2. ANOVA P-value results (Group SI, PD, OD)

Dissolved Metal	High Engine Load		Low Engine Load	
	p-Value	Significant Post- Hoc Comparison	p-Value	Significant Post- Hoc Comparison
Arsenic	0.527	NA	0.242	NA
Cadmium	0.451	NA	0.950	NA
Chromium	**0.024**	SI v. PD	**0.040**	SI v. PD
Copper	0.592	NA	0.435	NA
Lead	**0.006**	SI v. PD; PD v. OD	**0.001**	SI v. PD; PD v. OD
Mercury	0.397	NA	0.397	NA
Nickel	**0.010**	SI v. PD	**0.003**	SI v. PD; PD v. OD
Selenium	0.561	NA	0.391	NA
Thallium	**0.010**	SI v. PD	**0.028**	PD v. OD
Vanadium	**< 0.000**	SI v. PD; PD v. OD; SI v. OD	**< 0.000**	SI v. PD; PD v. OD; SI v. OD
Zinc	0.186	NA	0.104	NA

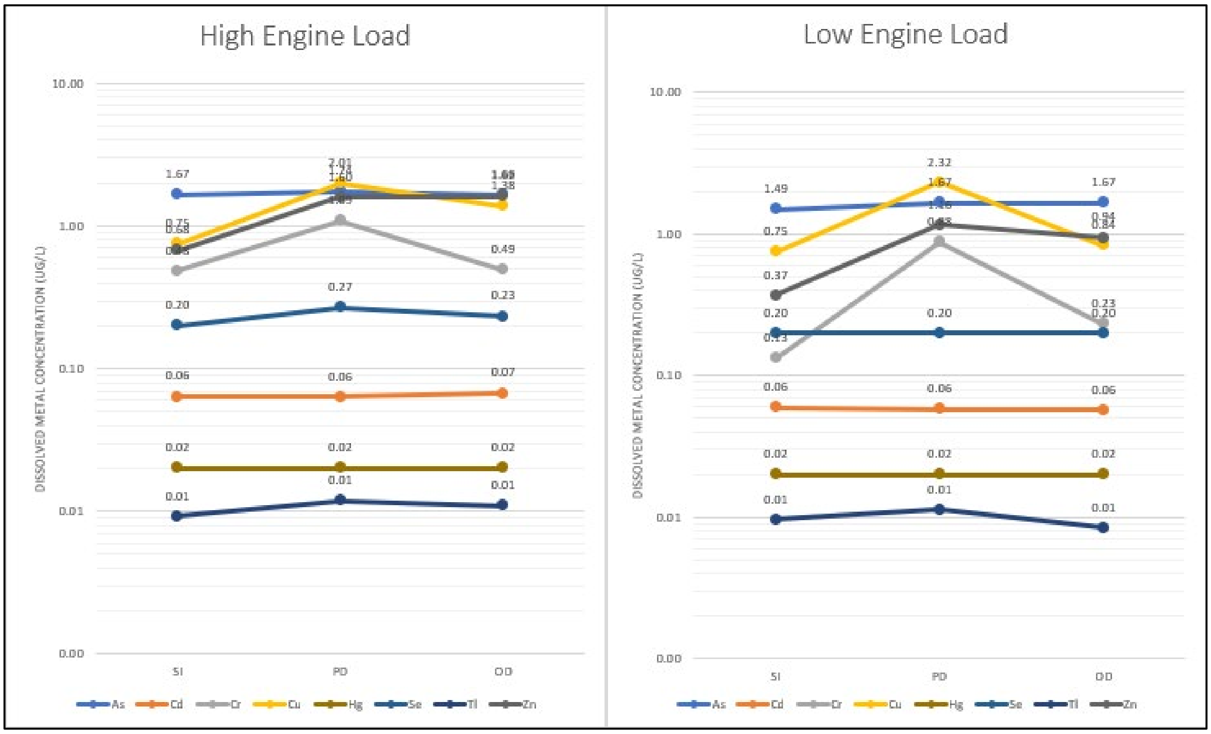

Figure 4-2. Metals eliminated from further analysis.

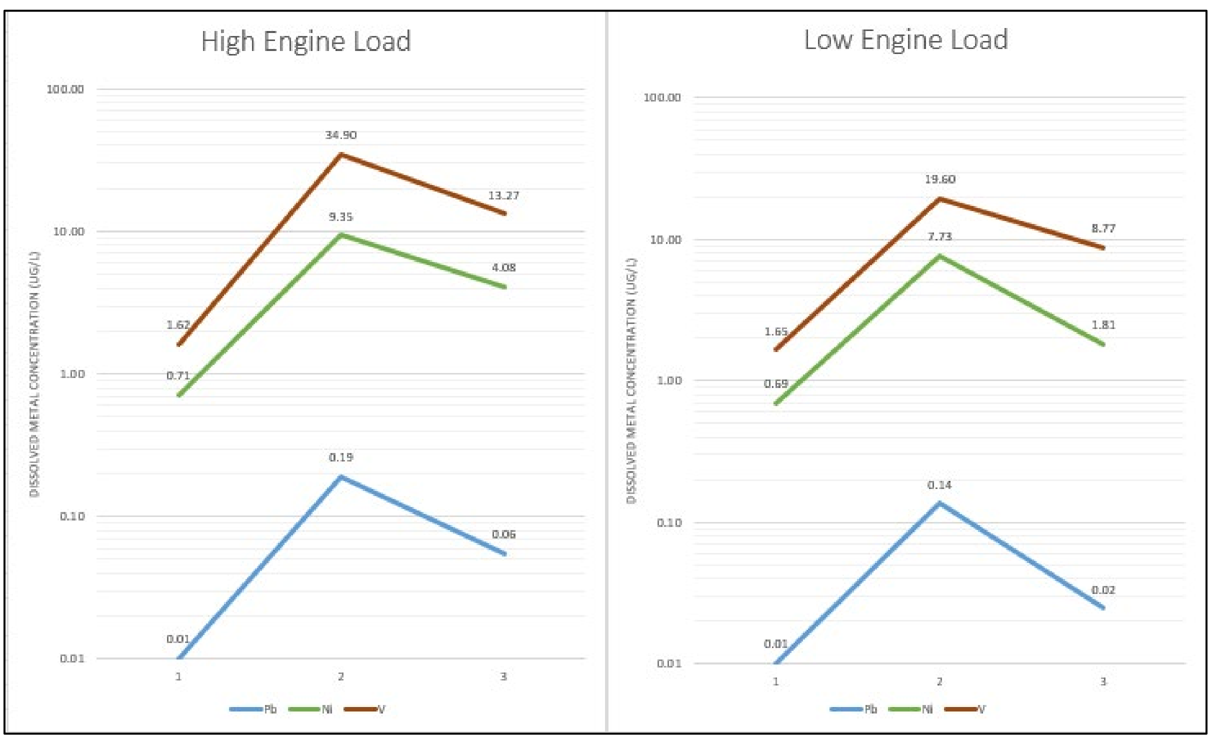

Figure 4-3. Metals elevated above background.

The next step in the evaluation was to compare measured metals concentrations to appropriate water quality criteria/standards (Table 4-3). WDOE has developed surface water quality standards which are used to "protect surface waters for public health and public enjoyment of the waters and the propagation and protection of fish, shellfish, and wildlife, pursuant to the provisions of chapter 90.48 RCW" (WAC 173-201A-010). WDOE aims to protect surface water designated uses that are defined as; aquatic life uses, recreational uses, water supply uses, miscellaneous uses. The surface water quality standards provide numeric values for specific chemicals that are intended to protect aquatic life from toxic effects. Aquatic life criteria/standards for metals are developed based on acute (short-term) and chronic (long-term) laboratory exposure data. The acute criterion assumes that adequate protection is achieved provided that the ambient surface water's metal concentration does not exceed the standard based on a 1-hour average concentration more than once every three years on average. The chronic criterion assumes the standard is protective provided that the waterborne chemical concentration does not exceed the numerical standard based on a 4-day average concentration that is exceeded more than once every three years on average (WAC 173-201A-240). The Washington State Standards are generally based on U.S. EPA Ambient Water Quality Criteria and were developed for specific metals in marine waters; they are provided in Table 4-3. Washington State Standards were available for nickel; however, no State Standard or U.S. EPA Ambient Water Quality Criteria (AWQC) exists for vanadium.

PNECs are European-based values that are similar to U.S. EPA AWQC but are generally more "conservative" based on the methods used to derive the values. Marine Predicted No Effect Concentrations (PNEC) (European Communities, 2003) have been developed for each of the metals measured in this study except for chromium (III). PNECs are concentrations of a metal that would not be expected to cause any toxic effect due to long term exposures and have been developed using predominately chronic laboratory toxicity data for marine organisms. Depending on the quantity and quality of the available data, a Species Sensitivity Distribution (SSD) or an Assessment Factor based approach can be used to derive the PNEC.

In all cases, the marine PNEC values were less than the EPA AWQC and Washington State aquatic life Standard, except for mercury. The Washington State marine chronic aquatic life criterion for mercury is 0.025 µg/L while the reported PNEC is 0.067 µg/L. Measured dissolved metal concentrations in the overboard discharge and post DeSOx samples were compared to the conservative PNEC values, except for mercury which was compared to the Washington state standard. The average vanadium concentration was the only metal that exceeded the PNEC[4] value in the overboard discharge samples for the high and/or low engine load (Table 4-4). Although nickel concentrations were not statistically elevated over SI values, it was identified as a major constituent

[4] The marine vanadium PNEC was developed based on a limited data set, i.e., three studies, and relies heavily on one study which determined an effect concentration with oyster larvae (*C. gigas*) of 50 µg/L (which was the lowest test concentration). Since this study was "unbounded," (the test was insufficient to define a "No Effect Concentration") the researchers took a conservative approach and used half this concentration as the base of the PNEC and then applied an assessment factor of 10 which resulted in a PNEC of 2.5 µg/L. In addition, the study would not meet today's standards of acceptability because no analytical confirmation of vanadium content in exposure solutions was made during the test. The approach employed in developing the PNEC may be overly conservative, nonetheless, the data are all that is available. The two other studies used to develop this PNEC were a fish (*L. limanda*) which had an LC50 of 27,800 µg/L and a mysid (*A. bahia*) which had an LC50 of 13,300 µg/L.

in the fuel samples. And although the average nickel concentrations in the overboard discharge waters were not statistically elevated over intake water concentrations, they were higher than the European PNEC values, but were lower than both the EPA and WDOE ambient water quality criteria.

