Water Quality Assessment and Proposed Objectives for Burrard Inlet: Polychlorinated Biphenyls (PCBs), Dioxins (PCDDs) and Furans (PCDFs) 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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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 polychlorinated biphenyls (PCBs), polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) in Burrard Inlet. The proposed objectives were developed using up-to-date research on relevant values and potential effects, sources and factors influencing PCB, PCDD and PCDF levels, benchmark screening, and historic and recent monitoring data for Burrard Inlet.

PCBs, PCDDs and PCDFs are three structurally related classes of persistent organic pollutants (POPs) that are almost exclusively derived from anthropogenic activities (ASTDR 2022). These contaminants bioaccumulate in the marine food web and are toxic to wildlife and humans (Kelly et al. 2007a, Letcher et al 2010, WHO 2002). PCBs have been identified as the contaminant of greatest concern for Southern Resident Killer Whales (Ross et. al 2000, ECCC 2021). PCDDs and PCDFs are endocrine disruptors, and the most toxic congener, 2,3,7,8-tetrachlorodibenzo-para-dioxin (2,3,7,8-TCDD), is a carcinogen (IARC 2018). Some PCBs exhibit a common mechanism of action as 2,3,7,8-TCDD and are referred to as "dioxin-like" (WHO 2002).

PCBs were introduced in the 1930s and were used in industrial and electrical equipment and other materials until their toxicity and persistence in the environment was discovered and their import and use was banned in Canada in 1977 (Environment Canada 1997). PCDDs and PCDFs are the by-products of industrial activities and synthetic processes involving the use or incineration of chlorinated substances (Environment Canada 1990). These substances are considered toxic under the *Canadian Environmental Protection Act* and are targeted for virtual elimination from the environment under the federal *Toxic Substances Management Policy* (Government of Canada 1995).

Although they have been banned for decades, these pollutants continue to persist in the environment. The levels of PCBs, PCDDs, and PCDFs in coastal waters are often related to historical point sources (Johannessen et al. 2008). Current sources include leaching from improper waste disposal and aging equipment, as well as long-range atmospheric transport (Garret and Ross 2010). Combined sewer overflows and urban runoff have also been found to transport PCBs, PCDDs, and PCDFs into aquatic environments (Garret 2010). Due to their hydrophobicity, they tend to settle into sediments and can accumulate in fish and mammals. Humans tend to be exposed through their diet, with a disproportional risk to Indigenous peoples, relative to other populations, due to subsistence fishing (UNEP 2019).

Water quality objectives were set for three PCBs in 1990; no objectives were set in 1990 for PCDDs or PCDFs.

PCB, PCDD and PCDF levels have exceeded sediment and tissue benchmarks throughout Burrard Inlet. The benchmarks used for the data analysis are from BC ENV approved and working water quality guidelines, some of which have been updated as a result of recommendations from the Southern Resident Killer Whale Contaminants Technical Working Group. Screening values protective of human consumption of seafood at rates relevant to coastal Indigenous populations were calculated for PCBs, PCDDs and PCDFs in tissue; these values are conservative enough to also be protective of apex predators such as Southern Resident Killer Whales.

The proposed water quality objectives for PCB, PCDD and PCDF are as follows:

Proposed Water Quality Objectives for PCBs, PCDDs and PCDFs

Sub-basin	Outer Harbour	False Creek	Inner Harbour	Central Harbour	Port Moody Arm	Indian Arm
Water						
Total PCBs in Water (interim screening benchmark)	*6.2 x 10^{-4} ng/L single sample maximum[1]*					
2,3,7,8-TCDD[7] in Marine Water (interim screening benchmark)	*5.1 x 10^{-6} ng/L single sample maximum[2]*					
Sediment						
Total PCBs in Sediment	3.7 x 10^{-6} µg/g dry weight single-sample maximum[3,4]					
Total PCDDs and PCDFs in Sediment	8.5 x 10^{-7} µg/g dry weight single sample maximum[3,4,5]					
Tissue[5]						
Total PCBs in Tissue	2.9 x 10^{-4} µg/g wet weight single-sample maximum[3,6]					
Total PCDDs, PCDFs, and Dioxin-like (planar) PCBs in Tissue	8.0 x 10^{-8} µg/g wet weight single-sample maximum[6,8]					
All						
All PCBs, PCDDs and PCDFs in all media	Decreasing concentrations					

