Water Quality Assessment and Proposed Objectives for Burrard Inlet: Contaminants of Emerging Concern Technical Report

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Tsleil-Waututh Nation
səlilwətaɬ

BRITISH COLUMBIA

This Technical Report forms part of a series of water quality parameter reports whose purpose is to inform updates to the 1990 Provincial Water Quality Objectives for Burrard Inlet. This report and others in the series assess the current state and impacts of contamination in Burrard Inlet; incorporate new scientific research and monitoring of water quality; and reflect a broader understanding of goals and values, including those of First Nations, to improve the health of the marine waters of Burrard Inlet. Updating the 1990 Provincial Water Quality Objectives is a priority action identified in the Tsleil-Waututh Nation's Burrard Inlet Action Plan which has been an impetus for this work.

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Cover Photograph:
Underwater monitoring equipment is installed from the Tsleil-Waututh Nation boat in Burrard Inlet.

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CHAPTER SUMMARY

This chapter presents proposed water quality objectives for Contaminants of Emerging Concern (CECs) in Burrard Inlet. These proposed objectives were developed using up-to-date research on relevant values and potential effects, sources and factors influencing CEC levels, benchmark screening, and historic and recent monitoring data for Burrard Inlet.

CECs can refer to many kinds of chemicals and there is not one unified list of CECs worldwide. CECs covered in this report include:

- Alkylphenols and their ethoxylates;
- Bisphenols;
- Phthalates;
- 6PPD and 6PPD-quinone;
- Organotins;
- Brominated flame retardants; and
- Perfluoroalkyl and polyfluoroalkyl substances.

The selected CECs are of anthropogenic origin and are organic molecules with a backbone of carbon and hydrogen, bonded with other elements and/or other carbon atoms. The list of contaminants of emerging concern to Burrard Inlet may expand as additional monitoring is performed and analytical methods are developed, so that more compounds, at lower concentrations, can be detected.

Once CECs enter marine waters, some of these compounds are persistent, bioaccumulative and toxic towards biota at low concentrations. Several CECs exert acutely toxic effects on aquatic organisms, as well as long-lasting effects that may have implications on future generations. Effects of CECs may not yet be known and breakdown products and mixtures of different contaminants interacting with each other also have the potential to affect the health of receiving aquatic environments.

Values requiring protection from CEC pollution include aquatic life and human consumption of finfish and shellfish. Wildlife that consumes fish, including marine mammals, are particularly vulnerable to CECs due to potential bioaccumulation in the food web. Humans can be exposed to these contaminants through their food or from the environment.

Available data from key monitoring programs established for Burrard Inlet were used to screen CECs against benchmarks that were developed using relevant BC guidelines, and guidelines from other jurisdictions in the absence of BC guidelines. The majority of measured CECs were below laboratory detection limits but, in many cases, detection limits were higher than the screening benchmark level. Among the available data, nonylphenol and its ethoxylates were detected in marine water samples; nonylphenol and its ethoxylates, bisphenol A, organotins, tributyltin, perfluorooctanesulfonic acid (PFOS) and hexabromobenzene (HBB) were detected in sediment samples; and nonylphenol and its ethoxylates were detected in tissue samples. The only exceedance of a screening benchmark was for nonylphenol and its ethoxylates; the screening benchmark was exceeded in 87% of 15 composite blue mussel tissue samples collected by the Ocean Wise PollutionTracker program between 2015 and 2019.

The proposed water quality objectives for CECs are as follows:

Proposed Water Quality Objectives for Contaminants of Emerging Concern

Sub-basin	False Creek	Outer Harbour	Inner Harbour	Central Harbour	Port Moody Arm	Indian Arm
All media	**All CECs** Decreasing trend in concentrations					
Water	**Alkylphenols and their Ethoxylates** Nonylphenol and its Ethoxylates: 0.7 µg/L (total toxic equivalent of nonylphenolic compounds)					
	Bisphenols Bisphenol A (BPA): 0.9 µg/L					
	Phthalates Di-methyl phthalate (DMP): 2000 µg/L Di-ethyl phthalate (DEP): 600 µg/L Di-(n)-butyl phthalate (DnBP): 30 µg/L Benzyl butyl phthalate (BBP): 0.1 µg/L Di-(2-ethylhexyl) phthalate (DEHP): 0.37 µg/L Di-n-butyl phthalate (DnOP): Do not detect* and/or decrease in current levels					
	6PPD and 6PPD-Quinone 6PPD and 6PPD-Quinone: Do not detect* and/or decrease in current levels					
	Organotins Tributyltin (TBT): 0.001 µg/L					
	Brominated Flame Retardants Hexabromocyclododecane (HBCD): 0.56 µg/L Tetrabromobisphenol A (TBBPA): 3.1 µg/L Hexabromobenzene (HBB): Do not detect* and/or decrease in current levels					
	Perfluoroalkyl and Polyfluoroalkyl Substances Perfluorooctanoic acid (PFOA): Do not detect* and/or decrease in current levels Perfluorooctanesulfonic acid (PFOS): 3.4 µg/L					
Sediment	**Alkylphenols and their Ethoxylates** Nonylphenol and its Ethoxylates: 1.0 µg/g dry weight (total toxic equivalent of nonylphenolic compounds; adjust objective to site-specific levels of total organic carbon [TOC])					
	Bisphenols BPA: 0.025 µg/g dry weight (adjust objective to site-specific levels of TOC)					
	Phthalates DMP: 0.53 µg/g dry weight (adjust objective to site-specific levels of TOC) DEP: 0.61 µg/g dry weight (adjust objective to site-specific levels of TOC) DnBP: 2.2 µg/g dry weight (adjust objective to site-specific levels of TOC) BBP: 0.049 µg/g dry weight (adjust objective to site-specific levels of TOC) DEHP: 0.47 µg/g dry weight (adjust objective to site-specific levels of TOC) DnOP: 0.58 µg/g dry weight (adjust objective to site-specific levels of TOC)					
	6PPD and 6PPD-Quinone 6PPD and 6PPD-Quinone: Do not detect* and/or decrease in current levels					
	Organotins Do not detect* and/or decrease in current levels					
	Brominated Flame Retardants HBCD: 1.6 µg/g dry weight TBBPA: 0.6 µg/g dry weight HBB: Do not detect* and/or decrease in current levels					

Sub-basin	False Creek	Outer Harbour	Inner Harbour	Central Harbour	Port Moody Arm	Indian Arm
	Perfluoroalkyl and Polyfluoroalkyl Substances PFOA: Do not detect* and/or decrease in current levels PFOS: Do not detect* and/or decrease in current levels					
Tissue	**Alkylphenols and their Ethoxylates** Nonylphenol and its Ethoxylates: 0.018 µg/g wet weight					
	Bisphenols BPA: Do not detect* and/or decrease in current levels					
	Phthalates DMP: Do not detect* and/or decrease in current levels DEP: Do not detect* and/or decrease in current levels DnBP: Do not detect* and/or decrease in current levels BBP: Do not detect* and/or decrease in current levels DEHP: Do not detect* and/or decrease in current levels DnOP: Do not detect* and/or decrease in current levels					
	6PPD and 6PPD-Quinone 6PPD and 6PPD-Quinone: Do not detect* and/or decrease in current levels					
	Organotins TBT: 0.0088 µg/g wet weight Organotins: 0.0088 µg/g wet weight					
	Brominated Flame Retardants HBCD: Do not detect* and/or decrease in current levels TBBPA: Do not detect* and/or decrease in current levels HBB: 0.00070 µg/g wet weight					
	Perfluoroalkyl and Polyfluoroalkyl Substances PFOA: 0.0007 µg/g wet weight PFOS: 0.0021 µg/g wet weight					
*Any objective of 'do not detect' implies non-detection when using best available detection limits.						

Where there is sufficient toxicological information to establish quantitative water quality objectives for the CECs outlined in this report, it is proposed to do so to be protective of marine aquatic life and human consumption of shellfish and finfish. Where there have been no detects of a given CEC, the objective should be 'do not detect' when using best available detection limits, and/or 'decrease from current levels' when detection limits are adequately low. Any objective of 'do not detect' implies non-detection when using best available detection limits. As more data is gathered, water quality objectives should be re-evaluated.

Where there is insufficient toxicological information, and in general, a qualitative objective is proposed for a decreasing trend in the concentrations of all CECs in all media over time.

Sampling for CECs in Burrard Inlet has been limited to date. Future monitoring recommendations include monitoring for potential sources of CECs in Burrard Inlet, monitoring for potential CEC accumulation in the marine food chain, monitoring potential adverse effects on the marine environment, and continuing to consider novel CECs in future monitoring.

Until marine-relevant toxicity data and additional high quality monitoring data are available, management priorities should focus on source control and monitoring, with the goal of reducing concentrations of CECs in water, sediment, and biota over time.

CONTENTS

TABLES

ACRONYMS

AP(E)	Alkylphenol (ethoxylate)
BBP	Benzyl butyl phthalate
BC	British Columbia
BFR	Brominated flame retardant
BPA	Bisphenol A
CEC	Contaminant of emerging concern
DBP/DnBP	Di-(n)-butyl phthalate
DEP	Di-ethyl phthalate
DEHP	Di-(2-ethylhexyl) phthalate
DIDP	Di-isodecyl phthalate
DINP	Di-isononyl phthalate
DMP	Di-methyl phthalate
DnOP	Di-n-butyl phthalate
ENV	Ministry of Environment and Climate Change Strategy
HBB	Hexabromobenzene
HBCD	Hexabromocyclododecane
HLTH	Ministry of Health
NP(E)	Nonylphenol (ethoxylate)
OP(E)	Octylphenol (ethoxylate)
PFAA	Perfluoroalkyl acid
PFAS	Perfluoroalkyl and polyfluoroalkyl substances
PFCA	Perfluorocarboxylic acid
PFOA	Perfluorooctanoic acid
PFOS	Perfluorooctanesulfonic acid
PVC	Polyvinyl chloride
SQG	Sediment Quality Guideline
SRKW	Southern Resident Killer Whales
SV	Screening values
TBT	Tributyltin
TBBPA	Tetrabromobisphenol A
TOC	Total organic carbon
TRV	Toxicological reference values
TWN	Tsleil-Waututh Nation
WQG	Water Quality Guideline
WQO	Water Quality Objective
WWQG	Working Water Quality Guideline
WWTP	Wastewater treatment plant

1. INTRODUCTION

Water quality objectives (WQOs) for Burrard Inlet represent an effort to inform water quality management and protect the water values associated with the marine waters of Burrard Inlet and its freshwater tributaries. The current Burrard Inlet WQOs were developed in 1990 and are updated to promote the protection of water quality and the associated water values.

This chapter includes relevant background information, an overview assessment of trends in concentrations of selected Contaminants of Emerging Concern (CECs) in Burrard Inlet, comparison to screening benchmarks, and a rationale for the proposed updated WQOs. Recommendations for future monitoring as well as management options to help achieve these objectives are also included. Tsleil-Waututh Nation (TWN) has identified CECs as priority issues in their Indigenous-led, science-based Burrard Inlet Action Plan (Tsleil-Waututh Nation 2017). Detailed context for this work and the Burrard Inlet area is provided in Rao *et al.* (2019).

CECs are not one group of chemicals with similar characteristics, but a unifying concept for chemicals for which environmental transport and fate, toxicity to aquatic biota, and human exposure may not yet be fully understood, and which are not yet subjected to regulatory criteria or norms for the protection of human health or the environment (Sauvé & Desrosiers, 2014). All CECs covered in this report have been used for decades, but their occurrence in and effects on the aquatic environment have not been investigated until recently and/or are still in need of research.