Mean metal concentrations for the post DeSOx samples are also provided for information about sample constituent concentrations prior to buffering and overboard discharge. Direct discharge of the post DeSOx water is not allowed because there is a regulatory pH standard (≥ 6.0) that must be met prior to discharge. All overboard discharge is buffered with seawater prior to release. Vanadium will be further addressed as a key constituent during the modeling effort.

Table 4-3. Reported Marine PNECs, US Federal Metals Criteria and Washington State Water Quality Standards.

Metal	Reported Marine European Predicted No Effect Concentrations (PNEC)			U.S. EPA (National Recommended Aquatic Life Criteria) µg/l		Washington State (Aquatic Life Standard - Marine) µg/l	
	PNEC (Marine)	Extrapolation Method	Reference	CMC - Saltwater	CCC - Saltwater	Acute	Chronic
Arsenic	4.7 µg/L	Sensitivity Distribution	(ECHA, n.d.)	69	36	69.0 (a, c, d)	36.0 (b, c, d, e)
Cadmium	1.14 µg/L	Sensitivity Distribution	(ECHA, n.d.)	33	7.9	42.0 (a, d)	9.3 (b, d)
Chromium (III)	-	-	-	-	-	-	-
Chromium (VI)	0.6 µg/L	Application factor	(Maycock, Sorokin, Atkinson, Rule, & Crane, 2007)	1,100	50	1,100.0 (a, c, d, f)	50.0 (b, c, d)
Copper	2.6 µg/L	Sensitivity Distribution	(EU, 2008)	4.8	3.1	4.8 (a, c, d)	3.1 (b, c, d)
Lead	11.0 µg/L	Sensitivity Distribution	(Church, Van Sprang, Chowdhury, & DeForest, 2017)	210	8.1	210.0 (a, c, d)	8.1 (b, c, d)
Mercury	0.067 µg/L	Sensitivity Distribution	(ECHA, n.d.)	1.8	0.94	1.8 (a, c, d)	0.025 (b, g, h)
Nickel	3.9 µg/L	Sensitivity Distribution	(De Forest & Schlekat, 2013)	74	8.2	74.0 (a, c, d)	8.2 (b, c, d)
Thallium	0.8 µg/L	Application factor	(ECCC, 2020)	-	-	-	-
Vanadium	2.5 µg/L	Application factor	(ECHA, n.d.)	-	-	-	-
Zinc	3.4 µg/L	Sensitivity Distribution	(Maycock, Peters, Merrington, & Crane, 2010)	90	81	90.0 (a, c, d)	81.0 (b, c, d)

a - A 1-hour average concentration not to be exceeded more than once every three years on the average.
b - A 4-day average concentration not to be exceeded more than once every three years on the average.
c - Marine conversion factors (CF) which were used for calculating dissolved metals concentrations are given below. Conversion factors are applicable to both acute and chronic criteria for all metals except mercury. The CF for mercury was applied to the acute criterion only and is not applicable to the chronic criterion. Conversion factors are already incorporated into the criteria in the table. Dissolved criterion = criterion x CF.
d - These ambient criteria in the table are for the dissolved fraction. The cyanide criteria are based on the weak acid dissociable method. The metals criteria may not be used to calculate total recoverable effluent limits unless the seasonal partitioning of the dissolved to total metals in the ambient water are known. When this information is

absent, these metals criteria shall be applied as total recoverable values, determined by back-calculation, using the conversion factors incorporated in the criterion equations. Metals criteria may be adjusted on a site-specific basis when data are made available to the department clearly demonstrating the effective use of the water effects ratio approach established by U.S. EPA, as generally guided by the procedures in U.S. EPA Water Quality Standards Handbook, December 1983, as supplemented or replaced by U.S. EPA or ecology. The adjusted site-specific criteria are not in effect until they have been incorporated into this chapter and approved by EPA. Information which is used to develop effluent limits based on applying metals partitioning studies or the water effects ratio approach shall be identified in the permit fact sheet developed pursuant to WAC 173-220-060 or 173-226-110, as appropriate, and shall be made available for the public comment period required pursuant to WAC 173-220-050 or 173-226-130(3), as appropriate. Ecology has developed supplemental guidance for conducting water effect ratio studies.
e - Nonlethal effects (growth, C-14 uptake, and chlorophyll production) to diatoms (*Thalassiosira aestivalis* and *Skeletonema costatum*) which are common to Washington's waters have been noted at levels below the established criteria. The importance of these effects to the diatom populations and the aquatic system is sufficiently in question to persuade the state to adopt the U.S. EPA National Criteria value (36 µg/L) as the state threshold criteria, however, wherever practical the ambient concentrations should not be allowed to exceed a chronic marine concentration of 21 µg/L.
f - Salinity dependent effects. At low salinity the 1-hour average may not be sufficiently protective.
g - These criteria are based on the total-recoverable fraction of the metal.
h - If the four-day average chronic concentration is exceeded more than once in a three-year period, the edible portion of the consumed species should be analyzed. Said edible tissue concentrations shall not be allowed to exceed 1.0 mg/kg of methylmercury.

Table 4-4. Average measured metal concentrations compared to PNEC and Washington State Chronic Aquatic Life Standard.

Metal (µg/L, Dissolved)	High Load		Low Load		PNEC	Exceedance of PNEC[1]	Washington State Chronic Aquatic Life Standard µg/l	Exceedance of Washington State Chronic Standard[1]
	PD	OD	PD	OD				
Arsenic	1.82 ± 0.11	1.75 ± 0.14	1.68 ± 0.26	1.65 ± 0.19	**4.7**	-	36.0	-
Cadmium	0.06 ± 0.00	0.07 ± 0.01	0.06 ± 0.01	0.06 ± 0.01	**1.14**	-	9.3	-
Chromium	0.69 ± 0.58	0.22 ± 0.07	0.88 ± 0.84	0.23 ± 0.20	**0.6**	-	50.0	-
Copper	2.01 ± 3.03	1.38 ± 1.24	2.32 ± 2.37	0.84 ± 0.38	**2.6**	-	3.1	-
Lead	0.22 ± 0.13	0.06 ± 0.02	0.14 ± 0.08	0.03 ± 0.00	11.0	-	**8.1**	-
Mercury	ND	ND	ND	ND	0.067	-	**0.025**	-
Nickel	9.35 ± 7.03	4.08 ± 1.85	7.73 ± 5.40	1.81 ± 0.50	**3.9**	+	8.2	-
Thallium	0.01 ± 0.00	0.01 ± 0.00	0.01 ± 0.00	0.01 ± 0.00	**0.8**	-	NA	NA
Vanadium	34.90 ± 6.52	13.27 ± 4.88	19.60 ± 1.84	8.77 ± 3.59	**2.5**	+	NA	NA
Zinc	1.60 ± 0.83	1.62 ± 1.21	1.16 ± 0.55	0.94 ± 0.90	**3.4**	-	81.0	-

[1] Exceedance when compared to overboard discharge (OD) average concentration.

Bold – Indicates most conservative value.

ND – non-detected in lab analyses

4.2 Target Lipid Model (PAH Toxic Unit Estimate)

The acute and chronic total toxic units estimated from the PAH analysis for each sample are provided in Table 4-5. Excluding samples SPL-OD-High-080922, SPL-OD-Low-080922, and SHP6-OD-Low-090522, the acute total toxic units were above 1.0, with a range of 1.1 to 3.1. The chronic toxic units were all above 1.0 ranging from 2.8 to 16. A toxic unit above 1 suggests there is a potential toxicological concern from exposure to the concentrations of PAHs in the buffered overboard discharge waters. The toxic units were developed based on the 5th percentile (most sensitive) organisms in the TLM distribution curve. However, it is not uncommon for a 5th percentile threshold to be exceeded with no discernable ecological effect, as the threshold does not represent the concentration at which effects are likely to occur. Rather, it is intended to represent the concentration below which effects are highly unlikely. Conservative thresholds such as the 5th percentile concentrations (HC5) are typically used in developing regulatory criteria/standards to establish concentrations that, if not exceeded, should prove “safe” to exposed organisms. These values are also frequently used as toxic thresholds in conducting screening level risk assessments to identify contaminates of potential concern. The exposure scenario embedded in the derivation of the 5th percentile concentration is extremely conservative with respect to application to a discharge plume such as the EGCS buffered overboard discharge water, as the threshold is based on a continuous exposure of a species over an extended period. In reality, the buffered overboard discharge water will be subjected to dispersion and biological and physical fate processes (e.g., biodegradation, photolysis) which will reduce the exposure concentrations. There were limited effects observed in the WET bioassays conducted for this project, suggesting that the test species, which are representative of the those found in the Puget Sound, may be less sensitive than the some of the organisms included in the derivation of the TLM Species Sensitivity Distribution (SSD). The WET test species included *Americamysis bahia*, *Menidia beryllina*, *Skeletonema costatum* and *Dendraster excentricus*. The species-specific TLM CTLBBs for the first three species are higher than the 5th percentile CTLBB in the TLM SSD, ranging from 34.7 to 248 µmol/g octanol, confirming that they are less sensitive than the 5th percentile species (McGrath, et al., 2018). The computed toxic units would be lower for these specific species and acute toxicity most likely would not be predicted. A CTLBB is not available for *D. excentricus*; however, single-species laboratory testing with this species indicates that it would typically be the most sensitive of the species used in the WET tests to PAHs. The echinoderm development endpoint has also been shown to be a sensitive life stage when testing with single PAHs (Bellas, Saco-Alvarez, Nieto, & Beiras, 2008).