[1]Calculated using method from F. Gobas (*pers. comm.* 2021); not currently proposed as an objective, pending improvements in laboratory detection limits.
[2]Adopted from US EPA (2002); not currently proposed as an objective, pending improvements in laboratory detection limits.
[3]From BC ENV 2020, adopted from Alava et al. 2012.
[4]Based on at least 1 composite sample consisting of at least 3 replicates.
[5]From BC ENV 2020, adopted from CCME 2001a. Expressed on a toxic equivalency basis using toxic equivalent factors for fish (CCME 2001a).
[6] Applies to all tissue types. Based on at least 1 composite sample consisting of at least 5 fish or 25 bivalves. See Rao et al. (in prep) for additional details.
[7]2,3,7,8-tetrachlorodibenzo-*p*-dioxin, the most toxic of the PCDDs and PCDFs.
[8]PCDDs, PCDFs, and dioxin-like PCBs are assessed by converting their concentrations to units of 2,3,7,8-TCDD toxic equivalence (TEQ) using toxic equivalency factors (TEFs) applied to human health. These TEFs are published in Van den Berg et al. 2006 and included in Health Canada 2021. The sum of the TEQs is then compared to the objective. TEFs are provided in Table 3.

Interim objectives for PCBs, PCDDs, and PCDFs in water are proposed until more information becomes available. Objectives for water will be useful for characterizing discharges and contaminated sites. Sediment and tissue objectives are proposed for the protection of aquatic life, including apex predators. Proposed tissue objectives are also protective of Indigenous consumers of seafood.
Improved monitoring of PCBs, PCDDs, and PCDFs is needed to understand the location of hotspots, the presence of different congeners, and to understand their levels in Burrard Inlet overall, including the water column.

International and federal measures have been successful in reducing PCB, PCDD and PCDF use and levels; however, their persistence in higher trophic level wildlife and humans indicates that more work needs to be done to remove them from the environment. This further work includes clean-up of contaminated sites, and increased measures to completely phase out and ensure proper disposal of PCBs, PCDDs, and PCDFs.

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ACRONYMS

AF	Allocation factor
AhR	Aryl hydrocarbon receptor
BC	British Columbia OR Background concentration
BW	Body weight
CCME	Canadian Council of Ministers of the Environment
CSO	Combined sewer overflow
ENV	Ministry of Environment and Climate Change Strategy
IR	Ingestion rate
ISQG	Interim Sediment Quality Guideline
K_{ow}	Octanol water partition coefficient
OC	Organic carbon
PCBs	Polychlorinated biphenyls
PCDDs	Polychlorinated dibenzo-p-dioxins
PCDFs	Polychlorinated dibenzofurans
POPs	Persistent organic pollutants
PEL	Probable effects level
RAF	Relative absorption factor
SV	Screening value
SRKW	Southern Resident Killer Whale
TDI	Tolerable daily intake
TEF	Toxic Equivalency Factor
TEL	Threshold effects level
TEQ	Toxic Equivalence
TCDD	Tetrachlorodibenzo-*p*-dioxin
TRV	Toxicological reference value
TWN	Tsleil-Waututh Nation
US	United States
US EPA	United States Environmental Protection Agency
WSQG	Working Sediment Quality Guideline
WQG	Water Quality Guideline
WQO	Water Quality Objective

1. INTRODUCTION

Tsleil-Waututh Nation (TWN) has identified persistent organic pollutants (POPs) such as polychlorinated biphenyls (PCBs) as contaminants of concern in their Indigenous-led, science based Burrard Inlet Action Plan (TWN 2014). A sediment characterization study conducted in 2017 identified PCBs, polychlorinated dibenzo-p-dioxins (PCDDs, also called dioxins), and polychlorinated dibenzofurans (PCDFs, also called furans) among the most prevalent contaminant classes (and POPs) detected in coastal BC (Morales-Caselles et al. 2017).

This chapter proposes water quality objectives for PCBs, PCDDs and PCDFs in Burrard Inlet. These contaminants are highly toxic, bioaccumulative and persistent in the marine environment (Jones and Voogt 1999). Due to these characteristics, PCBs, PCDDs, and PCDFs were among the first twelve persistent organic pollutants (POPs) targeted by the United Nations (UN) for global reduction and elimination under the Stockholm Convention signed in 2001 (UNEP 2019). In Canada, these substances are included in the List of Toxic Substances in Schedule 1 of the *Canadian Environmental Protection Act* (1999). Although their release to the environment has been banned for decades, they are still present in the marine environment as complex mixtures (Safe 1994, Johannessen et al. 2008a, Morales-Caselles et al. 2017). The toxicity pathway of some PCBs is similar to that of PCDDs and PCDFs and can express interactions among themselves and other pollutants (Giesy and Kannan 1998). As such, PCBs, PCDDs and PCDFs have been combined into a single chapter for data analysis and the setting of objectives.