CECs can refer to many kinds of chemicals – including those used in household, industrial and other products – and there is not one unified list of CECs worldwide. Because many CECs are potentially relevant to Burrard Inlet, certain criteria were developed to prioritize CECs for this report. Existing water quality data collected from Burrard Inlet were used to create an initial list of relevant CECs in water and sediment. This was followed by a review of international research papers on CECs in environmental matrices relevant to Burrard Inlet, such as marine water, sediment, wastewater, and stormwater, to identify additional CECs that have not yet been monitored in Burrard Inlet. Potential CECs with existing Burrard Inlet monitoring data and/or identified through international literature were abandoned if they were not deemed contaminants of *emerging* concern, but rather have established toxicological information and/or are regulated. These include for example legacy and current use pesticides (Braig *et al*., 2023, in prep) and chlorinated solvents such as methylene chloride (also referred to as DCM), chloroform, and vinyl chloride. Further, pharmaceuticals and personal care products (PPCPs) and microplastics fulfill all criteria of CECs but have been covered in separate technical reports (Braig *et al.,* 2019a,b). Finally, the newly discovered toxic compound 6PPD-quinone and its precursor 6PPD were added to the list of CECs because of the compounds' potential detrimental effects on salmon populations.

CECs covered in this report include the following, which are described in more detail in section 2.3:

- Alkylphenols and their ethoxylates;
- Bisphenols;
- Phthalates;
- 6PPD and 6PPD-quinone;
- Organotins;
- Brominated flame retardants (BFRs); and
- Perfluoroalkyl and polyfluoroalkyl substances (PFAS).

The selected CECs are of anthropogenic origin and are organic molecules with a backbone of carbon and hydrogen, bonded with other elements and/or other carbon atoms.

The list of contaminants of emerging concern to Burrard Inlet may expand as additional monitoring is performed and analytical methods are developed, so that more compounds, at lower concentrations, can be detected. There is also a constant development of chemicals, and the chemicals we use today may become the CECs of tomorrow. According to the US EPA, there are somewhere between 25,000 and 84,000 chemicals in commerce in the US and the US Department of Health estimates that 2000 new chemicals are being released every year (Board on Population Health and Public Health Practice 2014). There is currently insufficient toxicity and ecotoxicity data available for many of the chemicals we use and potentially many more chemicals are of emerging concern to the aquatic environment and human health.

This chapter includes relevant background information, an overview assessment of current status and trends in CEC levels in marine water, sediment, and biota in Burrard Inlet, compared to screening benchmarks, and a rationale for the proposed objectives. Recommendations for future monitoring as well as management options to help achieve these objectives are also included. Detailed context for this work and the Burrard Inlet area is provided in Rao *et al.* (2019).

2. BACKGROUND

A general description of Burrard Inlet, including the water uses, land uses and other factors that affect water quality and values in the inlet, is available in the first volume of this technical series (Rao et al. 2019). This technical report supplements that general description with CEC-specific information.

2.1 Values

Once CECs enter marine waters, some of these compounds are persistent, bioaccumulative and toxic towards biota at low concentrations. Several CECs exert acutely toxic effects on aquatic organisms, as well as long-lasting effects that may have implications on future generations. Effects of CECs may not yet be known and breakdown products and mixtures of different contaminants interacting with each other also have the potential to affect the health of receiving aquatic environments.

Values requiring protection from CEC pollution include aquatic life and human consumption of finfish and shellfish. Wildlife that consumes fish, including marine mammals, are particularly vulnerable to CECs due to potential bioaccumulation in the food web. Humans can be exposed to these contaminants through their food or from other pathways of exposure.

2.2 Potential Sources of CEC Pollution and Factors Influencing their Levels in Burrard Inlet

The CECs investigated in this report are entirely anthropogenic and are released into the environment from human activities. Because many CECs are persistent, e.g., PFAS and several BFRs, long-range transport and atmospheric deposition, either directly into the marine environment or on land, is likely (de Wit, Herzke, and Vorkamp 2010; Zhao et al. 2012). The main potential sources of CECs in Burrard Inlet are sewage discharges and stormwater, including wastewater treatment plant (WWTP) effluents, sanitary sewer overflows and combined sewer overflows, as well as industrial activities, and atmospheric deposition[1].

Because of their occurrence in many household products, most CECs are commonly occurring contaminants in wastewater influents and effluents (Gao and Wen 2016; Lenka, Kah, and Padhye 2021; Liu et al. 2021; Schreder and La Guardia 2014; Ying, Williams, and Kookana 2002). Removal of CECs in WWTPs is attributed to biodegradation, volatilization, or sorption to sludge. Wastewater treatment is not designed to remove CECs and has variable effectiveness in removing CECs, depending on the

[1] Sewer discharges into Burrard Inlet are described more in detail in Rao *et al.* (2019).

compound's physical-chemical properties.

Many CECs are functional additives, such as stabilisers, antistatic agents, flame retardants, plasticizers, and lubricants, in plastic materials. Additives can leach to the marine environment when the plastic weathers. Hence, plastic litter is a potential source of several CECs covered in this report, e.g., alkylphenols, bisphenols, and phthalates (Gallo et al. 2018).

Once released into the marine environment, the fate of CECs depends on environmental factors (e.g., pH, temperature, amount of sunlight exposure, and aerobic/anaerobic conditions) as well as the physical-chemical properties of the compound. The octanol/water partition coefficient, K_{ow}, is one of the most important factors to predict the environmental fate of organic compounds. A log $K_{ow} > 3$ indicates hydrophobic character and the bioaccumulation potential increases with increasing log K_{ow} (CSAP Society 2015). Hydrophobic compounds partition to the lipids of organisms and is the reason why oily fish such as salmon generally have a higher content of organic pollutants than lean fish. Hydrophobic compounds also tend to partition to solids and hence in sediments. Sediment has the potential to store CECs and release them back into the water column from the surface layer when there are changes in the physical or chemical environment, such as tidal changes, heavy storm events or changes in salinity and pH. Once in the sediment, some CECs will persist as the opportunities for biotic and abiotic degradation are limited (Mackay, Celsie, and Parnis 2015).

2.3 Use, Environmental Fate, and Toxicity of Selected CECs

2.3.1 Alkylphenols and their Ethoxylates

Alkylphenols (APs) are used to produce alkylphenol ethoxylates (APEs), widely used as industrial surfactants in detergents, cleaning products, pesticides, lubricants, hair care products, as emulsifiers and solubilizers, and as additives in plastics. Alkylphenol residues can also be found in products that contain alkylphenolic resin, such as tires and other rubber products, adhesives, and carbonless copy paper. Nonylphenol ethoxylates (NPEs), produced from nonylphenol (NP), are the most widely used APEs, making up at least 80% of the global market, followed by octylphenol ethoxylates (OPEs), produced from octylphenol (OP). The ethoxylate chain can vary in length from 1 to 40 ethoxylate units and hence different APEs exhibit different physical-chemical and toxicological properties (Acir and Guenther 2018; Bennie et al. 1997).

WWTP and stormwater effluents are major potential sources of APs in Burrard Inlet. Reported removal efficiency of alkylphenolic compounds in WWTPs vary between 10 and 90%. APEs may be biodegraded, forming several degradation products, including short chain APEs and the parent compounds NP and OP. In general, long chain APEs are hydrophilic, partition to the dissolved phase and are released with WWTP effluents. Short-chain APEs, such as mono- and diethoxylates, NP and OP are more lipophilic and partition upon particulate matter and sludge (Acir and Guenther 2018; Luo et al. 2014).

APs in stormwater are mainly related to traffic and leaching from construction materials, where they are used in vehicle components and tires, as well as in PVC (polyvinyl chloride), concrete, bitumen, and drainage materials (Bressy et al. 2011; Markiewicz et al. 2017).

APEs are degraded to more persistent, more toxic, more lipophilic, and less water soluble shorter-chain APEs such as mono- to triethoxylates, as well as alkylphenols including NP and OP. APs and APEs are ubiquitous in the environment because of their widespread use, inadequate removal in WWTPs, and persistence, and have been detected in air, water, sediment, soil, and biota around the world (Priac et al. 2017; Ying, Williams, and Kookana 2002).

The toxicity of APEs increases with decreasing ethoxylate chain. NP and OP are both acutely toxic to fish, invertebrates, and algae. APs and short chain APEs are also endocrine disruptors and the parent

compounds NP and OP have a stronger endocrine potential than the ethoxylates. Studies have indicated that OP may have a greater estrogenic potential than NP, but OP concentrations in the environment are usually lower than NP. NP may cause feminization of aquatic organisms, decreases male fertility and testicular weight in fish, and decreases survival in young fish (Acir and Guenther 2018). Studies show that APs can interfere with the activity of certain estrogenic hormones in humans, which can be linked to cancer and/or tumors in reproductive tissues like breasts, uteruses, and ovaries (Noorimotlagh et al. 2020). 4-nonylphenol has been identified as a contaminant of minor concern to Southern Resident Killer Whales (SRKW) (ECCC 2020a).

2.3.2 Bisphenols

Bisphenol A (BPA) is used in the manufacturing of polycarbonate plastics, epoxy resins and thermal paper. Polycarbonate plastics have many applications including food and drink packaging, e.g., water bottles, baby bottles and plastic dinnerware, impact-resistant safety equipment, automobile parts, and medical devices. Epoxy resins are used in protective linings to coat metal products such as food cans, bottle tops, and water supply pipes. BPA is one of the highest volume organic chemicals produced worldwide and because of its widespread use, it is ubiquitous in the environment and has been detected in water, sediment, soil, biomass, and air (Liu et al. 2021).

The removal efficiency of bisphenols in WWTPs is usually around 70%. Secondary and advanced treatment processes, which are currently not used at the Lions Gate WWTP in Burrard Inlet, show higher removal of bisphenols. BPA is primarily removed through partitioning to the solid fraction of sludge. Among the analogues, bisphenol A, F and S are most often detected in WWTPs (Guerra et al. 2015; Liu et al. 2021).

BPA has systematically been detected in studies performed in Europe of road runoff and atmospheric fallout as well as in sediments from stormwater detention basins (Gasperi et al. 2014). Samples collected across 17 American states showed higher BPA concentrations in stormwater than has been found in WWTP effluents worldwide (Masoner et al. 2019). Similar conclusions were drawn from a study of BPA concentrations in runoff samples collected in the UK (Spahr et al. 2020).

BPA is a moderately water-soluble compound and has a moderate tendency for sorption to particles. Its log K_{ow} is approximately 3.5 in marine waters and the partition coefficient appears to increase slightly with increasing pH and salinity but decreases with increasing temperature. BPA is relatively rapidly degraded in the environment with half-lives of 5 days or less, depending on the medium of release, and is not considered persistent. BPA has a moderate potential for bioaccumulation, with higher potential in marine bivalves than in fish. The analogues bisphenol F and S have log $K_{ow} < 3$ and therefore a lower potential to adsorb to sediment and accumulate into tissue. Other bisphenol analogues, e.g., bisphenol AF, are more hydrophobic than BPA. Bisphenols are overall more persistent in sediment than in water (Liu et al. 2021).

BPA has been detected in tissues of several different aquatic species collected from marine and freshwater systems. The primary source of exposure to BPA for most people is through the diet as BPA can leach into food and beverages from food packaging. Studies show that canned seafood is more contaminated than the non-canned (Repossi et al. 2016). A Korean study of the relationship between exposure to BPA and seafood consumption showed higher BPA concentrations in men who frequently consumed large fish and tuna, shellfish, and other seafood (Y. Kim et al. 2020).

Human exposure to BPA is widespread; a 2016 study revealed that 81.5% of Canadians had BPA detected in their urine (Statistics Canada 2019). BPA exposure is a concern because of the possible health effects, including prostate cancer, breast cancer, obesity, diabetes, cardiovascular problems, some neurobehavioral effects, including anxiety, as well as reproductive effects (Schug and

Birnbaum 2014). Because many countries have restricted the use of BPA, analogues such as Bisphenol AF, Bisphenol F, and Bisphenol S have started taking over market shares from BPA (Liu et al. 2021).

BPA shows moderate acute toxicity towards aquatic organisms and has low bioaccumulation potential. Invertebrates and fish are more sensitive to BPA than plants or amphibians. BPA is an endocrine disruptor that has been shown to lead to abnormal sex ratios, reduced reproductive health, delayed development, and altered metabolism in fish (Canesi and Fabbri 2015). The bisphenol family has been identified as contaminants of major concern to Chinook salmon, the primary prey of SRKW (ECCC 2020a).