The PAH fingerprint, based on contribution percentage to the TU summation is similar across sampling rounds and is typical for petroleum sources such that the parent PAHs are present at lower concentrations then the corresponding alkylated parent compounds (Figure 4-4). For example, naphthalene is present at lower concentrations than C1-, C2-, C3-, and C4-naphthalenes. In all samples, the parent PAHs have a minor contribution to the total toxic units ranging from 4.6 to 6.9%. The alkylated homologs of the parent PAHs contribute 93.1 to 95.4% of the toxic units. The distribution of the toxic units from the alkylated PAH homologs is similar across sampling rounds. The alkylated phenanthrene/anthracene homolog series contributed the largest percentage of the toxic units ranging from 25.6 to 32.7%. The alkylated dibenzothiophene, fluoranthene/pyrene and naphthobenzothiophene homolog series each contribute approximately to 20% (actual range is 9.8 to 21.9%). The alkylated decalin and benzothiophene homolog series have a minor toxic unit contribution of less than 1%. A report including all the inputs and results for this assessment are included in Appendix E.

Table 4-5. Summary of acute and chronic toxic units predicted from TLM.

Sampling Round	Sample Identification	Acute Toxic Units	Chronic Toxic Units
High Engine Load			
1	Eurodam-OD-High-072322	2.1	11
2	XP-OD-High-073122	3.1	16
3	SPL-OD-High-080922	0.54	2.8
4	KP-OD-High-082022	1.1	5.5
5	WEDM-OD-High	2.7	14
6	SHP6-OD-High-090522	1.4	7.2
Low Engine Load			
1	Eurodam-OD-Low-072322	2.7	14
2	XP-OD-Low-073122	2.9	15
3	SPL-OD-Low-080922	1.0	5.0
4	KP-OD-Low-082022	1.4	7.5
5	WEDM-OD-Low	3.0	15
6	SHP6-OD-Low-090522	1.0	5.4

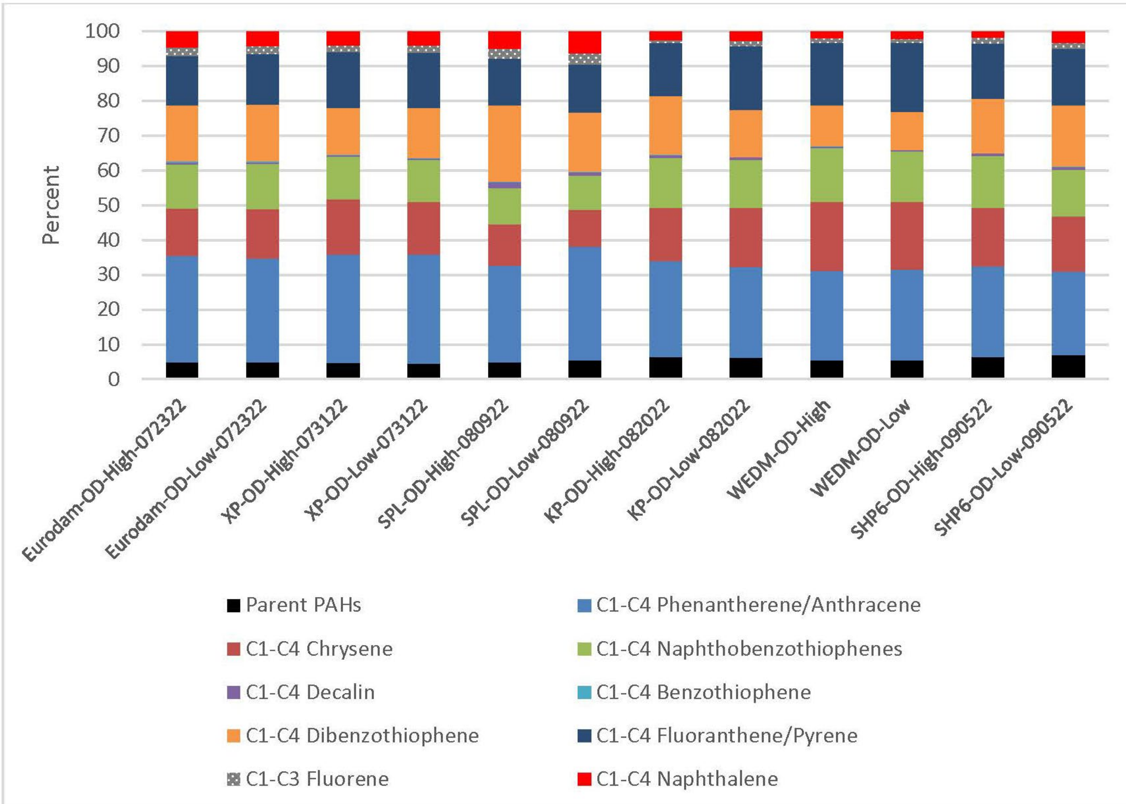

Figure 4-4. Distribution of toxic units in each sample

4.2.1 TLM Conclusions

The TLM and toxic unit framework were applied to predict the toxicity of buffered overboard discharge waters collected from six sampling rounds. The chemicals included hydrocarbons and cyclic aromatics typically associated with petroleum. The acute and chronic TLM thresholds were based on the 5th percentile of a species sensitivity distribution of CTLBBs using U.S. EPA methods for deriving water quality criteria. With the exception of three samples out of twelve, the toxic units were above 1.0 suggesting that toxicological effects may occur from exposure to the concentrations of petroleum hydrocarbons in the buffered overboard discharge water. These acute and chronic thresholds are conservative values representative of concentrations that would affect only the most sensitive species. The species-specific CTLBBs for three test species used in the bioassays indicate that they are less sensitive than the 5th percentile species, and application of the 5th percentile thresholds to the bioassay results will result in an overly conservative estimate of effects.

These results suggest that chemistry-based toxicity predictions (e.g., TLM) may overstate cumulative risks when compared to observed results from laboratory-based empirical toxicity tests. Several reasons may exist for these discrepancies including: 1) uncertainty in characterizing the toxicity assessments used in model predictions, 2) environmental fate processes may decrease exposures under lab and field conditions thus reducing observed toxicity, and 3) possible interactions among effluent constituents may be over-estimated in the TLM model due to the assumption of response additivity.

4.3 Predicted Environmental Concentrations

4.3.1 MAMPEC Model

This section discusses the MAMPEC modeling results, which have been divided into two sections; operation while at berth in port under low engine loads (4.3.1.1) and while underway in Puget Sound under high engine loads (4.3.1.2). Discussion of the contribution of a single ship and operations of multiple ships (similar to what occurs during the cruise ship operational calendar) are provided. A report including all the model inputs and results are included in Appendix G.

4.3.1.1 MAMPEC – Operation while in Port of Seattle

The Estuarine Harbour model is influenced by the extent of the area included in the assessment. Figure 2-4 provides the overall area that was used by the MAMPEC Estuarine model for estimating EGCS discharged water influence in the Port of Seattle. The extent of the area was restricted to Pier 91 area to get a conservative estimation. Individual grid cells were 65 x 70 meters. Model estimates are based on one ship in port with the EGCS running continuously for 24 hours a day every day for the entire year. However, for the Carnival Corporation ships participating in this study for CLIA, traffic at this location would be 6 ships in port each week of the 5.5-month cruise season, for 8-10 hours per ship, with two ships in port on each weekend day, for a total for 60 hours per week. This is approximately 15% of the in-port EGCS operating time assumed in the Study, before accounting for the time that visiting ships are typically on shore power (about 50% of the port time in the 2022 season). Background concentrations were assessed based on the results from the Seawater Intake samples collected during the low engine load scenario. It should be noted that these samples were collected from the Puget Sound waters during the transit from Victoria to Seattle and are likely conservative estimates of actual concentrations in ambient waters of the port.

Table 4-6 presents a statistical summary of the simulated incremental concentration due solely to the ship effluent (assuming zero background concentration) of key constituents in the Port of Seattle, as well as the corresponding mean measured discharge and background concentrations. The maximum modeled concentration in Table 4-6 most likely occurred in the grid cells adjacent to the discharge while the mean modeled concentration is the average concentration over the 100 grid cells (10 by 10, see Section 2.8.1.1). As shown by these results, the incremental changes in the concentrations of the receiving environment due to ship at berth were less than the background concentrations by two to four orders of magnitude. These low modeled values indicate that it would be unlikely for the ship's EGCS discharge to cause a substantial build-up of constituent concentrations in the Port of Seattle, even with the conservative assumptions of a continuous discharge and limited removal mechanisms from the water.

Table 4-6. Mean measured discharge and ambient concentrations and the statistics for modeled incremental concentrations in the Port of Seattle.

Key Constituents	Measured Discharge Conc. (µg/L)	Measured Background Conc. (µg/L)[1]	Modeled Incremental Conc. (µg/L) from Discharge Alone ^				
			Max.	95th Percentile	Mean	Median	Min
Vanadium	10.4	1.7	0.259	0.259	0.150	0.150	0.039
Nickel	2.66	0.78	0.066	0.066	0.038	0.038	0.010
C1-phenanthrene/ anthracenes	0.98	**0.02**	0.023	0.023	0.013	0.013	0.003
C2-phenanthrene/ anthracenes	1.23	**0.02**	0.029	0.029	0.017	0.017	0.004
C3-phenanthrene/ anthracenes	1.2	**0.02**	0.029	0.029	0.017	0.017	0.004
C4-phenanthrene/ anthracenes	0.83	**0.02**	0.020	0.020	0.012	0.012	0.003

Bold values are detection limits, which were used when the measured concentrations were below the detection limit (ND).
^ The incremental concentrations are modeled in the absence of any background concentrations.
[1] Measured background concentrations are based on the results of the Seawater Intake samples which were collected from the Puget Sound while in route to Seattle.

Table 4-7 presents the simulated receiving water concentrations within the Port of Seattle computed using Equation 4. Note that more significant figures were reported for modeled concentrations compared to the measured values in the table to show differences in concentrations. The maximum calculated receiving water concentration in Table 4-7 most likely occurred in the grid cells adjacent to the discharge while the mean calculated receiving water concentration is the average concentration over the 100 grid cells (10 by 10, see Section 2.8.1.1). The MAMPEC results estimate a dilution factor of 40 under this scenario.

Table 4-7. Mean measured discharge and ambient concentrations and the statistics for calculated receiving water concentrations in the Port of Seattle.

Key Constituents	Measured Discharge Conc. (µg/L)	Measured Background Conc. (µg/L)[1]	Calculated Receiving Water Conc. (µg/L)				
			Max.	95th Percentile	Mean	Median	Min
Vanadium	10.4	1.7	1.9166	1.9166	1.8252	1.8252	1.7327
Nickel	2.66	0.78	0.8268	0.8268	0.8071	0.8071	0.7871
C1-phenanthrene/ anthracenes	0.98	**0.02**	0.0429	0.0429	0.0331	0.0331	0.0234
C2-phenanthrene/ anthracenes	1.23	**0.02**	0.0485	0.0485	0.0364	0.0363	0.0242
C3-phenanthrene/ anthracenes	1.2	**0.02**	0.0485	0.0485	0.0364	0.0363	0.0242
C4-phenanthrene/ anthracenes	0.83	**0.02**	0.0397	0.0397	0.0313	0.0313	0.0229

Bold values are detection limits, which were assumed when the measured concentrations were below the detection limit (ND).
[1] Measured background concentrations are based on the results of the Seawater Intake samples which were collected from the Puget Sound while in route to Seattle.

While Table 4-7 summarized the simulated concentrations based on the average sample discharge concentrations for the selected analytes, in order to assess a worst-case scenario specific to metals a second MAMPEC model was run using the highest single sample concentrations for Vanadium and Nickel. Table 4-8 provides the results from this additional modeling. The modeled results show a slightly higher maximum concentrations in the receiving waters, although still below the most conservative PNEC concentration, 2.5 µg/L (Vanadium) and 3.9 µg/L (Nickel) (Table 4-4).

Table 4-8. Maximum measured discharge and the statistics for calculated receiving water concentrations in the Port of Seattle.

Key Constituents	Measured Maximum Discharge Conc. (µg/L) *	Measured Background Conc. (µg/L)	Modeled Incremental Conc. (µg/L) from Discharge Alone ^				
			Max.	95%	Mean	Median	Min
Vanadium	23.3	1.7	0.58	0.58	0.34	0.34	0.09
Nickel	5.52	0.78	0.14	0.14	0.08	0.08	0.02

* Measured total discharge concentrations are used to be conservative because the total discharge concentrations for the three constituents are higher than dissolved discharge concentrations.
^ The incremental concentrations are modeled in the absence of any background concentrations.

4.3.1.2 MAMPEC – Shipping Lanes in Puget Sound

Table 4-9 presents the simulated incremental concentrations due solely to the ship effluent (assuming zero background concentration) for the simulated constituents in the shipping lane of Puget Sound. These results describe the distribution of concentrations in the vicinity (in the 20 m by 10 m by 4 m body of water) of the discharge corresponding to acute exposures. The maximum modeled concentration in Table 4-9 most likely occurred in the 1-by-1-by-4 m grid cells adjacent to the discharge while the mean modeled concentration is the average concentration over the 200 cells (20 by 10 cells, see Section 2.8.1.2) in the model domain. Based on the modeled incremental concentrations due to the discharge of effluent, the maximum fraction of discharge water in the model domain was 0.49%, corresponding to a dilution factor of 205.

Table 4-9. Mean measured discharge and ambient concentrations and the statistics for modeled incremental concentrations in the shipping lane.

Key Constituents	Measured Discharge Conc. (µg/L)	Measured Background Conc. (µg/L)	Modeled Incremental Conc. (µg/L) from Discharge Alone ^				
			Max.	95%	Mean	Median	Min
Vanadium	14.9	1.6	0.0728	0.0652	0.0086	0	0
Nickel	4.19	0.85	0.0205	0.0183	0.0024	0	0
C1-phenanthrene/ anthracenes	0.88	**0.02**	0.0043	0.0039	0.0005	0	0
C2-phenanthrene/ anthracenes	1.11	0.03	0.0054	0.0049	0.0006	0	0
C3-phenanthrene/ anthracenes	1.1	0.03	0.0054	0.0048	0.0006	0	0
C4-phenanthrene/ anthracenes	0.78	0.03	0.0038	0.0034	0.0004	0	0

Note the zero values for median and minimum modeled concentrations occur within the model domain ahead (upstream) of the point of discharge.
Bold values are detection limits, which were used when the measured concentrations were below the detection limit (ND).
^ The incremental concentrations are modeled in the absence of any background concentrations.

Table 4-10 presents the simulated receiving water concentrations within Puget Sound computed using Equation 4 from Section 2.8.1.1. Note that more significant figures were reported for modeled concentrations compared to the measured values to show differences in concentrations. Background concentrations were assessed based on the results from the Seawater Intake samples collected during the high engine load scenario. The maximum calculated receiving water concentration in Table 4-10 most likely occurred in the 1-by-1-by-4 m grid cells adjacent to the discharge while the mean calculated receiving water concentration is the average concentration over the 200 grid cells (20 by 10, see Section 2.8.1.2). Table 4-10 indicates the maximum difference between the background and modeled receiving water concentrations was less than 17%, which most likely[5] occurred in the grid cells adjacent to the discharge. However, when averaged over the 20 m by 10 m by 4 m model domain representing the limited region near the discharge, the mean differences between the background and modeled receiving water concentration were less than 2% for all constituents of interest.

[5] The spatial distribution of the simulated concentrations are not provided in the model outputs.

Table 4-10. Mean measured discharge and ambient concentrations and the statistics for calculated receiving water concentrations constituents in the shipping lane.

Key Constituents	Measured Discharge Conc. (µg/L)	Measured Background Conc. (µg/L)	Calculated Conc. (ug/L) Based on Relative Fraction of Discharge Water				
			Max.	95th Percentile	Mean	Median	Min
Vanadium	14.9	1.6	1.6650	1.6582	1.6076	1.6000	1.6000
Nickel	4.19	0.85	0.8663	0.8646	0.8519	0.8500	0.8500
C1-phenanthrene/ anthracenes	0.88	**0.02**	0.0242	0.0238	0.0205	0.0200	0.0200
C2-phenanthrene/ anthracenes	1.11	0.03	0.0353	0.0347	0.0306	0.0300	0.0300
C3-phenanthrene/ anthracenes	1.1	0.03	0.0352	0.0347	0.0306	0.0300	0.0300
C4-phenanthrene/ anthracenes	0.78	0.03	0.0337	0.0333	0.0304	0.0300	0.0300

Bold values are detection limits, which were used when the measured concentrations were below the detection limit (ND).

4.3.1.3 Summary of MAMPEC Results

Based on the MAMPEC simulation results, the PECs of key constituents in the Port of Seattle (Pier 91 area) and in the vicinity of ship discharges in Puget Sound due to the discharge of EGCS waters from a single ship will be reduced by a factor of 40. In the Port, the concentrations are not expected to accumulate over time due to the high hydrodynamic exchange rate (flushing) with Puget Sound. In the shipping lanes of Puget Sound, the effluent from the ship is rapidly diluted within a 1 m by 1 m by 4 m body of water immediately adjacent to the discharge to levels of more than 200:1. Dilution increases (and receiving water concentrations decrease) as a function of the relative speed between the moving ships and ambient water.

While the MAMPEC model provides an integrated tool for analyzing the chemical fate and transport of various constituents on a site-specific basis, there exist a number of considerations for the model which may impact the results of this analysis:

- The two modules of MAMPEC used assume a fixed number of grid cells. The Estuarine Harbour module assumes 10 by 10 grid cells while the Shipping Lane module assumes a 20 by 10 grid. Therefore, the size of each cell scales with the size of the model domain and would be large (less refined) when the domain is large. The cell size in the Estuarine Harbour model for studying Port of Seattle (Pier 91 area) is 65 m × 70 m. This level of spatial resolution may not be sufficient to describe the concentrations immediately adjacent to the discharge. Therefore, for simulating maximum concentrations in the vicinity of ship at berth in Port, which would occur during slack tide, we have also used a model capable of resolving concentration distributions immediately adjacent to the discharge and is able to simulate dilution under tidal actions, such as Visual Plumes (CORMIX Model).

- Limited options exist for modifying the geometry describing the model domain, thus requiring simplifications of the topography in the simulated environments.
- The 2-D spatial structure of the model output is not provided or is difficult to access for different modules. Therefore, the spatial distribution of the concentration values cannot be assessed to determine the extent of the region impacted by the discharges.
- Differences were observed between the receiving water concentration computed by the model and the receiving water concentrations calculated using principles of mass conservation. For example, the MAMPEC model generated simulated receiving water concentrations of copper that were slightly higher than either the discharge or receiving water concentrations used as model input, which may have occurred because the discharge concentrations of copper in the effluent were less than the ambient concentrations. This may indicate a non-conservative treatment of scalars and source terms within the numerical scheme of the model.
- The MAMPEC model simplifies hydrodynamic processes. Various processes related to mixing and dilution in the nearfield of the discharge plume do not appear to be resolved by the model based on the lack of relevant input parameters. As these processes can influence the maximum values and spatial distributions of the modeled concentrations, we have also used the CORMIX Model to provide additional dynamic detail.

4.3.2 CORMIX Model

This section discusses the CORMIX modeling results, which have been divided into two sections; operation while at berth in Port of Seattle (Pier 91) during a slack tide scenario (4.3.2.1) and operation at berth in Port during a neap tide scenario (4.3.2.2). A report including all the model inputs and results are included in Appendix H. All CORMIX estimates for dilutions are based on the centerline of the discharge. Additional dilution of this estimate is expected to occur perpendicular to this centerline. Background concentrations were assessed based on the results from the Seawater Intake samples collected during the low engine load scenario. It should be noted that these samples were collected from Puget Sound waters during the transit from Victoria to Seattle and are likely conservative estimates of actual concentrations in ambient waters of the port.

4.3.2.1 Slack Tide Scenario

The slack tide scenario is a conservative estimation of dilution of the EGCS discharge waters which assumes no lateral flow or flushing. Figure 4-5 presents a plan view of the spatial distribution of the dilution factor for water in the vicinity of the discharge under slack tide conditions. The nearfield region (NFR)'s limits are located within the mixing zone, which extends up to 60 m away from the discharge, and a centerline dilution factor of 16.3 is estimated at the edge of the nearfield region, located 51.5 m away from the discharge (Figure 4-6). This suggests that the discharged water dilution outside of the mixing zone will likely be greater than a factor of 16.3, which corresponds to a maximum concentration of 6% water in the receiving waters (Figure 4-7).