2.3.3 Phthalates

Phthalates are a family of organic industrial chemicals used in large amounts in the polymer industry and as solvents in consumer products. Low molecular-weight phthalates such as di-methyl and di-butyl phthalate (DMP and DBP, respectively) are used as solvents in cosmetics and other personal care products, in pesticides, paints, inks and dyes, sealants and car care products. High molecular-weight phthalates such as di-(2-ethylhexyl) phthalate (DEHP) are widely used as plasticizers in PVC and other plastics to make the polymer more flexible and durable. PVC may contain up to 50% phthalates by weight. High molecular-weight phthalates are found in a wide range of consumer products such as medical devices, food wrap, building materials, packaging, automotive parts, toys, and childcare articles. DEHP has long been the most used phthalate but because of its potential health risks, including for example reproductive effects, DEHP is being replaced in many products by di-isononyl phthalate (DINP) and di-isodecyl phthalate (DIDP) with similar chemical properties (Gao and Wen 2016; Heudorf, Mersch-Sundermann, and Angerer 2007).

Phthalates are not chemically bound to the plastic polymer and can leach out of products over time and diffuse into the surrounding environment, particularly under conditions involving heat and in contact with liquids or fats. Researchers have found measurable levels of many phthalate metabolites in the general population, reflecting their widespread use in household products (U.S. Department of Health & Human Services 2019).

Most studies show efficient removal of phthalates in WWTPs; reduction rates are usually 70% or higher. Higher-molecular weight phthalates such as DEHP are highly hydrophobic and partition strongly to sludge, where the compounds are likely to persist. The sorption to sludge decreases with lower molecular weight, while the susceptibility to biological degradation increases (Armstrong et al. 2018; Clara et al. 2010).

Phthalates have been detected in runoff from all types of land-uses but appear to be most strongly related to traffic. Phthalates are used in several road- and vehicle-related products, such as road paint, bitumen/asphalt, car care products, tires, and undercoating: the latter being a suspected major pathway to stormwater (Markiewicz et al. 2017; Wicke et al. 2021).

Phthalates are considered degradable and will not persist in the environment. Experimental data indicate biodegradation of phthalates in surface waters and aerobic sediments and is expected to be the dominant loss mechanism in surface waters. However, biodegradation rates are reduced by anaerobic conditions and at low temperatures. The removal and degradation of phthalates by abiotic processes such as hydrolysis and photodecomposition are, on the other hand, very slow or insignificant. Phthalates with partitioning coefficients log $K_{ow} > 3$, which is most phthalates except DMP and diethyl phthalate (DEP), will partition to particles in water and are likely to be detected in sediments rather than the water column. Biodegradation rates in sediments have been estimated at weeks for low-molecular weight phthalates and months for high-molecular weight phthalates. Low-molecular weight phthalates are bioaccumulative in aquatic invertebrates, fish, and amphibians. Larger, hydrophobic phthalates show

reduced bioaccumulation because of both lower permeability (larger molecules) and increased rates of biodegradation or metabolism. Studies also indicate that phthalates do not biomagnify in aquatic food-webs, which may in part be due to effective metabolism in higher organisms (Cousins, Mackay, and Parkerton 2003).

Studies indicate that the lower molecular weight phthalates show acute and chronic toxicity towards microorganisms, algae, aquatic invertebrates, and fish. Chronic effects include adverse effects on reproduction, damage to the liver, kidney, and other organs (Baloyi et al. 2021). DEHP has been identified as a contaminant of major concern to Chinook salmon (ECCC 2020a).

Phthalates are known endocrine disruptors binding to the estrogen receptors and are associated with decreased fertility, low birth weight, preterm birth, and pregnancy loss in humans. Study findings also suggest that exposure to phthalates during adolescence is associated with behaviours of attention-deficit/hyperactivity disorder (ADHD) (Darbre 2020; Shoaff et al. 2020). DEHP is also suspected to be carcinogenic (U.S. Department of Health & Human Services 2016).

2.3.4 6PPD and 6PPD-Quinone

Antiozonants are added to tire rubber to help prevent ozone from causing cracking, splitting, and degradation of the tire surface, with the end goal to improve tire safety characteristics and prolong the useful life of tires. A common antiozonants in tire rubber is 6PPD (N-phenyl-N'-(1,3-dimethylbutyl)-p-phenylenediamine), which reacts with ozone in the air to produce the analog 6PPD-quinone. 6PPD is present in tires at 1-2% of the weight and migrates over the life of the tire to the tire surface to supply a constant source of 6PPD. Both 6PPD and 6PPD-quinone are spread to the aquatic environment with tire wear particles (California Department of Toxic Substances Control 2021).

6PPD and 6PPD-quinone may be leached from the tire rubber surface by rain, evaporate into the atmosphere, or through mechanical tire abrasion on the roads. Tire wear particles transported with stormwater to receiving waters is the suspected major transport pathway of 6PPD and 6PPD-quinone to the environment. It is not yet known whether recycled tire rubber, used for example on artificial turfs, can leach 6PPD and 6PPD-quinone (California Department of Toxic Substances Control 2021). Recent data from a study of road runoff in Metro Vancouver has shown that 6PPD-quinone enters tributaries to Burrard Inlet at levels of concern during rain events (Brown et al. 2022); a preliminary summary of these data is presented in Appendix B. Other studies in Toronto, Puget Sound Area and San Francisco as well as in Germany have found detectable concentrations of 6PPD-quinone in both stormwater and receiving waters (Johannessen et al. 2021; Müller et al. 2022; Tian et al. 2021).

6PPD and 6PPD-quinone can reach WWTPs in areas with combined sewers. Data on the removal of 6PPD and 6PPD-quinone in WWTPs are very limited but suggest that 6PPD is removed to at least 90%. It is not known whether removal is by sorption to sludge or through degradation (California Department of Toxic Substances Control 2021).

Much is yet unknown about the toxicity and environmental fate of 6PPD-quinone, as studies about its occurrence in the environment started to appear as late as 2020. However, it is expected that 6PPD-quinone is relatively soluble in water and is transported with surface runoff to the aquatic environment (California Department of Toxic Substances Control 2021). 6PPD is not stable under environmental conditions and unlikely to persist in the aquatic environment. The compound degrades quickly, e.g., to 6PPD-quinone, when exposed to oxygen or ozone or by photodegradation (e.g., on road surfaces) and in water; its half-life is less than one day under aerobic conditions. 6PPD is hydrophobic but is not considered bioaccumulative. The persistence of 6PPD-quinone is unknown, but monitoring data suggests that its half-life in aquatic environments is longer than for 6PPD. 6PPD-quinone is not suspected to bioaccumulate. Given the high log K_{ow} of both 6PPD and 6PPD-quinone, the compounds

are assumed to partition to soil and sediment. Abiotic degradation of 6PPD is more limited in sediments and because of lack of data, it has been suggested that 6PPD should be considered persistent in sediments until further evidence show otherwise (California Department of Toxic Substances Control 2021; OECD 2005).

6PPD is toxic toward aquatic organisms at multiple trophic levels, including algae, the small crustacean *Daphnia magna* and fish (e.g., medaka, fathead minnow) (California Department of Toxic Substances Control 2021; OSPAR Commission 2006). Its endocrine disrupting properties have not been studied to date.

Studies in Washington found that the degradation product 6PPD-quinone cause acute mortality in coho salmon (*Oncorhynchus kisutch*), possibly through blood–brain barrier disruption (Blair et al. 2020; Tian et al. 2021). The LC-50 (the concentration of a chemical that is lethal to 50% of the test animals during the observation period) for 6PPD-quinone to coho salmon is < 0.1 µg/L, a concentration frequently found in stormwater, and should be categorized as a "very highly toxic" pollutant for aquatic organisms (Peter et al. 2022). Because the toxicity of 6PPD-quinone to salmon was newly discovered, very few toxicity studies on other aquatic species have been performed to date. Acute toxicity tests of freshwater fish and crustacean species (*Danio rerio, Oryzias latipes, Daphnia magna*, and *Hyalella azteca*) revealed that 6PPD-quinone did not exhibit acute lethal toxicity to any tested species at its maximum water solubility (Hiki et al. 2021). Concentrations lethal to coho salmon do not have an equivalent negative response in the closely related chum salmon (*Oncorhynchus keta*). Rainbow trout (*Oncorhynchus mykiss*) and brook trout (*Salvelinus fontinalis*) are also negatively affected by 6PPD-quinone, although the estimated LC-50 concentrations are higher at 0.59 and 1.0 µg/L, respectively, than those observed for coho (< 0.1 µg/L). It is currently not known why a more specific toxicity to 6PPD-quinone is seen in coho salmon than other tested species (Brinkmann et al. 2022; McIntyre et al. 2021). The broader impact of 6PPD-quinone on seafood consumption and human health is not yet understood. The human exposure potential and toxicity of 6PPD-quinone have not yet been evaluated directly, but its structural similarity to other quinones suggests that 6PPD-quinone may be reactive in biological systems (California Department of Toxic Substances Control 2021).

Recent studies have identified additional and potentially toxic compounds in leachate from tire rubber, including PREPOD[2], as well as the occurrence of additional rubber-derived quinones formed from tire antioxidants (similar to 6PPD-quinone) in environmental matrices (Cao et al. 2022; Müller et al. 2022). The authors concluded that there are many, still unknown, chemicals emitted from tire rubber to the environment.

2.3.5 Organotins

Organotins are tin (Sn) compounds having one to four organic groups attached. Many organotins are powerful biocides and are used as fungicides, pesticides, bactericides, and molluscicides. Since the early 1960s, organotins have been extensively used as antifouling additives in paints used on marine vessels and marine structures to prevent growth of algae, barnacles, molluscs, and other organisms. Tributyltin (TBT) is the active compound in most antifouling paints. Organotins leach out from the paint and into the surrounding water, and sediments found in harbors, marinas, and other marine waterways have shown especially high concentrations of organotins (De Carvalho Oliveira and Santelli 2010).

[2] PREPOD is used as an ingredient in the manufacture of rubber products, including tires, and has the potential to cause harm to aquatic organisms, persist in the environment, and accumulate in aquatic organisms. More information about PREPOD is found here: https://www.canada.ca/en/health-canada/services/chemical-substances/challenge/batch-11/prepod.html

The main source for organotins in many marine environments is antifouling paint, while wastewater and stormwater are considered less significant (Health Canada 2010c). Organotins have, however, been detected in a few studies of urban runoff, and is believed to leach from house paint, PVC and treated timber structures. Few updated studies of organotins in wastewater exist, but because organotins are still used in products other than antifouling paints, their occurrence in wastewater is possible (Cornelissen et al. 2008; Flanagan et al. 2021).

Marine paints with organotins were banned completely in Canada in 2002 and globally in 2008 by the International Convention on the Control of Harmful Anti-fouling Systems on Ships, but continue to be used in parts of the world (ECCC 2020b). Organotins are still used for other applications, including industrial and agricultural biocides, as wood preservatives, in the production of polyurethane foams and silicone, and as PVC stabilizers. Some organotins are currently in use in Canada (Health Canada 2010c).

The physical and chemical properties of organotins vary widely with the number and type of organic groups attached to the tin atom. Water solubility of organotins is inversely related to the number and molecular weight of the bonded organic groups and most commercially used organotins show high affinity to aquatic sediments. Factors affecting organotins' sorption to sediment include organic content, particle size, pH, salinity, redox condition, and iron oxide conditions. Organotins can be degraded through abiotic and biotic methylation and demethylation. TBT's degradation products dibutyltin and monobutyltin are considerably less toxic than TBT. TBT has very long half-life in sediments, estimated to 10-90 years depending on light penetration and anoxic conditions, and is considered persistent. TBT is accumulated by all taxa that have been examined, including fish, mammals, and humans. Mammals, birds, crustaceans, and fish seem to be able to break down TBT and the risk for biomagnification is therefore limited (De Carvalho Oliveira and Santelli 2010; Hall and Bushong 1996).