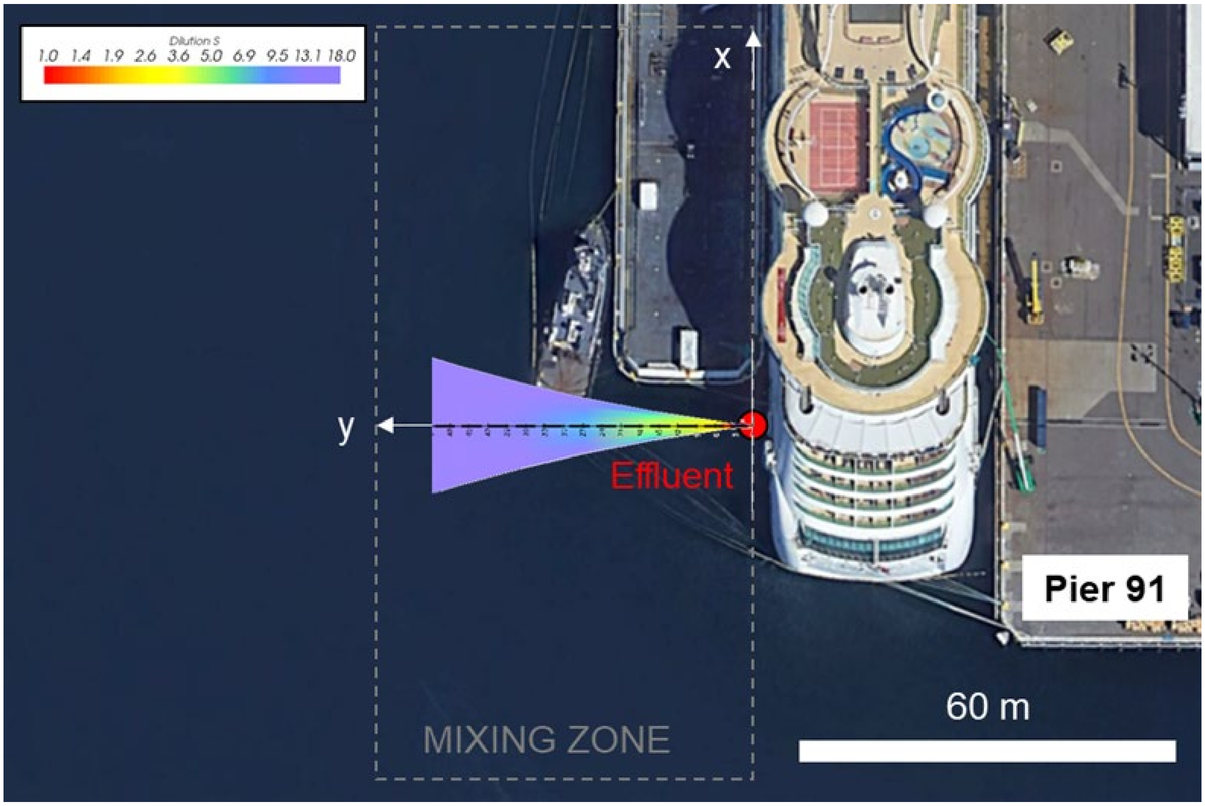

Figure 4-5. Spatial distribution of discharged water dilution under slack tide conditions overlaid on Google Earth satellite imagery near Pier 91 of the Smith Cove Cruise Terminal. Note the vessel depicted in the satellite imagery represents a generic cruise ship and the exact discharge location may vary depending on the ship.

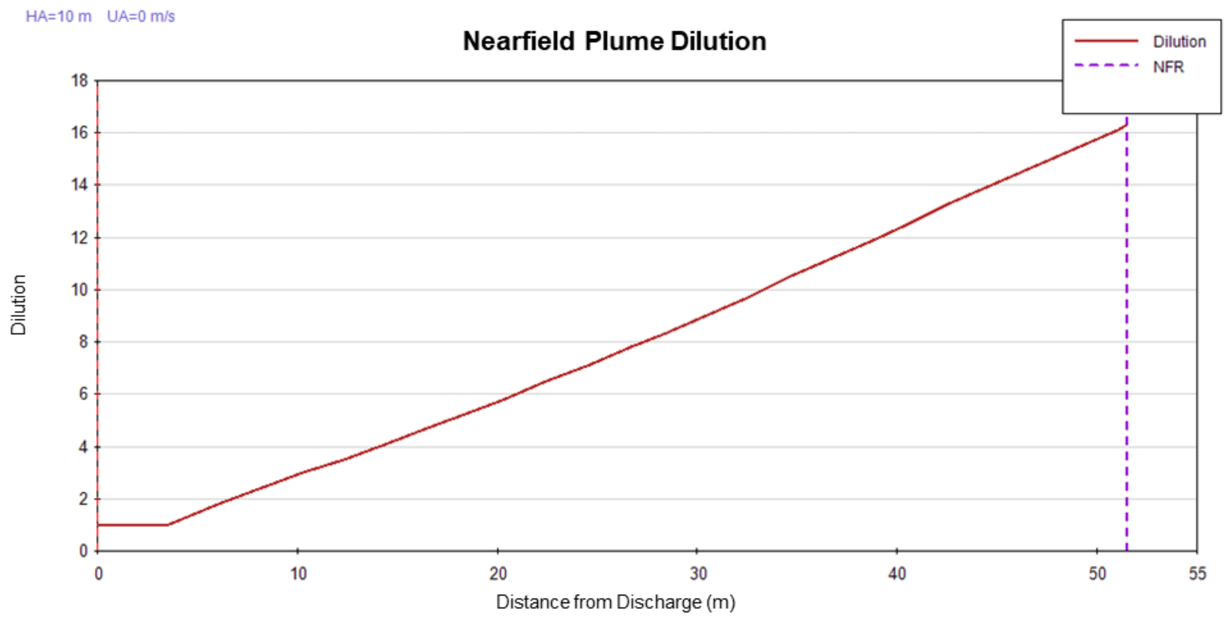

Figure 4-6. Discharge water dilution along the centerline of the plume versus distance from discharge (y) under slack tide conditions. The edge of the nearfield region is indicated by the dotted purple line.

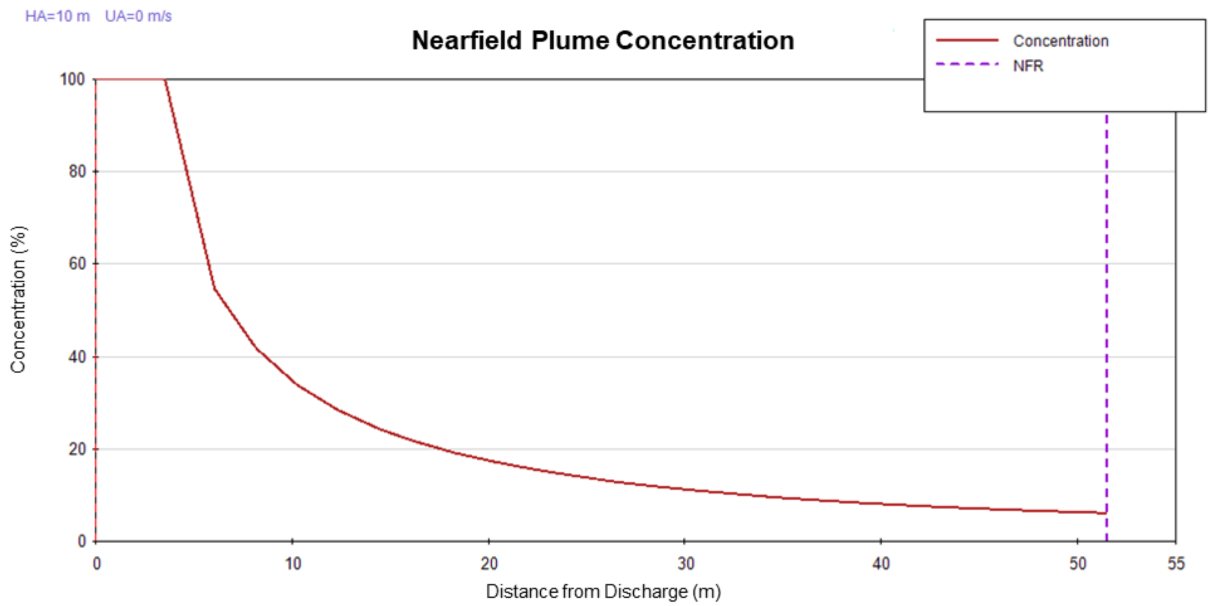

Figure 4-7. Fractional concentration of discharge water along the centerline of the plume versus distance from discharge (y) under slack tide conditions. The edge of the nearfield region is indicated by the dotted purple line.

4.3.2.2 Neap Tide Scenario

In addition to the conservative scenario of minimal dilution at slack tide, simulations were also conducted to assess water concentrations one hour after slack tide, corresponding to a period of weak crossflow (u_a = 0.1 m/s). This scenario is more representative of the typical ambient conditions in the receiving waters, as slack tide only accounts for a small fraction of the tidal period. Stronger tidal currents are also typically present in this environment.

Figure 4-8 displays the spatial distribution of the dilution factor for EGCS water in the vicinity of the discharge during the tidal cycle. The dilution factors were found to be greater than those during slack tide, with a dilution factor of 24.1 achieved at the edge of the nearfield region located 57 m downstream from the discharge (Figure 4-9). This suggests that the discharge water dilution outside of the mixing zone will likely be greater than a factor of 24.1, which corresponds to a maximum concentration of 4% EGCS water in the receiving waters (Figure 4-10).

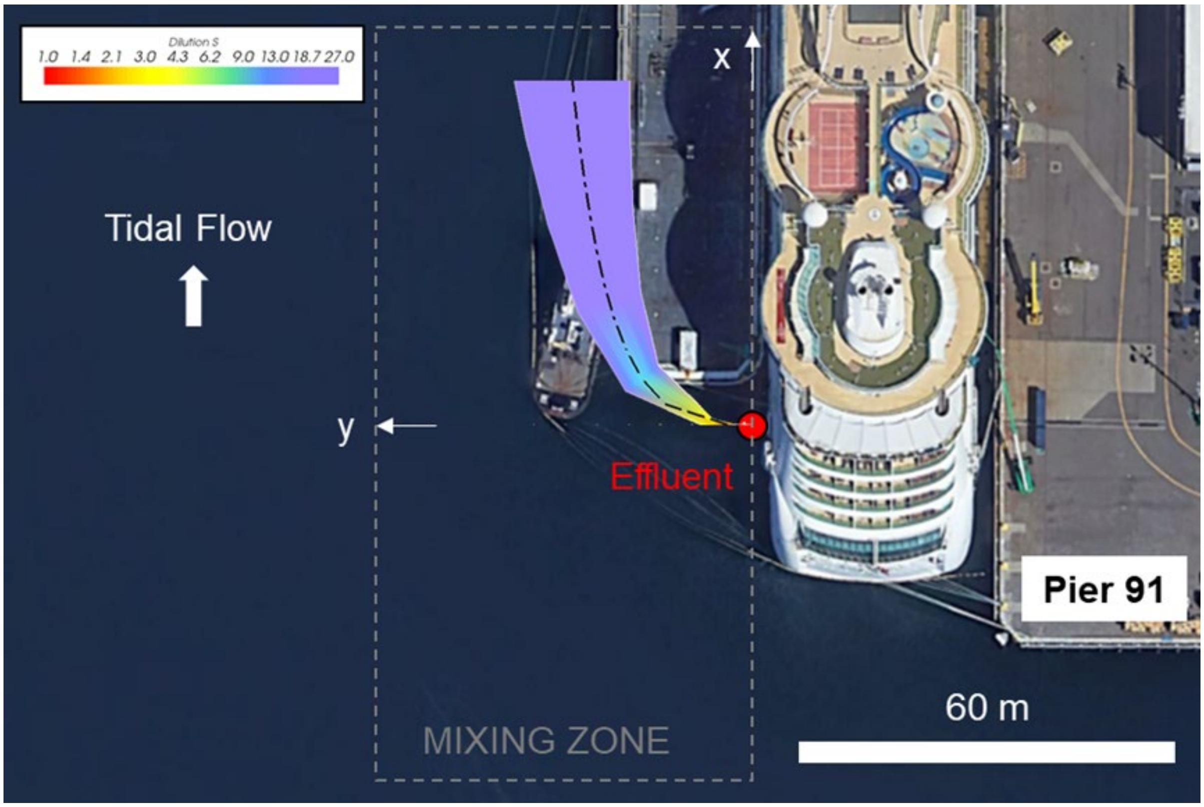

Figure 4-8. Spatial distribution of discharge water dilution one hour after slack tide overlaid on Google Earth satellite imagery near Pier 91 of the Smith Cove Cruise Terminal. Note the vessel depicted in the satellite imagery represents a generic cruise ship and the exact discharge location may vary dependent on the ship.

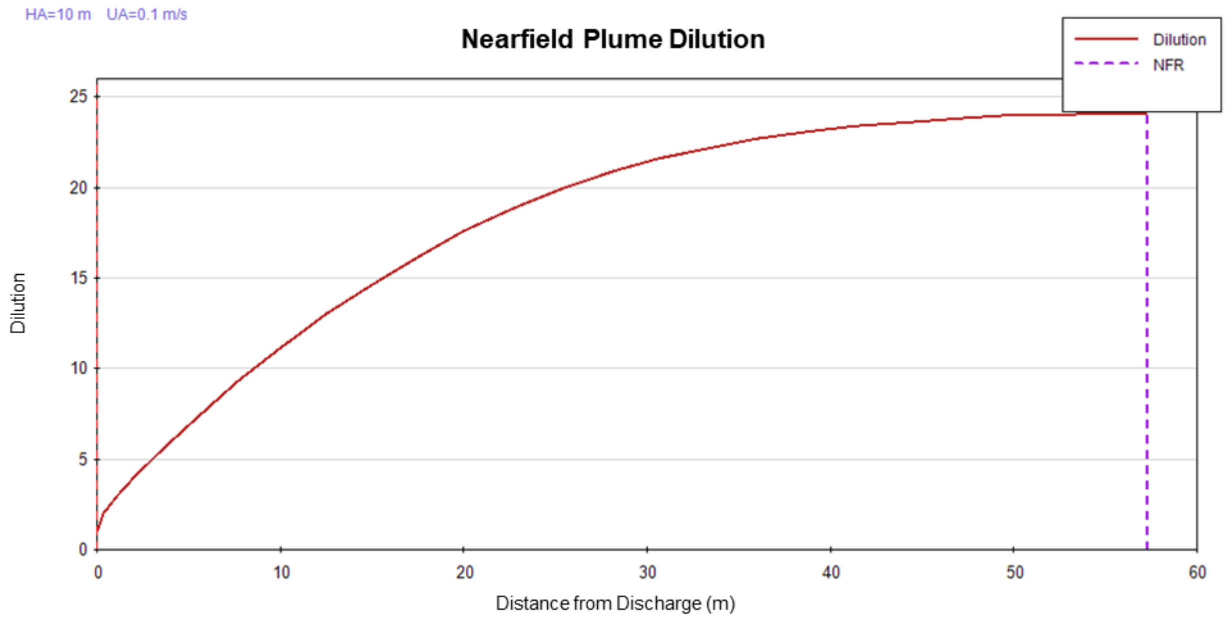

Figure 4-9. Discharge water dilution along the centerline of the plume versus downstream distance from discharge (x) one hour after slack tide. The edge of the nearfield region is indicated by the dotted purple line.

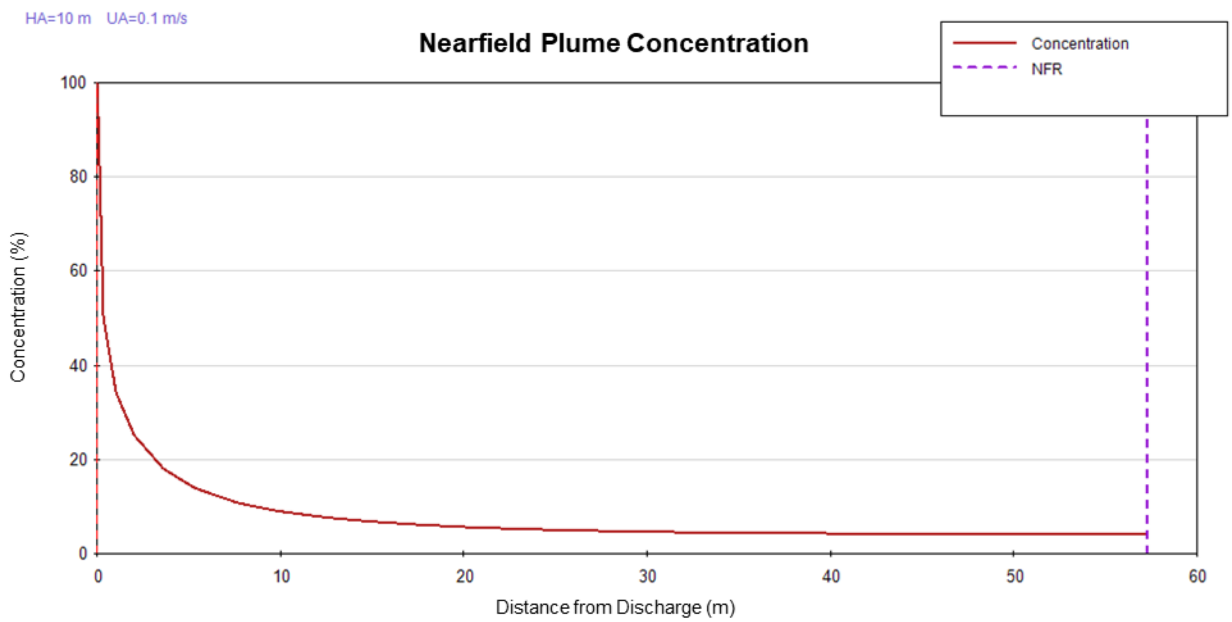

Figure 4-10. Fractional concentration of discharge water along the centerline of the plume versus downstream distance from discharge (x) one hour after slack tide. The edge of the nearfield region is indicated by the dotted purple line.

4.3.2.3 Modeled Nearfield Concentrations of Key Constituents

Table 4-11 reports the concentrations of the constituents near the discharge and at the edge of the nearfield region. It is noteworthy that in one ship sample the background measurement for copper was higher than the discharge concentration, resulting in a calculated concentration in the receiving waters higher than the discharge concentration. However, for all the other constituents, the concentrations 10 m away from the discharge were significantly reduced, reaching less than 60% of the initial discharge concentration during slack tide conditions and less than 50% of the discharge concentration one hour after slack tide. Moreover, at the edge of the nearfield region, these concentrations decreased even further, approaching the background concentrations.

Table 4-11. Estimated receiving water concentrations for key chemical constituents of concern in the Port of Seattle near Smith Cove Cruise Terminal.

Key Constituents	Measured Discharge Conc. (µg/L)	Measured Background Conc. (µg/L)	Slack Tide		Neap Tide	
			10 m Conc. (µg/L)	Edge of NFR Conc. (µg/L)	10 m Conc. (µg/L)	Edge of NFR Conc. (µg/L)
Vanadium	10.40	1.70	5.05	2.23	2.48	2.09
Nickel	2.66	0.78	1.50	0.90	0.95	0.86
C1-phenanthrene/anthracenes	0.98	0.02	0.39	0.08	0.11	0.06
C2-phenanthrene/anthracenes	1.23	0.02	0.49	0.09	0.13	0.07
C3-phenanthrene/anthracenes	1.2	0.02	0.47	0.09	0.13	0.07
C4-phenanthrene/anthracenes	0.83	0.02	0.33	0.07	0.09	0.05

Note: Values are shown for a location 10 m away from the discharge and at the edge of the nearfield region (51.5 m and 57.5 m for slack tide and one hour after slack tide, respectively).