Many organotins are toxic, hence their widespread use as biocides. However, the toxicity of different organotin compounds varies greatly and is related to the number and type of attached organic groups. TBT is extremely toxic toward nontarget organisms and is the reason behind the destruction of many seabed ecosystems. TBT is considered one of the most toxic contaminants introduced by humankind to the marine environment. Because of its toxicity, much of the research performed on organotins has focussed on TBT. TBT causes chronic and acute effects on many aquatic organisms and is a hormone disruptor with effects such as imposex (acquisition of male sex organs by female snails), and is also believed to be neurotoxic, cancerogenic, obesogenic, and have effects on the immune system (Hall and Bushong 1996; US EPA 2003). TBT and dibutyltin have been identified as contaminants of minor concern to SRKW (ECCC 2020a).

Only limited data are available on the levels at which humans are exposed to organotins through their diet. TBT and its metabolites dibutyltin and monobutyltin have been detected in human blood and in the liver and is believed to be correlated to seafood consumption. There are few toxicity studies concerning the effects of organotins compounds on humans (U.S. Department of Health & Human Services 2005).

2.3.6 Brominated Flame Retardants

Brominated flame retardants (BFRs) are human-made chemicals added to materials to prevent the start or slow the growth of fire. Flame retardants are commonly used in plastics, textiles, and electrical/electronic equipment (Daso et al. 2010; Health Canada 2019a).

Polybrominated diphenyl ethers (PBDEs) have been used widely as BFRs since the 1970s and have attracted the most attention because of their toxicity potential and persistence in the environment. PBDEs are covered in a separate technical report (Braig et al. 2021).

Hexabromocyclododecane (HBCD) has been produced since the 1960s and is mainly used in thermal insulation foams for the construction industry as well as in textiles. Tetrabromobisphenol A (TBBPA) and

its derivatives are the most widely used BFRs in the world. TBBPA is mainly used in epoxy and polycarbonate resins and/or electrical and electronic equipment such as computers, automotive and consumer electronics applications. BFRs are released to the environment during preparation, use, and disposal of products containing the chemicals. Because BFRs are found in many household products, the compounds can reach the environment through wastewater effluents. Waste incineration and landfilling are known sources of BFRs but are not suspected major pathways of BFRs to Burrard Inlet. Studies of BFRs in urban matrices including stormwater are generally lacking, but TBBPA and HBCD are known to leach from products during use and both compounds have been detected in rainwater (Flanagan et al. 2021; Peters 2003).

Studies of the fate of HBCD and TBBPA in WWTPs are limited but results indicate that approximately 70% of the incoming mass is removed, mainly through sorption to sludge. Both TBBPA and HBCD appear to degrade during anaerobic digestion of sewage sludge (Gerecke et al. 2006; U.-J. Kim and Oh 2018).

HBCD released to the marine environment is expected to partition onto particulate matter and sediment because of its hydrophobic nature. HBCD has a very long half-life in sediment, estimated to years or decades, and is deemed persistent in the environment. Because of its extensive use and persistence, HBCD is ubiquitous in the environment and humans. HBCD has a strong potential to bioaccumulate and because high concentrations of HBCD have been found in top predators, such as marine mammals and birds of prey, it is suggested to biomagnify in food chains (Environment Canada and Health Canada 2011b). HBCD has been identified as a contaminant of medium concern to SRKW and Chinook salmon (ECCC 2020a). HBCD produces adverse effects in a variety of aquatic organisms including algae, fish, and invertebrates. It is acutely toxic to fish embryos and imposes several chronic effects including altered thyroid and liver status, protein metabolism, neurotoxicity, and oxidative stress. Studies show that HBCD also has endocrine disrupting and reproductive effects on aquatic organisms (Daso et al., 2010; ECCC, 2016; Kodavanti and Loganathan, 2014). In Canada, the import, manufacture, use, sale and offer for sale of HBCD, and certain products containing HBCD, is prohibited since 2017 through the Prohibition of Certain Toxic Substances Regulations (ECCC 2020c).

TBBPA is manufactured by bromination of BPA and TBBPA can degrade into BPA in the environment (Wei et al. 2018). Despite its extensive use, studies of TBBPA's occurrence in the environment are limited compared to data available for other BFRs, but existing data indicate that TBBPA is frequently detected in the environment. TBBPA has low to moderate water solubility, low vapour pressure, and a moderately high log K_{ow} estimated to be 4.5. When released into water, TBBPA is expected to adsorb to suspended solids and sediment and partition to the lipid fraction of biota. At pH levels found in marine waters, TBBPA ionizes to some degree, leading to a lower log K_{ow} than in its neutral form (Environment Canada and Health Canada 2013). TBBPA does not appear to bioaccumulate. Its potential to persist in the environment has been debated but it meets the persistence criteria in water, soil, sediment and air as defined in the *Persistence and Bioaccumulation Regulations* under CEPA 1999 (EFSA Panel on Contaminants in the Food Chain 2011; Environment Canada and Health Canada 2013). Toxicological studies of TBBPA are generally lacking but existing studies indicate that TBBPA may cause acute toxicity, endocrine disruption, immunotoxicity, neurotoxicity, and effects on kidney and livers in animals (Yu et al. 2019). Adverse effects on survival, reproduction and development of aquatic organisms, including amphibians, fish, and bivalves, have been observed at very low concentrations (Environment Canada and Health Canada 2011a).

Because the use of legacy BFRs, including PBDEs and HBCD, have been restricted due to adverse effects on the environment, novel brominated flame retardants are being developed to replace the legacy BFRs. Hexabromobenzene (HBB) is an example of a novel BFR that has been detected in indoor air, food including seafood, as well as in humans (Xiong et al. 2019; Zuiderveen, Slootweg, and de Boer 2020). HBB has a high log K_{ow} at 6.1 and can be assumed to partition to particulate matter and sediment once

released into the marine environment. There are limited data on the environmental behaviour, but based on its chemistry, HBB is suspected to be hazardous to the aquatic environment, bioaccumulate, and persist in the environment (EFSA Panel on Contaminants in the Food Chain 2012). Toxicity data for novel BFRs are generally lacking but existing studies indicate that several novel compounds may cause adverse effects, including hormone disruption and endocrine disruption, genotoxicity, and behavioral alterations (Xiong et al. 2019; Zuiderveen, Slootweg, and de Boer 2020).

Humans are exposed to BFRs via inhalation and ingestion of contaminated dust in homes, offices and cars, from using certain products, and through their diet, where fish and shellfish are major contributors (Daso et al. 2010). The Government of Canada has concluded that TBBPA and HBCD are not harmful to human health at current levels of exposure (Health Canada 2013, 2019b).

2.3.7 Perfluoroalkyl and Polyfluoroalkyl Substances

Perfluoroalkyl and Polyfluoroalkyl Substances (PFAS) consist of partially or fully fluorinated carbon chains with different functional groups attached. Perfluoroalkyl substances are fully saturated with fluorine, whereas the carbon chain in polyfluoroalkyl substances is partially saturated. The carbon-fluoride bond is very strong, and the molecules are chemically very stable with long half-lives and therefore persistent in the environment (Ahrens and Bundschuh, 2014).

PFAS have been in use since the 1940s as surface treatment chemicals, polymerization aids, and surfactants. Many PFAS are resistant to grease, oil, water, and heat, and have extensively been used in stain- and water-resistant applications such as fabrics and carpeting, cleaning products, paints, and fire-fighting foams. Some PFAS are also used in cookware (e.g., Teflon), food packaging, and food processing equipment (Glüge et al. 2020).

Perfluorooctanoic acid (PFOA) and perfluorooctanesulphonate (PFOS) are historically two of the most used PFAS worldwide but have been largely phased out in many parts of the world (Department of the Environment and Department of Health 2022). In Canada, PFOS, PFOA, and long-chain (C_9-C_{20}) perfluorocarboxylic acids (PFCAs) are prohibited through regulations. However, there are more than 4700 possible PFAS compounds to replace restricted ones (Health Canada 2021a).

Several studies have reported low removal efficiencies for PFAS in WWTPs, and concentrations of perfluoroalkyl acids (PFAAs) increase after wastewater treatment, indicating that degradation of PFAS to PFAA occur. Biodegradation appears to be the primary mechanism responsible for the transformation of PFAS precursors to PFAAs. As conventional WWTP processes are not effective, advanced processes such as adsorption using ion exchange resins, electrochemical degradation, and nanofiltration are needed to remove PFAS from wastewater (Lenka, Kah, and Padhye 2021).

PFAS can enter stormwater through leaching from consumer products, industrial activities, and through atmospheric deposition and have been detected in both urban and rural stormwater at a wide range of concentrations. Applications of firefighting foams are point sources of PFAS (Saifur and Gardner 2021).

Given the many different PFAS compounds, they exhibit a wide range of different physical and chemical characteristics affecting their environmental fate. PFAS are generally stable in the environment and resist typical environmental degradation processes, resulting in the compounds' persistence in the environment. Several polyfluorinated substances can be partially degraded via several different biological and abiotic mechanisms; however, and typically form PFAAs, which are not further degraded under ambient environmental conditions. PFAS most frequently detected in the environment typically have a hydrophobic tail and a polar and hydrophilic head. In general, shorter PFAS tend to partition to the water phase and longer PFAS to particles and sediments. The wide distribution of PFAS in higher trophic level organisms, such as predatory fish (e.g., salmon) and birds (e.g., bald eagles) strongly suggest that they are bioaccumulative and biomagnify. The bioaccumulation and biomagnification of

PFAS appear to vary with physical-chemical properties. In contrast to many other persistent organic chemicals, PFAS tend to partition to proteins rather than lipids in biota (Ahrens and Bundschuh 2014; Saifur and Gardner 2021).

PFAS are ubiquitous in the environment and although concentrations of single PFAS compounds may be too low to cause adverse effects, mixtures of these compounds can be of significant environmental concern. PFAS may induce several toxic effects in fish, including disturbance of metabolism, reproduction disruption, oxidative stress, and developmental toxicity (Ahrens and Bundschuh 2014; Lee et al. 2020). PFOS and PFOA have been identified as contaminants of major concern to SKRW and Chinook salmon (ECCC 2020a).

PFAS have several health effects on humans including liver and kidney disease, decreased fertility, developmental effects or delays in children, increased risk of some cancers as well as obesity, and impaired immune response (Fenton et al. 2021).

2.4 1990 Provisional Water Quality Objectives for CECs

Among the selected CECs, there were Water Quality Objectives developed in 1990 only for tributyltin (TBT, a type of organotin) in Burrard Inlet.

The 1990 Burrard Inlet water quality objectives for TBT in water are summarized in Table 1. They were set for protection of marine aquatic life from chronic toxicity and were based on an advisory level set by the US EPA (Nijman and Swain 1990).

Table 1: 1990 Provisional Water Quality Objectives for Tributyltin.

Sub-basin	False Creek	Outer Harbour	Inner Harbour	Central Harbour	Port Moody Arm	Indian Arm
Water	0.01 µg/L maximum	0.01 µg/L maximum	0.01 µg/L maximum	N/A	0.01 µg/L maximum	0.01 µg/L maximum
Sediment	N/A					
Tissue	N/A					

3. WATER QUALITY ASSESSMENT

3.1 Benchmarks Used in this Assessment

Benchmarks are points of reference against which water, sediment and tissue concentrations can be assessed and were used to screen available data for potential acute and chronic effects and to inform the derivation of proposed objectives for CEC levels in Burrard Inlet. Canadian guidelines[3] for the protection of water values were used as screening benchmarks where available. The following potential sources of screening benchmarks were considered:

1. BC Approved Water Quality Guidelines (BC WQGs) (BC ENV, 2021a);
2. BC Working Water Quality Guidelines (BC WWQGs, also includes sediment and tissue guidelines) (BC ENV, 2021b);
3. Calculated tissue screening values protective of human health (ENV and HLTH 2021);
4. Guidelines from other Canadian jurisdictions (e.g., CCME and ECCC); and

[3] Water quality guidelines are science-based numeric concentrations or narrative statements that are recommended to protect various water values, such as aquatic life, wildlife and their habitats, and recreation.

5. Benchmarks from other jurisdictions in the absence of Canadian guidelines.

The 1990 objective for tributyltin (i.e., the only previous objective for CECs) was not used as a benchmark because the science on impacts has evolved in the last thirty years.

The screening benchmarks used for the data assessment in this report are summarized in Table 2, together with the values that are most sensitive to each CEC in different media.

Because guidelines for CECs in tissue were not available, human-health based screening values (SVs) for fish and shellfish tissue were derived. Human-health based SVs for fish and shellfish tissue were derived from Health Canada toxicological reference values and risk assessment methodologies. The BC method for derivation of SVs for contaminants in fish tissue is described in WLRS (2023) with a Burrard Inlet specific context in ENV and HLTH (2021). Screening values are defined as conservative threshold values against which contaminant concentrations in fish (including finfish and shellfish) tissue can be compared and assessed for potential risks to human health. Fish and shellfish in this report refer to country foods, that is, foods produced in an agricultural (not for commercial sale) backyard setting or harvested through hunting, gathering or fishing activities (Health Canada 2010a).

Three SVs were used in the data assessment to examine multiple levels of protection for three receptors. An allocation factor of 0.2 was used in the calculation to reflect the percentage of each CEC assumed to come from country foods (in this case, wild seafood). Three SVs were selected to capture a range of potential fishers. The most conservative SV is for a toddler from a subsistence fisher population while the adult subsistence fisher and adult recreational fisher are less conservative. In the case of a carcinogen, the most sensitive receptor is an adult from a subsistence fishing or Indigenous population, as the SV is based on a lifetime of exposure.

Table 2: Screening Benchmarks for CECs in Water, Sediment, and Tissue Used in this Assessment

Sample Type	Screening Benchmark	Value	Reference
Alkylphenols			
Nonylphenol and its Ethoxylates in Water (toxic equivalent[1])	0.7 µg/L	Marine aquatic life	BC WWQG (CCME 2002b)
Nonylphenol and its Ethoxylates in Sediment[2]	Lower ISQG (threshold effects level): 1.0 µg/g dry weight Upper SQG (probable effects level): N/A	Marine aquatic life	BC WWQG (CCME 2002a)
Nonylphenol and its Ethoxylates in Tissue	0.018 µg/g wet weight	Human consumption of finfish and shellfish	(ENV and HLTH 2021)
Bisphenols			
BPA in Water	0.9 µg/L	Marine aquatic life	BC WWQG (ECCC, 2018)
BPA in Sediment[2]	Lower ISQG (threshold effects level): 0.025 µg/g dry weight Upper SQG (probable effects level): N/A	Marine aquatic life	BC WWQG (ECCC, 2018)
BPA in Tissue	Do not detect and no increase from existing levels[4]		
Phthalates			
DEHP in Water	0.37 µg/L	Human consumption of finfish and shellfish	Human Health for the consumption of Organism Only (µg/L) (US EPA 2015)
DEHP in Sediment[2]	Lower ISQG (threshold effects level): 0.47 µg/g dry weight Upper SQG (probable effects level): 0.78 µg/g dry weight	Marine aquatic life	BC WWQG (DOE, 2013)
BBP in Water	0.1 µg/L	Human consumption of finfish and shellfish	Human Health for the consumption of Organism Only (µg/L) (US EPA 2015)
BBP in Sediment[2]	Lower ISQG (threshold effects level): 0.049 µg/g dry weight Upper SQG (probable effects level): 0.64 µg/g dry weight	Marine aquatic life	BC WWQG (DOE, 2013)

[4] The European Food Safety Authority has re-evaluated the risks of BPA in food and proposed a tolerable daily intake of 0.04 nanograms per kilogram of body weight per day, compared to the previous interim standard of 4 micrograms (4,000 nanograms) per kilogram per day (EFSA 2021). A screening value for BPA has not been calculated and applied, as it would be orders of magnitude below the current detection limit of 0.002 µg/g wet weight.

Sample Type	Screening Benchmark	Value	Reference
DEP in Water	600 µg/L	Human consumption of finfish and shellfish	Human Health for the consumption of Organism Only (µg/L) (US EPA 2015)
DEP in Sediment[2]	Lower ISQG (threshold effects level): 0.61 µg/g dry weight Upper SQG (probable effects level): 1.10 µg/g dry weight	Marine aquatic life	BC WWQG (DOE, 2013)
DMP in Water	2000 µg/L	Human consumption of finfish and shellfish	Human Health for the consumption of Organism Only (µg/L) (US EPA 2015)
DMP in Sediment[2]	Lower ISQG (threshold effects level): 0.53 µg/g dry weight Upper SQG (probable effects level): 0.53 µg/g dry weight	Marine aquatic life	BC WWQG (DOE, 2013)
DnOP in Sediment[2]	Lower ISQG (threshold effects level): 0.58 µg/g dry weight Upper SQG (probable effects level): 45 µg/g dry weight	Marine aquatic life	BC WWQG (DOE, 2013)
DnBP in Water	30 µg/L	Human consumption of finfish and shellfish	Human Health for the consumption of Organism Only (µg/L) (US EPA 2015)
DnBP in Sediment[2]	Lower ISQG (threshold effects level): 2.2 µg/g dry weight Upper SQG (probable effects level): 17 µg/g dry weight	Marine aquatic life	BC WWQG (DOE, 2013)
6PPD and 6PPD-quinone			
6PPD/6PPD-quinone in Water/Sediment/ Tissue	No data available	Marine aquatic life	N/A
Organotins			
TBT in Water	0.001 µg/L	Marine aquatic life	BC WWQG (CCME 1999)
Organotins (incl. TBT) in Tissue	0.0088 µg/g wet weight	Human consumption of finfish and shellfish	ENV and HLTH 2021
Brominated Flame Retardants			
HBCD in Water	0.56 µg/L	Marine aquatic life	BC WWQG (ECCC, 2016b)
HBCD in Sediment	Lower ISQG (threshold effects level): 1.6 µg/g dry weight	Marine aquatic life	BC WWQG (ECCC, 2016b)

Sample Type	Screening Benchmark	Value	Reference
	Upper SQG (probable effects level): N/A		
TBBPA in Water	3.1 µg/L	Marine aquatic life	ECCC, 2016b
TBBPA in Sediment	0.6 µg/g dry weight	Marine aquatic life	ECCC, 2016b
HBB in Tissue	0.0070 µg/g wet weight	Human consumption of finfish and shellfish	ENV and HLTH 2021
Perfluoroalkyl and Polyfluoroalkyl Substances			
PFOA in Tissue	0.0007 µg/g wet weight	Human consumption of finfish and shellfish	ENV and HLTH 2021
PFOS in Water	3.4 µg/L	Freshwater aquatic life	BC WWQG (ECCC, 2018b)
PFOS in Tissue	0.0021 µg/g wet weight	Human consumption of finfish and shellfish	ENV and HLTH 2021

[1.] The BC WWQG is for the total concentration of nonylphenols and nonylphenols equivalents, which is calculated as the concentration of the mixture of nonylphenolic compounds expressed as the toxic equivalent of nonylphenolic compounds.
[2.] Concentrations are expressed as µg/g sediment containing 1% organic carbon. A guideline expressed as µg/g is based on the sediment as a whole and does not require adjustment for organic carbon content. Adjustments to guidelines are required when they are expressed in terms of the sediment containing 1% organic carbon. For sediments with organic carbon other than 1%, an adjustment in guidelines should be made by multiplying the guideline by the % organic carbon content of the sediment.

3.2 Data Sources

Data for CEC levels in Burrard Inlet were gathered from several studies and monitoring programs and a summary of the datasets used for this assessment is presented in Table 3 (more information on available water quality data is found in Rao et al. [2019]). Although other datasets containing CEC sampling data may exist, the priority datasets were found to be the best available data for assessing the status of CECs within Burrard Inlet within the constraints of the project. It should be noted that, at the time of writing, there were no marine monitoring data on 6PPD, 6PPD-quinone and phthalates available for Burrard Inlet matrices and hence the compounds are not included in the assessment. At the time of writing, stream data on vehicular and stormwater-related contaminants, including 6PPD and 6PPD-quinone, were being actively collected by Fisheries and Oceans Canada (Brown et al. 2022); a preliminary summary of these data, showing their entry into Burrard Inlet, is provided in Appendix B.

Table 3. Studies and Monitoring Programs Contributing Data Used for the Assessment

Source	Program, Years	Parameter	No. of Observations	No. of Sites	Sampling Frequency	Years with Data
BC ENV	Provincial Water Quality Objectives Attainment Monitoring	Nonylphenol and its Ethoxylates	8 sediment	7	Single event	2020
		BPA	8 sediment	7	Single event	2020
		Organotins	None	NA	NA	NA
		TBT	4 marine water	1	Irregular	1993
		HBCD	None	NA	NA	NA
		TBBPA	None	NA	NA	NA
		HBB	11 sediment	10	Single event	2020
		PFOA	8 sediment	7	Single event	2020
		PFOS	8 sediment	7	Single event	2020
Metro Vancouver	Burrard Inlet Ambient Monitoring Program	Nonylphenol and its Ethoxylates	454 marine water 21 sediment	8 marine water 7 sediment	Annually 3 marine water samples from top and bottom of water column per year at 7 sites but 5 samples in 2011 and 3 samples in 2008 at 1 additional site Single composite sediment samples	2007-2016
		BPA	None	NA	NA	NA
		Organotins	None	NA	NA	NA
		TBT	None	NA	NA	NA
		HBCD	None	NA	NA	NA
		TBBPA	None	NA	NA	NA
		HBB	21 sediment	7	Single event	2015
		PFOA	None	NA	NA	NA
		PFOS	None	NA	NA	NA
Ocean Wise	Pollution Tracker	Nonylphenol and its Ethoxylates	10 mussel tissue 20 sediment	12 mussel tissue 15 sediment	Single composites of 200 mussel specimens Single event for sediment	2015, 2016, 2019
		BPA	3 mussel tissue 5 sediment	3 mussel tissue 5 sediment	Single composites of 200 mussel specimens Single event for sediment	2015
		Organotins	9 mussel tissue 15 sediment	9 mussel tissue 8 sediment	Single composites of 200 mussel specimens Single event for sediment	2015, 2016, 2018
		TBT	9 mussel tissue 15 sediment	9 mussel tissue 8 sediment	Single composites of 200 mussel specimens	2015, 2016, 2018

Source	Program, Years	Parameter	No. of Observations	No. of Sites	Sampling Frequency	Years with Data
					Single event for sediment	
		HBCD	7 mussel tissue 7 sediment	7 mussel tissue 7 sediment	Single composites of 200 mussel specimens Single event for sediment	2015, 2019
		TBBPA	3 mussel tissue 8 sediment	3 mussel tissue 8 sediment	Single composites of 200 mussel specimens Single event for sediment	2015
		HBB	3 mussel tissue	2	Single composites of 200 mussel specimens	2016
		PFOA	4 mussel tissue 9 sediment	4 mussel tissue 9 sediment	Single composites of 200 mussel specimens Single event for sediment	2015
		PFOS	5 mussel tissue 10 sediment	5 mussel tissue 10 sediment	Single composites of 200 mussel specimens Single event for sediment	2015, 2016, 2019
No data		BPA Organotins HBCD TBBPA PFOA PFOS	Marine water	No data		
		DEHP BBP DEP DMP DnOP DNBP 6PPD and 6PPD-quinone	Marine water Sediment Tissue			

3.3 Assessment Results

Available monitoring data were compared to screening benchmarks with a summary provided in Table 4. Because of the very limited dataset for CECs, observations are presented for Burrard Inlet as a whole, and not for each sub-basin. Because of variations in the sampling and analytical methods and distribution of sites, results from each monitoring program are discussed separately.
Detection limits were frequently variable and above the screening benchmarks, which created challenges for providing comparisons. Where the highest detection limit within a dataset for a CEC was greater than a screening benchmark, this was noted as a limitation. Non-detects were not included in the calculation of summary statistics or count of benchmark exceedances as the detection limits were frequently an order of magnitude or higher than the screening benchmarks.

Because of the wide range of detection limits, samples that were below detection limits were excluded from the summary of mean, minimum, and maximum concentrations. Field replicates were averaged prior to the assessment. Key observations for detection frequency, exceedances, and maximum observed CEC concentrations are described by monitoring program. Overall summaries of status and observations for marine water, sediment and tissue are provided alongside the rationale for the

proposed water quality objectives. As a conservative approach, analytical data were not blank-corrected.