4.3.2.4 CORMIX Limitations

The CORMIX model has some considerations that have been identified for this study:

- The geometry of the model domain is limited to the options available in the model, which means local topographic effects are not fully included. Similarly, the discharge geometry must also be simplified for the model simulations.
- Complex ambient flow patterns and interactions between the plume and local structures are not explicitly modeled in the domain.
- The steady-state formulation of the model does not account for concentration variance over time. A time-varying solution, such as one that includes numerical integration of the concentration field over a tidal cycle, would provide a more comprehensive analysis of local conditions throughout the tidal cycle.
- Environmental data at the Smith Cove Cruise Terminal are not available, data from nearby buoys were used to estimate ambient tidal conditions and stratification.

- Discharge flow rates and effluent concentrations were assumed to be constant, and representative of low engine load conditions based on measurements. However, discharge characteristics may be variable in magnitude and timing and may depend on the ship configuration.

4.3.3 Predicted Environmental Concentrations Summary

Predicted environmental concentrations (PECs) were estimated for identified contaminants of concern (COCs) using the dilution estimates from the MAMPEC model for the model cell located adjacent to the overboard discharge. The MAMPEC model was used to estimate overboard discharge water concentrations adjacent to the discharge while in transit (under high engine load) and while in Port (under low engine load). The highest toxic unit estimates from the TLM and the WET tests were divided by the dilution estimate to estimate toxic units in the MAMPEC cell adjacent to the discharge. The MAMPEC model estimated that overboard discharge water constituent concentrations while in transit (under high engine load scenario) will be reduced by a factor of 205. The MAMPEC also estimated overboard discharge water constituent concentrations to dilute by a factor of 40 while in Port under the low engine load scenario during average tidal conditions.

The highest toxic units as well as the estimated toxic units for each round of overboard discharge water analysis are provided in Table 4-12 and Table 4-13. Acute toxic unit values based on the TLM are included for reference, but since the chronic estimates were more conservative, the toxic unit dilution estimates were calculated for the chronic estimates. Toxic units for the WET tests are provided based on the chronic estimate using the EC_{10} values which are more conservative than the other endpoints. Chronic toxic unit post-dilution estimates for EGCS overboard discharge waters while transiting are low, ranging from 0.027 to 0.078; WET test results indicated lower toxic unit estimates from 0.000 to 0.017. Chronic toxic unit estimates for EGCS overboard discharge waters while in port are higher than estimates while underway but are still relatively low ranging from 0.125 to 0.375 based on the TLM estimates and 0.000 to 0.133 based on the WET test results.

Table 4-12. Risk Characterization Ratios – MAMPEC Shipping Lane Module

EGCS Water Collection Round	TLM TU Chronic	TLM (Max Cell TU Est.)	TLM TU Acute	WET Tests TU (EC_{10})	WET Test (Max Cell TU Est.)
High Engine Load					
1	11.0	0.054	2.1	1.4	0.008
2	16.0	0.078	3.1	3.4	0.017
3	2.8	0.014	0.5	1.8	0.009
4	5.5	0.027	1.1	1.8	0.009
5	14.0	0.068	2.7	1.8	0.009
6	7.2	0.035	1.4	<1.0	0.000

Table 4-13. Risk Characterization Ratios – MAMPEC Estuarine Harbor Module

EGCS Water Collection Round	TLM TU Chronic	TLM (Max Cell TU Est.)	TLM TU Acute	WET Tests TU (EC_{10})	WET Test (Max Cell TU Est.)
Low Engine Load					
1	14.0	0.350	2.7	1.5	0.038
2	15.0	0.375	2.9	4.1	0.103
3	5.0	0.125	1.0	2.4	0.060
4	7.5	0.188	1.4	<1.0	0.000
5	15.0	0.375	3.0	5.3	0.133
6	5.4	0.135	1.0	<0.1	0.000

The CORMIX model was used to refine the toxicity estimates provided by the MAMPEC model. This model estimated the dilution of overboard discharge waters based on a centerline of expected flow while in port. Based on this assessment, the centerline of flow would be expected to contain the highest concentration of overboard discharge water constituents. Overboard discharge water concentrations would reduce laterally from the centerline. The dilution factor for the slack tide scenario was 16.3, while the one-hour post slack tide scenario estimated a 24.1-fold dilution, located at 51.5 m and 57 m, respectively, from the discharge.

Using the CORMIX results we estimated the distance along the centerline where the toxic units provided by the TLM, and the WET test would be reduced below 1.0 during the slack tide and one-hour post slack tide scenarios. Estimated distances are provided in Table 4-14. During the slack tide scenario, the TLM toxic units are estimated to reduce below 1.0 from around 18 to 50 meters and from 4-25 meters during the one-hour post slack tide scenario. Toxic units calculated based on WET test results were less than those based on TLM calculations. During the slack tide scenario, the WET test toxic units would be estimated to reduce below 1.0 from 0 to 6 m from the point of discharge for the slack tide scenario, or from 0 to less than 1 m during the one-hour post slack tide scenario.

Table 4-14. The distance CORMIX would predict toxic units would be less than 1.

EGCS Water Collection Round	TLM Chronic	CORMIX Distance Slack/Post Slack (m)	TLM Acute	WET Tests (EC_{10})	CORMIX Distance Slack/Post Slack (m)
Low Engine Load					
1	14.0	46/14	2.7	1.5	5/<1
2	15.0	49/16	2.9	4.1	15/2
3	5.0	17/3.2	1.0	2.4	8/<1
4	7.5	26/5.7	1.4	<1.0	0/0
5	15.0	49/16	3.0	5.3	20/3.5
6	5.4	19/3.5	1.0	<0.1	0/0

WebPlotDigitizer was used to estimate the distance where TU = 1 for the CORMIX model results (WebPlotDigitizer, 2022)

5. CONCLUSIONS

This study has identified PAH and metals constituents present in the overboard discharge water samples that have the greatest potential to impact organisms in the Puget Sound. Nickel and vanadium, found to be present in ship fuels, were the only metals that exceeded PNEC values in overboard discharge waters. PNECs are guideline concentrations developed by European regulators derived from empirical laboratory-based ecotoxicity testing. PNECs are intentionally conservative, i.e., protective, and provide guidance to regulators and risk assessors on the concentration of a substance below which adverse effects in the environmental are not expected to occur. Exceedance of a PNEC does not necessarily portend adverse environmental effects, that must be considered by evaluation on a site-specific basis. Average nickel concentrations were less than the U.S. EPA water quality criteria and Washington State water quality standards. European PNEC values, like US water quality criteria, have been developed using data derived from empirical laboratory-based toxicity studies; however, the quantity and quality of the base data are not equal for all metals. Even with the potentially conservative PNEC, concentrations of vanadium would be expected to be below this PNEC value based on the MAMPEC results for the waters nearest to the overboard discharge location.

The TLM predicted both acute and chronic toxicity based on PAHs present in the overboard discharge water samples; however, this is a conservative model and the results of the WET tests empirically showed less toxicity in the actual discharge water samples. Whole effluent toxicity tests are considered “method-defined analyte[s]” which means that the analyte does not have a known composition; therefore, the analyte result is dependent upon the response observed in the toxicity assay (U.S. EPA, 1999). These tests not only test the chemical constituents that are measured analytically, but also any other compounds present in the mixture. These tests can also estimate any potential interactive actions that occur between compounds within the mixture. The WET tests performed for this study used marine organisms and methodologies consistent with those used by WDOE for monitoring other industrial discharges to the Puget Sound marine waters. The tests were based on U.S. EPA and WDOE methods for short-term chronic testing which are used for assessing other discharges in Puget Sound.

A MAMPEC model (*Marine Antifoulant Model to Predict Environmental Concentrations*) was used to assess the fate and distribution of EGCS associated waters when discharged. MAMPEC has the ability to estimate hydrodynamic exchanges in six generalized “typical” environments two scenarios were evaluated in this study. The Shipping Lane Module was used to predict the fate of the constituents associated with the discharge and *Predicted Environmental Concentrations* (PEC) during transit while the Estuarine Harbor Module was used for assessing the fate of the constituents associated with the discharge and the PECs while in port. MAMPEC modeling results of simulated PECs in the discharge areas showed no significant increase from the background concentration for constituents which were identified over time, primarily due to local conditions and the high flushing rate of the constituents from the discharge area. The MAMPEC model produced estimates for dilutions of 205-fold (while in transit) and 40-fold (while in port). The highest toxic units resulting from the TLM, and the WET tests were used to estimate *Risk Characterization Ratios* (RCR)[6] in the nearest cell to the discharge by dividing these values by the MAMPEC derived dilution estimates. The resulting RCRs were less than 1.0 suggesting minimal potential for ecological risk in the characterized area.

The *Cornell Mixing Zone Model* (CORMIX) was used to refine the estimates from the MAMPEC model for the in-port scenario. CORMIX model estimates were simulated under two discharge scenarios, discharge during slack tide and one-hour post slack tide. The CORMIX model estimated a dilution of 16.3-fold (model extent was 51.5 meters) and 24.1-fold (model extent was 57 meters) would occur along the centerline of discharge. Reduction of toxic units from TLM calculations and WET tests were estimated based on the dilution of the overboard discharge waters by distance along the centerline of discharge using the dilution curves from the CORMIX model. Dilution modeling estimated reduction of the toxic units from the conservative TLM to result in a RCRs of less than 1.0 between 4 and 50 meters along the centerline of discharge. Using toxic units calculated from the WET tests this estimate would result in an RCRs of less 1.0, between 0 and 6 meters; again, suggesting minimal ecological risk.

5.1 Appreciation

The comments and recommendations received to date from the Port of Seattle and Washington State Department of Ecology were particularly useful and appreciated by the Study Team, as were the onboard observers provided by the Port of Seattle for the collection of samples.

[6] RCR = Predicted Environmental Concentration (PEC)/Predicted No Effect Concentration (PNEC)) An RCR < 1 indicates that the identified risks are acceptable.

6. REFERENCES

ASTM. (1998). Standard Guide for Selection of Resident Species as Test Organisms for Aquatic and Sediment Toxicity Tests. Designation E-1850-97. Annual Book of ASTM Standards. Vol. 11.04.