Table 4. Summary of Monitoring Data versus Screening Benchmarks for Contaminants of Emerging Concern

Parameter	Media	Source	Count	% Detection	Range of Detection Limits	% Benchmark Exceedance	Detected Levels		
							Minimum	Maximum	Mean
					µg/L or µg/g	% > SV, µg/L or µg/g	µg/L or µg/g	µg/L or µg/g	µg/L or µg/g
Nonylphenol and its Ethoxylates[5]	Marine Water	Metro Vancouver	454	7%	0.01 to 2 *	0.4% > 0.7	0. 012	5.2	0.28
	Sediment	Metro Vancouver	21	100%	0.01 to 0.2 *	0% > 1.0	0.05	0.18	0.09
		BC ENV	8	38%	0.063 to 0.5 *	0% > 1.0	0.05	0.09	0.11
		Pollution Tracker	20	100%	0.0002 to 0.0006 *	0% > 1.0	0.004	0.161	0.062
	Tissue	Pollution Tracker	15	100%	0.0004 to 0.0009 *	87% > 0.018	0.014	0.048	0.024
BPA	Sediment	BC ENV	8	13%	0.05	0% > 0.09	0.07	0.07	0.07
		Pollution Tracker	5	0%	0.4 to 0.5	NA	NA	NA	NA
	Tissue	Pollution Tracker	2	0%	0.002	0% detected	NA	NA	NA
Organotins	Sediment	Pollution Tracker	15	80%	0.001	NA	0.006	0.134	0.04
	Tissue	Pollution Tracker	9	0%	0.01	NA	NA	NA	NA
TBT	Marine Water	BC ENV	4	0%	0.0023 to 0.0031	NA	NA	NA	NA
	Sediment	Pollution Tracker	15	87%	0.001	NA	0.001	0.031	0.009
	Tissue	Pollution Tracker	9	0%	0.01	NA	NA	NA	NA
HBCD	Sediment	Pollution Tracker	7	57%	0.0001	0% > 1.6	0.00017	0.00753	0.0026
	Tissue	Pollution Tracker	7	86%	0.00013 to 0.00020	NA	0.000108	0.000198	0.00135
TBBPA	Sediment	Pollution Tracker	8	0%	0.0005 to 0.0022	NA	NA	NA	NA
	Tissue	Pollution Tracker	3	0%	0.002	NA	NA	NA	NA
HBB	Sediment	BC ENV	11	91%	0.0000002	NA	0.0004	0.0000013	0.00006
		Metro Vancouver	21	100%	0.000000097	NA	0. 0000162	0. 000364	0.0000783

[5] Minimum, maximum and mean levels of nonylphenol, octylphenol, and their ethoxylates are expressed in terms of toxic equivalencies (TEQ) relative to nonylphenol. For more information on nonylphenol TEQ, see https://ccme.ca/en/res/nonylphenol-and-its-ethoxylates-canadian-sediment-quality-guidelines-for-the-protection-of-aquatic-life-en.pdf

Parameter	Media	Source	Count	% Detection	Range of Detection Limits	% Benchmark Exceedance	Detected Levels		
							Minimum	Maximum	Mean
					µg/L or µg/g	% > SV, µg/L or µg/g	µg/L or µg/g	µg/L or µg/g	µg/L or µg/g
	Tissue	Pollution Tracker	3	0%	0.000029 to 0.000033	0% > 0.0007	NA	NA	NA
PFOA	Sediment	BC ENV	8	0%	0.0001	NA	NA	NA	NA
		Pollution Tracker	9	0%	0.00009 to 0.001	NA	NA	NA	NA
	Tissue		4	0%	0.0005	NA	NA	NA	NA
PFOS	Sediment	BC ENV	8	0%	0.0005	NA	NA	NA	NA
		Pollution Tracker	9	11%	0.00017 to 0.00148	NA	0.00147	0.00147	0.00147
	Tissue		4	0%	0.0001 to 0.001	NA	NA	NA	NA

*Detection limit range reported for all sub-types including nonylphenol, nonylphenol ethoxylate, nonylphenol diethoxylate, and nonylphenol triethoxylate, and nonylphenol + octylphenol

3.3.1 Marine Water

Of the identified CECs in this review, only nonylphenol and its ethoxylates and TBT have been measured in water samples collected by any of the key monitoring programs. As mentioned above, other CECs associated with stormwater and vehicular traffic, including 6PPD and 6PPD-quinone, have recently been recorded in streams around Burrard Inlet (Brown et al. 2022); a summary of those data is presented in Appendix B.

<u>Nonylphenol and its ethoxylates</u>

- The Metro Vancouver Burrard Inlet Monitoring Program is the only program that has measured nonylphenol and its ethoxylates in marine water. Samples of both the top and bottom of the water column were collected from seven ambient sites on three different dates each year between 2007 and 2016, except for in 2011 when samples were collected on five different dates and in 2008 when samples were collected on three different dates from eight sites. A total of 454 unique samples were collected. In 2007 and from 2009 to 2016, nonylphenol + octylphenol as well as nonylphenol diethoxylate were measured in the samples. In 2008 Nonylphenol, nonylphenol monoethoxylate, and nonylphenol triethoxylate were each measured in the samples. Overall, there were 967 unique measurements of the various nonylphenol compounds.
- Of the 967 unique measurements, 38 measurements (3%) were above detection limits. Of the 454 unique samples, 33 samples had measurements above detection limits. The detection limit for nonylphenol was 0.048 µg/L, the detection limit for nonylphenol + octylphenol ranged from 0.01 to 0.018 µg/L, the detection limit for nonylphenol monoethoxylate was 0.048 µg/L, the detection limit for nonylphenol diethoxylate ranged from 0.02 to 0.96 µg/L, and the detection limit for nonylphenol triethoxylate was 2 µg/L.
- Among the measurements that were above detection limits include:
    - Ten detects of nonylphenol ranging between 0.048 µg/L and 0.2 µg/L with average concentration of 0.09 µg/L. All detects occurred in 2008 and detects occurred in every basin on at least on occasion, excluding False Creek, which is not included in this monitoring program.

    - Twenty-one detects of nonylphenol + octylphenol ranging between 0.012 µg/L and 0.535 µg/L with average concentration of 0.10 µg/L. Detects occurred in most years between 2007 to 2015 and detects occurred in every basin on at least on occasion, excluding False Creek, which is not included in this monitoring program.
    - One detect of nonylphenol diethoxylate at 2 µg/L.
    - Two detects of nonylphenol monoethoxylate at 0.34 µg/L and 2 µg/L.
    - Two detects of nonylphenol triethoxylate at 1 µg/L and 7 µg/L.
- The toxic equivalency factor (relative to nonylphenol) in water is 1 for octylphenol and is 0.5 for nonylphenol monoethoxylate, nonylphenol diethoxylate, and nonylphenol triethoxylate (BC ENV, 2017). These factors were applied to the detected measurements to calculate total toxic equivalencies relative to nonylphenol. Among detected concentrations, the minimum toxic equivalency was 0.012, the maximum toxic equivalency was 5.2, and the mean toxic equivalency was 0.28. Overall, 0.4% of all unique 454 samples exceeded the screening benchmark.

Tributyltin

- The ENV Provincial Water Quality Objectives Attainment Monitoring measured TBT in marine water at one site (False Creek Marina, E216034) on four occasions between August 18 and September 15, 1993.. The detection limits for TBT in ENV's monitoring program ranged between 0.0023 and 0.0031 µg/L and TBT was below detection limits in all (four) samples.

3.3.2 Sediment

Of the identified CECs in this review, nonylphenol and its ethoxylates, BPA, organotins, TBT, PFOA, PFOS, TBBPA, and HBB have been measured in sediment collected by the key monitoring programs. Metro Vancouver has measured nonylphenol and its ethoxylates, and HBB in sediment. ENV has measured nonylphenol and its ethoxylates, BPA, PFOA, PFOS, and HBB in sediment. Pollution Tracker has measured nonylphenol and its ethoxylates, and BPA, organotins, TBT, PFOA, PFOS, and TBBPA in sediment.

Nonylphenol and its ethoxylates

- Metro Vancouver collected nonylphenol and its ethoxylates samples in 2008. Twenty-one samples were collected and analyzed across seven monitoring stations (three samples at each location) and nonylphenol and its ethoxylates were detected in all samples, with an average concentration of 0.09 µg/g dry weight or 0.035 µg/g dry weight normalized to 1% total organic carbon. The highest detection limit was 0.2 µg/g dry weight and there have been no exceedances of the 1 µg/g dry weight benchmark.
- ENV collected samples of nonylphenol and its ethoxylates in 2020. Eight samples were collected and analyzed across seven monitoring stations (one sample at each location with one duplicate sample). Nonylphenol and its ethoxylates were detected in three (38%) of the samples, with an average concentration of 0.0.08 µg/g dry weight, normalized to 1% total organic carbon. The detection limits ranged from 0.025 µg/g and 0.5 µg/g dry weight (not normalized), creating challenges for interpretation of the results, though there were no exceedances of the 1 µg/g dry weight benchmark.
- PollutionTracker collected nonylphenol and its ethoxylates samples in 2015, 2016, and 2019. Twenty samples were collected and analyzed across 15 monitoring stations. Nonylphenol and its ethoxylates were detected in all samples, with an average concentration of 0.062 µg/g dry weight or 0.039 µg/g dry weight normalized to 1% total organic carbon. The highest detection limit was 0.0006 µg/g dry weight and there have been no exceedances of the 1 µg/g dry weight benchmark.

BPA

- ENV collected BPA samples in 2020. Eight samples were collected and analyzed across seven monitoring stations (one sample at each location with one duplicate sample). BPA was detected in one (13%) of the samples, at a concentration of 0.07 µg/g dry weight or 0.021 µg/g dry weight normalized to 1% total organic carbon. This sample did not exceed the screening benchmark (0.025 µg/g dry weight normalized to 1% total organic carbon).
- PollutionTracker collected one BPA sample each from five monitoring locations in 2015. Detection limits ranged from 0.4 to 0.5 µg/g dry weight and no samples were above the detection limit. The detection limits were significantly higher than the screening benchmark (0.4 µg/g dry weight vs 0.025 µg/g dry weight).

Organotins

- PollutionTracker collected one organotins (mono-, di-, tri- organotin and mono-, di-, tri-organotin chloride) sample each from eight monitoring locations in 2015 and from seven monitoring locations in 2018. Organotins were detected in 87% of the 15 samples (detection limit 0.001 µg/g dry weight). The mean total organotins concentration across all sites was 0.04 µg/g dry weight.
- No benchmark was identified for total organotins in sediment.

Tributyltin

- PollutionTracker collected one TBT sample each from eight monitoring locations in 2015 and from seven monitoring locations in 2018. TBT was detected in 87% of the 15 samples (detection limit 0.001 µg/g dry weight). The mean TBT concentration across all sites was 0.009 µg/g dry weight.
- No benchmark was identified for TBT in sediment.

HBCD

- PollutionTracker collected one HBCD sample each from seven monitoring locations in 2015 or 2019. HBCD was detected in four (57%) of seven samples (detection limit 0.0001 µg/g dry weight), with an average concentration of 0.0026 µg/g dry weight. The detection limits were well below the screening benchmark level at 1.6 µg/g dry weight.

TBBPA

- PollutionTracker collected one TBBPA sample each from eight monitoring locations in 2015. TBBPA was below the detection limit in all samples (detection limit range from 0.0005 to 0.002 µg/g dry weight). The detection limits were well below the screening benchmark level at 0.6 µg/g dry weight.

HBB

- Metro Vancouver collected HBB samples in 2008 and 2013. In each year, 21 samples were collected and analyzed across seven monitoring stations (3 samples at each location). HBB was detected in all 21 samples, with an average concentration of 0.00008 µg/g dry weight.
- ENV collected HBB samples in 2020. Eleven samples were collected and analyzed across ten monitoring stations (one sample at each location with one duplicate sample); HBB was detected in ten (91%) of these samples, with an average concentration of 0.00006 µg/g dry weight.
- No benchmark was identified for HBB in sediment.

PFOA

- ENV collected PFOA samples in 2020. Eight samples were collected and analyzed across seven monitoring stations (one sample at each location with one duplicate sample). PFOA was below detection limit (0.0001 µg/g dry weight) in all samples.
- PollutionTracker collected one PFOA sample each from nine monitoring locations in 2015. PFOA was below the detection limit in all samples (detection limits range from 0.00009 to 0.0001 µg/g dry weight).
- No benchmark was identified for PFOA in sediment.

PFOS

- ENV collected PFOS samples 2020. Eight samples were collected and analyzed across seven monitoring stations (one sample at each location with one duplicate sample). PFOS was below the detection limit (0.0005 µg/g dry weight) in all samples.
- PollutionTracker collected one PFOA sample each from nine monitoring locations across 2015 and 2016. PFOA was detected (detection limit range from 0.000017 to 0.00148 µg/g dry weight) in one of the samples (11%) at a concentration of 0.00147 µg/g dry weight.
- No benchmark was identified for PFOS in sediment.

3.3.3 Animal Tissue

Of the identified CECs in this review, only the PollutionTracker program has analyzed CECs in tissue, including, nonylphenol and its ethoxylates, BPA, organotins, TBT, PFOA, PFOS, and TBBPA.

Nonylphenol and its ethoxylates

- PollutionTracker measured nonylphenol and ethoxylates in 15 composite blue mussel tissue samples from ten monitoring locations across 2015, 2016, and 2019. Nonylphenol and ethoxylates were detected in 100% of the samples (detection limits range from 0.0004 to 0.0009 µg/g wet weight). 87% of the samples exceeded the screening benchmark (0.018 µg/g wet weight) with a mean concentration of 0.024 µg/g wet weight.

BPA

- PollutionTracker measured BPA in one composite blue mussel tissue sample from each of two monitoring locations in 2015. BPA was below the detection limit (0.002 µg/g wet weight) in both samples.
- The benchmark for BPA in tissue is ‘do not detect.’

Organotins

- PollutionTracker measured organotins (mono-, di-, tri- organotin chloride) in one composite blue mussel tissue sample from each of nine monitoring locations across 2015 and 2016. Organotins was below the detection limit (0.01 µg/g wet weight) in all samples.
- The screening benchmark for organotins in tissue is 0.0088 µg/g wet weight.

Tributyltins

- PollutionTracker measured TBT in one composite blue mussel tissue sample from each of nine monitoring locations in 2015. TBT was below the detection limit (0.01 µg/g wet weight) in all samples.
- No benchmark was identified for TBT in tissue.

HBCD

- PollutionTracker measured HBCD in one composite blue mussel tissue sample from each of seven monitoring locations in 2015 or 2019. HBCD concentrations were above the detection limit (0.00013 to 0.00020 µg/g dry weight) in six (86%) of the samples, with an average concentration of 0.00135 µg/g dry weight.
- No benchmark was identified for HBCD in tissue.

TBBPA

- PollutionTracker measured TBBPA in one composite blue mussel tissue sample from each of three monitoring locations in 2015. TBBPA was below the detection limit (0.002 µg/g wet weight) in all samples.
- No benchmark was identified for TBBPA in tissue.

HBB

- PollutionTracker measured HBB in one or two composite blue mussel tissue samples from each of two monitoring locations in 2016. HBB was below the detection limit (range from 0.000033 to 0.000155 µg/g wet weight) in all three samples. All detection limits were lower than the screening benchmark of 0.0070 µg/g wet weight.

PFOA

- PollutionTracker measured PFOA in one composite blue mussel tissue sample from each of four monitoring locations in 2015. PFOA was below the detection limit (0.0005 µg/g wet weight) in all samples.
- The screening benchmark for PFOA in tissue is 0.0007 µg/g wet weight. The detection limit is less than the screening benchmark for the PollutionTracker program.

PFOS

- PollutionTracker measured PFOS in one composite blue mussel tissue sample from each of four monitoring locations in 2015. PFOS was below the detection limit (range from 0.0001 to 0.001 µg/g wet weight) in all samples.
- The screening benchmark for PFOS in tissue is 0.0021 µg/g wet weight. The detection limit is less than the screening benchmark for the PollutionTracker program.

3.4 Knowledge Gaps and Research Needs

There is a general lack of monitoring data for high production chemicals with potential negative impacts on the marine environment, such as legacy and novel BFRs, and 6PPD and 6PPD-quinone. Data on the occurrence, fate, persistence, and toxicity toward marine organisms of 6PPD-quinone are lacking although studies are underway. Legacy BFRs and PFAS are being replaced with novel compounds with similar chemical structure and use patterns, but general knowledge about these novel compounds is lacking.

There is also a general lack of knowledge about the occurrence of investigated CECs in Burrard Inlet. Additional monitoring of CECs in water, sediment, and biota (including finfish) is recommended. Existing data for marine water, sediment and tissue samples collected from Burrard Inlet do not provide an adequate number of sample points and do not allow for an assessment of temporal trends in detected concentrations. Sampling should be conducted in both dry and wet seasons for at least three years to account for potential seasonal variability in CEC concentrations associated with variations in WWTP, stormwater, and overflow discharges.

Several analyzed CECs (nonylphenol and its ethoxylates, BPA, organotins, TBT, HBCD, HBB, PFOS) were found at low concentrations in the environment. Lower laboratory detection limits are needed to better understand the inputs of CECs to the marine environment and their potential effects.

Additional studies are required to better understand the long-term chronic effects of CEC exposure in marine biota and humans, including food chain magnification, CEC levels in finfish, and the effects of long-term environmental exposures to transformation products, mixtures of CEC and other environmental contaminants. Many CECs have endocrine disrupting effects but the accumulated impact of exposure to a range of endocrine disruptors is very poorly understood.

4. PROPOSED OBJECTIVES FOR CONTAMINANTS OF EMERGING CONCERN IN BURRARD INLET

4.1 Proposed Objectives

Proposed objectives for CECs are presented in Table 5.

Table 5: Proposed Water Quality Objectives for CECs

Sub-basin	False Creek	Outer Harbour	Inner Harbour	Central Harbour	Port Moody Arm	Indian Arm
All media	**All CECs** Decreasing trend in concentrations					
Water	**Alkylphenols and their Ethoxylates** Nonylphenol and its Ethoxylates: 0.7 µg/L (total toxic equivalent of nonylphenolic compounds)					
	Bisphenols Bisphenol A (BPA): 0.9 µg/L					
	Phthalates Di-methyl phthalate (DMP): 2000 µg/L Di-ethyl phthalate (DEP): 600 µg/L Di-(n)-butyl phthalate (DnBP): 30 µg/L Benzyl butyl phthalate (BBP): 0.1 µg/L Di-(2-ethylhexyl) phthalate (DEHP): 0.37 µg/L Di-n-butyl phthalate (DnOP): Do not detect* and/or decrease in current levels					
	6PPD and 6PPD-Quinone 6PPD and 6PPD-Quinone: Do not detect* and/or decrease in current levels					
	Organotins Tributyltin (TBT): 0.001 µg/L					
	Brominated Flame Retardants Hexabromocyclododecane (HBCD): 0.56 µg/L Tetrabromobisphenol A (TBBPA): 3.1 µg/L Hexabromobenzene (HBB): Do not detect* and/or decrease in current levels					
	Perfluoroalkyl and Polyfluoroalkyl Substances Perfluorooctanoic acid (PFOA): Do not detect* and/or decrease in current levels Perfluorooctanesulfonic acid (PFOS): 3.4 µg/L					
Sediment	**Alkylphenols and their Ethoxylates** Nonylphenol and its Ethoxylates: 1.0 µg/g dry weight (total toxic equivalent of nonylphenolic compounds; adjust objective to site-specific levels of total organic carbon [TOC])					
	Bisphenols BPA: 0.025 µg/g dry weight (adjust objective to site-specific levels of TOC)					
	Phthalates					

Sub-basin	False Creek	Outer Harbour	Inner Harbour	Central Harbour	Port Moody Arm	Indian Arm
	DMP: 0.53 µg/g dry weight (adjust objective to site-specific levels of TOC) DEP: 0.61 µg/g dry weight (adjust objective to site-specific levels of TOC) DnBP: 2.2 µg/g dry weight (adjust objective to site-specific levels of TOC) BBP: 0.049 µg/g dry weight (adjust objective to site-specific levels of TOC) DEHP: 0.47 µg/g dry weight (adjust objective to site-specific levels of TOC) DnOP: 0.58 µg/g dry weight (adjust objective to site-specific levels of TOC)					
	6PPD and 6PPD-Quinone 6PPD and 6PPD-Quinone: Do not detect* and/or decrease in current levels					
	Organotins Do not detect* and/or decrease in current levels					
	Brominated Flame Retardants HBCD: 1.6 µg/g dry weight TBBPA: 0.6 µg/g dry weight HBB: Do not detect* and/or decrease in current levels					
	Perfluoroalkyl and Polyfluoroalkyl Substances PFOA: Do not detect* and/or decrease in current levels PFOS: Do not detect* and/or decrease in current levels					
Tissue	**Alkylphenols and their Ethoxylates** Nonylphenol and its Ethoxylates: 0.018 µg/g wet weight					
	Bisphenols BPA: Do not detect* and/or decrease in current levels					
	Phthalates DMP: Do not detect* and/or decrease in current levels DEP: Do not detect* and/or decrease in current levels DnBP: Do not detect* and/or decrease in current levels BBP: Do not detect* and/or decrease in current levels DEHP: Do not detect* and/or decrease in current levels DnOP: Do not detect* and/or decrease in current levels					
	6PPD and 6PPD-Quinone 6PPD and 6PPD-Quinone: Do not detect* and/or decrease in current levels					
	Organotins TBT: 0.0088 µg/g wet weight Organotins: 0.0088 µg/g wet weight					
	Brominated Flame Retardants HBCD: Do not detect* and/or decrease in current levels TBBPA: Do not detect* and/or decrease in current levels HBB: 0.00070 µg/g wet weight					
	Perfluoroalkyl and Polyfluoroalkyl Substances PFOA: 0.0007 µg/g wet weight PFOS: 0.0021 µg/g wet weight					
*Any objective of 'do not detect' implies non-detection when using best available detection limits.						

4.2 Rationale

Where available toxicological information has been applied to establish quantitative water quality objectives for the CECs outlined in this report, it is proposed to implement those objectives for Burrard

Inlet to be protective of marine aquatic life and human consumption of shellfish and finfish. Where there have been no detects of a given CEC, the objective should be 'do not detect' when using best available detection limits, and/or 'decrease from current levels' when detection limits are adequately low. Any objective of 'do not detect' implies non-detection when using best available detection limits. As more data are gathered, water quality objectives should be re-evaluated.

Where there is insufficient toxicological information and/or monitoring data, in general, a qualitative objective is proposed for a decreasing trend in the concentrations of all CECs in all media over time.

5. MONITORING RECOMMENDATIONS

Sampling for CECs in Burrard Inlet has been limited to date and the analyses have been complicated by the wide range of detection limits among the monitoring programs. The following recommendations are provided to help guide future monitoring programs, and to inform the future revision of water quality objectives for Burrard Inlet:

1. Monitor potential sources of CECs for Burrard Inlet (listed as highest priority to lowest):
   - Include additional CECs that currently lack data (e.g., 6PPD and 6PPD-quinone, phthalates) in routine monitoring and coordinate efforts to avoid duplication of efforts.
   - Evaluate the presence and concentrations of CECs in biosolids, and stormwater runoff.
   - Continue sampling for CECs in Burrard Inlet to inform on spatial and temporal trends. A combination of sampling sites near point sources and at background locations will be essential to better understand the sources of CECs to Burrard Inlet.
   - Monitor CECs in WWTP effluent and the initial dilution zone (IDZ). Detection limits of certain CECs may be too high compared to expected concentrations in the receiving environment; for these compounds, the concentrations in the IDZ may be calculated based on WWTP effluent measurements.

2. Monitor potential CEC accumulation in the food chain:
   - Evaluate CECs in the local marine food web, including finfish, to better understand CECs' potential for biomagnification and impacts on marine top predators.