ASTM. (2021). *Standard Guide for Conducting Static Toxicity Tests with Microalgae.* American Society for Testing and Materials.

Bellas, J., Saco-Alvarez, L., Nieto, O., & Beiras, R. (2008). Ecotoxicological evaluation of polycyclic aromatic hydrocarbons using marine invertebrate embryo-larval bioassays. *Marine Pollution Bulletin*, 493-502.

Burgess, R. M., Berry, W. J., Mount, D. R., Ankley, G. T., Ireland, S., Di Toro, D. M., . . . Zarba, C. S. (2008). *Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: compendium of tier 2 values for nonionic organics.* Washington, D.C.: U.S. Environmental Protection Agency.

Church, B. G., Van Sprang, P. A., Chowdhury, M., & DeForest, D. K. (2017). Updated Species Sensitivity Distribution Evaluations for Acute and Chronic Lead Toxicity to Saltwater Aquatic Life. *Environmental Toxicology & Chemistry*, 2974-2980.

De Forest, D. K., & Schlekat, C. E. (2013). Species Sensitivity Distribution Evaluation for Chronic Nickel Toxicity to Marine Organisms. *Integrated Environmental Assessment and Management*, 9:580-589.

de Vries, P., Jak, R. G., & Frost, T. K. (2022). Comparison of Substance-Based and Whole-Effluent Toxicity of Produced Water Discharges from Norwegian Offshore Oil and Gas Installations. *Environmental Toxicology and Chemistry*, 2285-2304.

Di Toro, D. M., McGrath, J. A., & Hansen, D. J. (2000). Technical basis for narcotic chemicals and PAH criteria in water and tissue. *Environmental toxicology and Chemistry*, 1951-1970.

ECCC. (2020). *Draft Screening Assessment Thallium and its Compounds.* Environment and Climate Chage Canada Health Canada.

ECHA. (2008). *Guidance on information requirements and chemical safety assessment.* European Chemical Agency.

ECHA. (n.d.). *Arsenic*. Retrieved 11 2022, from Registered Dossier: https://echa.europa.eu/registration-dossier/-/registered-dossier/22366/6/1

ECHA. (n.d.). *Cadmium*. Retrieved 11 2022, from Registered Dossier: https://echa.europa.eu/registration-dossier/-/registered-dossier/15342/6/1

ECHA. (n.d.). *Lead*. Retrieved 11 2022, from Registered Dossier: https://echa.europa.eu/registration-dossier/-/registered-dossier/16063/6/1

ECHA. (n.d.). *Mercury*. Retrieved 11 2022, from Registration Dossier: https://echa.europa.eu/registration-dossier/-/registered-dossier/5169/6/1

ECHA. (n.d.). *Vanadium*. Retrieved from https://echa.europa.eu/registration-dossier/-/registered-dossier/15421/6/2/1

EcoAnalysts. (2022). *Puget Sound EGCS Washwater Study Plan.* Prepared for Carnival Corporation & PLC.

EU. (2008). *Voluntary Risk Assessment of Copper, Copper II Sulphate Pentahydrate, Copper(I)Oxide, Copper(II)Oxide, Dicopper Chloride Trihydroxide.* Brussels, Belgium: European Copper Institute.

European Communities. (2003). *Technical Guidance Document on Risk Assessment.* European Commission Joint Research Centre.

Fridell, E., & Salo, K. (2016). Measurements of abatement of particles and exhaust gases in a marine gas scrubber. *Engineering for the Maritime Environment*, 154-162.

Graphpad. (2023). Retrieved from https://www.graphpad.com/quickcalcs/grubbs1/

Grubbs, F. E. (1969). Procedures for Detecting Outlying Observations in Samples. *Technometrics*, 1-21.

IMO. (2019). *IMO 2020 - Cutting Sulphur Oxide Emissions*. Retrieved from https://www.imo.org/en/MediaCentre/HotTopics/Pages/Sulphur-2020.aspx

Isnard, P. P.-S. (2001). Statistical analysis of regulatory ecotoxicity tests. *Chemosphere*, 659-669.

Koski, M., Stedmon, C., & Trapp, S. (2017). Ecological effects of scrubber water discharge on coastal plankton: Potential synergistic effects of contaminants reduce survival and feeding of the copepod Acartia tonsa. *Marine Environmental Research*, 374-385.

Linders, J., Adam, E., Behrends, B., Dock, A., Hanayama, S., Luit, R., . . . Tronczynski, J. (2019). *Evaluation and Harmonization of Rules and Guidance on the Discharge or Liquid Effluents from EGCS into Waters, Including Condition and Areas.* GESAMP Task Team on EGCS.

Linders, J., Adams, E., Behrends, B., Dock, A., Hanayama, S., Luit, R., . . . Tronczynski, J. (2019). *Advice to the Marine Environment Protection Committee and its Sub-Committee on Pollution Prevention and Response Report of the GESAMP Task Team on EGCS.* PPR 7 - INF.23, GESAMP Task Team on exhaust gas cleaning systems.

Maycock, D., Peters, A., Merrington, G., & Crane, M. (2010). *Proposed EQS for Water Framework Directive Annex VIII substances: zinc (For consultation).* Edinburgh: Water Framework Directive.

Maycock, D., Sorokin, N., Atkinson, C., Rule, K., & Crane, M. (2007). *Proposed EQS for Water Framework Directive Annex VIII substances: chromium (VI) and Chromium(III) (dissolved).* Bristol: Environment Agency.

McGrath, J. A., & Di Toro, D. M. (2009). Validation of the target lipid model for toxicity assessment of residual petroleum constituents: monocyclic and polycyclic aromatic hydrocarbons. *Environmental Toxicology and Chemistry*, 28:1130-1148.

McGrath, J. A., Fanelli, C., Di Toro, D. M., Parkerton, T. F., Redman, A. D., Paumen, M. L., . . . den Haan, K. (2018). Re-evaluation of target lipid model-derived HC5 predictions for hydrocarbons. *Environmental Toxicology & Chemistry*, 37:1579-1593.

McGrath, J. A., Parkerton, T. F., Hellweger, F., & Di Toro, D. M. (2005). Validation of the narcosis target lipid model for petroleum products: Gasoline as a case study. *Environmental Toxicology & Chemistry*, 24:2382-2394.

McGrath, J., Getzinger, G., Redman, A. D., Edwards, M., Aparicio, A. M., & Vaiopouloud, E. (2021). Application of the Target Lipid Model to Assess Toxicity of Heterocyclic Aromatic Compounds to Aquatic Organisms. *Environmental Toxicology & Chemistry*, 40:3000-3009.

Newman, M. C., Dixon, P. M., Looney, B. B., & Pinder, J. E. (1989). Estimating mean and variance for environmental samples with below detection limit observations. *Water Resources Bulletin*, 25:905-916.

Norwood, W. P., Borgmann, U., Dixon, D. G., & Wallace, A. (2003). Effects of Metal Mixtures on Aquatic Biota: A Review of observations and Methods. *Human and Ecological Risk Assessment: An international Journal*, 795-811. doi:10.1080/713610010

Parkerton, T. F., Bok, M., Ireland, A. W., & Prosser, C. M. (2018). An evaluation of cumulative risks from offshore produced water discharges in the Bass Strait. *Marine Pollution Bulletin*, 610-621.

R Core Team. (2021). A language and environment for statistical computing. R Foundation for Statistical Computing. Vienna, Austria. Retrieved from https://www.R-project.org/

Redman, A. D., Parkerton, T. F., MacGrath, J. A., & Di Toro, D. M. (2012). Petrotox: An aquatic toxicity model for petroleum products. *Environmental Toxicology & Chemistry*, 31:2498-2506.

Redman, A. D., Parkerton, T. F., Paumen, M. L., McGrath, J. A., den Haan, K., & Di Toro, D. M. (2014). Extension and validation of the target lipid model for deriving predicted no-effect concentrations for soils and sediments. *Environmental Toxicology & Chemistry*, 33:2679-2687.

Stephen, C. E., Mount, D. I., Hansen, D. J., Gentile, J. R., Chapman, G. A., & Brungs, W. A. (1985). *Guidelines for deriving numberical national water qualtiy criteria for the protection of aquatic organisms and their uses.* Office of Research and Development. Springfield, VA: U.S. Environmental Protection Agency.

U.S. EPA. (1995). *Short-Term Methods for Estimating the Chronic Toxicity of effluents and Receiving waters to West Coast Marine and Estuarine Organisms.* Cincinnati: United States Environmental Protection Agency.

U.S. EPA. (1999). *Protocol for EPA Approval of New Methods or Alternate Test Procedures for Whole Effluent Toxicity.* Washington D.C.: Office of Water.

U.S. EPA. (2002). *Short-Term Methods for Estimating the Chronic Toxicity of Effluents and receiving waters to Marine and Estuarine Organisms.* Washington DC: U.S. Environmental Protection Agency.

U.S. EPA. (2003). *Procedures for the Derivation of Equilibrium Partitioning Sediment Benchmarks (ESBs) for the Protection of Benthic Organisms: PAH Mixtures.*

U.S. EPA. (2010). *Designation of North American Emission Control Area to Reduce Emissions from Ships.* United States Environmental Protection Agency.

van Hattum, B., van Gils, J., Elzinga, H., & Baart, A. (2014). *MAMPEC 3.0 Handbook.* Amsterdam: Institute for Environmental Studies.

VGP. (2013). *Vessel General Permit for Discharges Incidental to the Normal Operation of Vessels (VGP).*

Vijver, M. G., Elliott, E. G., Peijnenburg, W., & De Snoo, G. R. (2011). Response predictions for organisms water-exposed to metal mixtures: A meta-analysis. *Environmental Toxicology and Chemistry*, 30:1482-1487.

Warne, M., & van Dam, R. (2008). NOEC and LOEC Data Should no Longer be Generated or Used. *Australasian Journal of Ecotoxicology*, 1-5.

WDOE. (2016). *Whole Effluent Toxicity Testing Guidance and Test Review Criteria.* Olympia: Washington State Department of Ecology.

WebPlotDigitizer. (2022). Retrieved from https://automeris.io/WebPlotDigitizer/