3. Monitor potential adverse effects on the marine environment:
   - Increase understanding of potential adverse effects on marine biota by conducting long-term toxicity assessments using environmentally relevant concentrations and local species.
   - Based on toxicity test results, conduct human health and ecological risk assessments for CECs.

4. Consider novel CECs in future monitoring:
   - Remain up to date with current toxicological research and adapt monitoring programs to include investigations of novel CECs as required (e.g., BFR and PFAS).
   - Perform screening (non-target analysis) of additional CECs that are potentially occurring in Burrard Inlet. Compounds prioritized for screening could be identified by a knowledgeable group of technical experts and could be revised every two to three years to remain updated with current research.

5. Standardize monitoring methods:
   - Coordinate monitoring methods between agencies that sample water quality in Burrard Inlet.

- Investigate method detection capabilities for local laboratories and support work towards improving detection limits for CECs with detection limits that are greater than available screening benchmarks.

6. MANAGEMENT OPTIONS

The following management options could decrease the effects of CECs on water values in Burrard Inlet:

- Until marine-relevant toxicity data and additional high quality monitoring data are available, management priorities should focus on source control and monitoring, with the goal of reducing concentrations of CECs in water, sediment, and biota over time.
- Implementation of green stormwater infrastructure; for example, stormwater bioretention systems have proven effective at reducing 6PPD-quinone mass loadings to receiving waters by approximately 90% (Rodgers et al. 2023).
- Detailed risk assessments (both human health and ecological) should be completed for any detected CECs whose objectives include 'do not detect', and warnings considered for the most sensitive members of a population.

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APPENDIX A: CALCULATIONS OF SCREENING VALUES FOR CONTAMINANTS OF EMERGING CONCERN RELEVANT TO SEAFOOD CONSUMPTION BY HUMANS

Human health-based tissue screening values for select CECs were calculated from the following equation (see ENV and HLTH [2021] for details) and listed in the tables below. Tolerable daily intakes (TDIs) were obtained from Health Canada (2021b) for organotins, from EPA (2016a,b) for PFOA and PFOS, and from EPA (2005) for HBB. For organotins, the TDI is derived based on findings for immunosuppression in rats as the critical effect. For PFOA, the TDI is derived based on findings for reduced ossification of the proximal phalanges (forelimb and hindlimb) and accelerated puberty in male mice pups as the critical effects. For PFOS, the TDI is derived based on the consistency of the response and of the use of the most sensitive endpoint, developmental toxicity, as the critical effect. For HBB, the TDI is based on the assumption that thresholds exist for certain toxic effects such as cellular necrosis and was derived from studies for induced serum carboxylesterase activity in rats.

$$SV_n = \frac{TDI \times BW \times AF}{IR_{Food} \times RAF_{Oral}} + BC$$

Where:

- SV_n = screening value for a noncarcinogen (µg/g);
- *TDI* = tolerable daily intake (µg/kg BW/day); the contaminant dose deemed safe or acceptable;
- *BW* = body weight (kg);
- *AF* = allocation factor; the fraction of the contaminant allocated to come from country foods; an AF of 0.2 was applied;
- IR_{Food} = ingestion rate of fish by humans (g/day);
- RAF_{Oral} = relative absorption factor from the gastrointestinal tract for a contaminant; and
- *BC* = background concentration (µg/g); the naturally occurring background concentration in environmental media or tissue.

Table A1. Tissue screening value calculation for CECs

CEC	Receptor Population	Receptor Life Stage	Ingestion Rate (g/day)	Reference Dose (TDI) (µg/kg bw/day)	Standard Body Weight (kg)	Relative Absorption Factor (%)	Allocation Factor (unitless)	Screening Value (µg/g, wet weight)
Organotins	Subsistence Fishers	Toddler	94	0.25	16.5	100%	0.2	0.0088
PFOA				0.02				0.0007
PFOS				0.02				0.0021
HBB				2				0.07

APPENDIX B: PRELIMINARY DATA ON STORMWATER-RELATED CONTAMINANTS IN STREAMS FLOWING INTO BURRARD INLET

The following preliminary data were collected and compiled (Brown, Reger *et al.* 2022, unpublished data) as part of a larger project to understand contaminants such as 6PPD quinone in road runoff. Samples were collected from more than 35 creeks in Metro Vancouver, Squamish and Vancouver Island before, during and after[6] rain events.

Data for five sites in Burrard Inlet are presented below. This data presentation is incomplete; it is presented solely for the purposes of demonstrating that tire-derived contaminants have been detected in streams entering Burrard Inlet. Brown et al. intend to publish a more detailed presentation of the results.

Tsleil-Waututh Nation staff were involved in site selection and data collection for McCartney Creek. Sample sizes are provided in Table B1, sampling locations are provided in Figure B1, and averages for the sampling periods in fall 2021, summer 2022 and fall 2022 are presented in Figures B2, B3 and B4, separated into samples processed with and without ascorbic acid (which had been added in an unsuccessful attempt to preserve the samples). The site labelled MosquitoC in Figure B1 is the Highway 1 run-off site presented in Figure B4. It has since been determined that the storm drain at that site is more likely connecting to MacKay Creek, and not Mosquito Creek. The results from that site demonstrate very elevated concentrations, so are separated into Figure B4 so as not to skew the creek sample results in Figures B2 and B3.

The data demonstrate that 6PPD quinone is entering Burrard Inlet tributaries via road runoff.

Table B1. Sample sizes for 6PPD quinone data collection per site and per collection phase (samples were collected before, during and after a rain event). Analytical labs were AXYS, University of Washington (UW) and Institute of Ocean Sciences (IOS).

		Maplewood	McCartney	Mosquito A	Mosquito B	Mosquito C (Hwy 1)
With Ascorbic Acid	**Before**	2	2	2	2	NA
	During	5	5	5	5	1
	After	2	2	2	2	NA
No Ascorbic Acid	**Before**	4	3	2	2	NA
	During	8	4	5	6	3
	After	5	3	3	3	NA

[6] The “after” duration varied from about a half day to a full day of no rain.

Figure B1. Sample locations for 6PPD quinone data collection

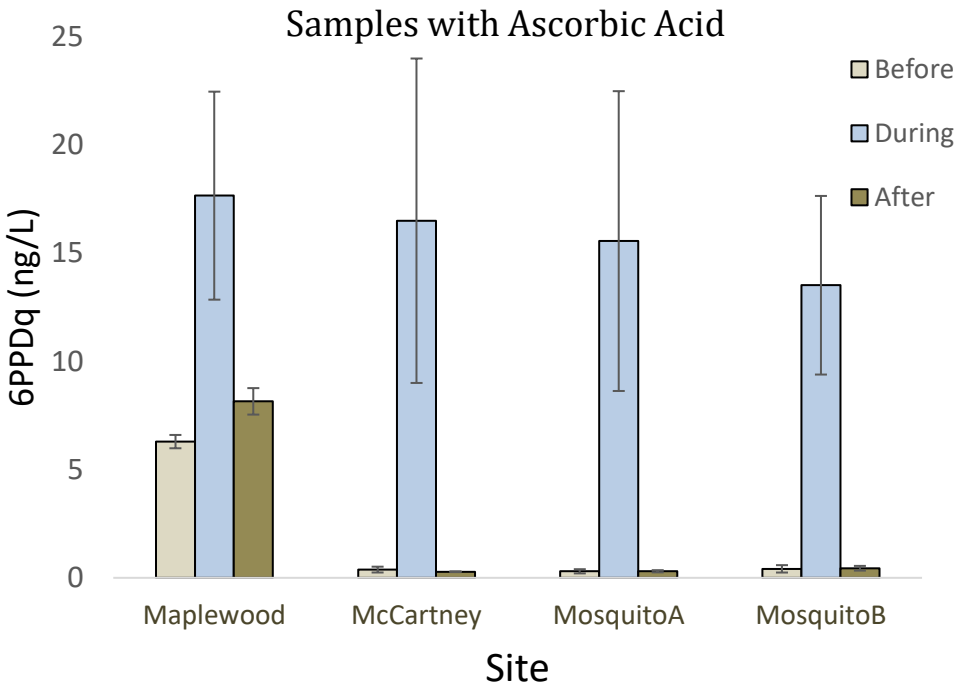

Figure B2. Average 6PPD quinone measured in road runoff before, during and after rain events in fall 2021 and summer 2022, at Maplewood, McCartney Creek and two locations at Mosquito Creek. Samples processed with ascorbic acid. See Table B1 for sample sizes.

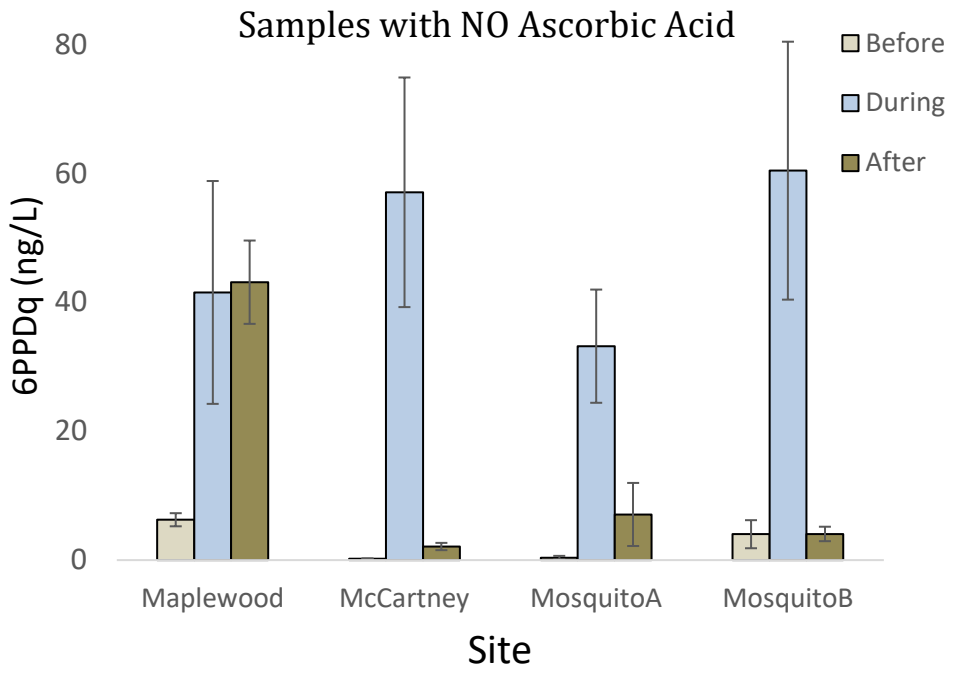

Figure B3. Average 6PPD quinone measured in road runoff before, during and after rain events in fall 2021 and summer 2022, at Maplewood, McCartney Creek and two locations at Mosquito Creek. Samples processed without ascorbic acid. See Table B1 for sample sizes.

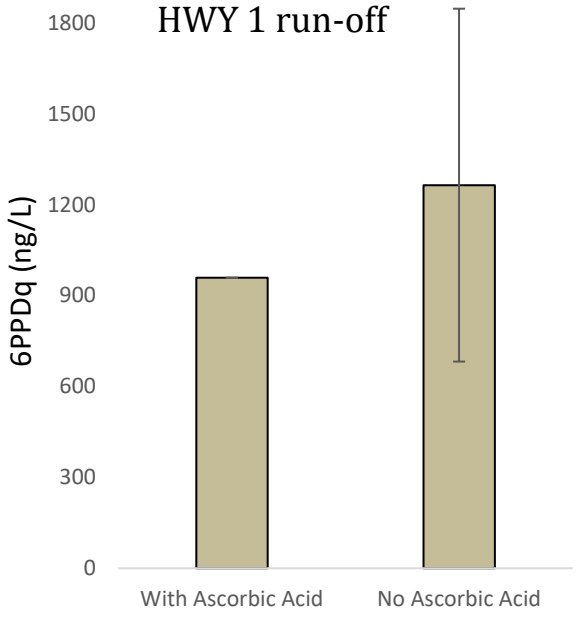

Figure B4. Average 6PPD quinone measured in Highway 1 runoff (labelled MosquitoC in Figure B1) during rain events in summer and fall 2022. See Table B1 for sample sizes.