This chapter includes relevant background information, an overview assessment of current status and trends in PCB, PCDD and PCDF levels in sediment and biota in Burrard Inlet, comparison to 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

2.1 Values and Potential Effects

PCBs, PCDDs and PCDFs are three classes of legacy POPs that are structurally related and that have been extensively studied due to their toxicity to aquatic life and humans, their persistence in the environment, and their potential for biomagnification in food chains (Environment Canada 1997, ASTDR 2022, Jones et al. 1999). PCBs, PCDDs, and PCDFs are toxic to wildlife and humans and have been linked to endocrine disruption, immunosuppression, and reproductive impairment (Letcher et al. 2010, Buckman et al. 2011, Vos et al. 2000). More literature on toxic effects in marine environments is available on PCBs than is available for PCDDs and PCDFs. Wildlife that consume fish, including marine mammals, are particularly vulnerable to PCBs, PCDDs, and PCDFs due to biomagnification in the food web. Humans can be exposed to these contaminants through their food, particularly fish, meat, and milk, or from the environment (Van den Berg et al. 2006).

Southern resident and transient killer whales of British Columbia can be considered among the most contaminated cetaceans in the world (Ross et al. 2000). Research has suggested that exposure to PCBs is the biggest threat to killer whales (Desforges et al. 2018, Hall et al. 2018, ECCC 2021). PCB levels found in these whales have exceeded levels known to be immunotoxic and endocrine disrupting in other marine mammals, particularly harbour seals (Ross et al. 2013). The elevated PCB concentrations observed in southern resident killer whales relative to northern resident killer whales could be partially attributed to consumption of highly contaminated prey near industrialized areas of southwestern British Columbia and northwestern Washington State (Ross et al. 2000).

PCBs, PCDDs, and PCDFs have different physicochemical characteristics that affect their toxicity (Giesy and Kannan 1998). The most toxic PCDD congener, 2,3,7,8-tetrachlorodibenzo-*p*-dioxin, exhibits toxic effects when it binds and activates the aryl hydrocarbon receptor (AhR) present in most vertebrates, causing a multitude of biological and toxic responses (Safe et al. 1985, Giesy and Kannan 1998). Some PCBs, PCDFs, and other PCDD congeners share this common mechanism of action and are therefore referred to as "dioxin-like" substances (Giesy and Kannan 1998). PCBs, PCDDs, and PCDFs have also been shown to elicit toxic responses through non-dioxin like pathways (Giesy and Kannan 1998). Non-dioxin-like effects may involve multiple unrelated mechanisms of action, such as vitamin A and thyroid hormone disruption, and have been linked to developmental neurotoxicity in humans and animals (Brouwer et al. 1989, Ross and Birnbaum 2003, Giesy and Kannan 1998, Klocke and Lein 2020).

2.1.1 PCBs

PCBs are synthetic organic chemicals that are resistant to metabolism and environmental degradation (ASTDR 2002). PCBs consist of 209 individual chemicals, called congeners, with different degrees of chlorination and different molecular structures. The degree of chlorination increases with congener number; for example, in PCB 209, all the hydrogen atoms have been substituted by chlorine atoms. PCBs with the same number of chlorines are called homologs and homologs with different substitution patterns are referred to as isomers.

Individual PCB congeners have different physicochemical characteristics that affect their environmental distribution and toxicity (Giesy and Kannan 1998). For example, the vapour pressure of PCBs decreases with increasing chlorination, therefore lower chlorinated congeners are more volatile and can undergo long range atmospheric transport (Wania and Mackay 1996, Mackay et al. 2006). The hydrophobicity of PCBs, measured by the log octanol-water partition coefficient (K_{ow}), increases with the degree of chlorination, and affects the potential of accumulation in sediments or living organisms (Mackay et al. 2006, ASTDR 2000). Because of their low water solubility, sublethal and chronic toxicity is more likely than acute and lethal effects (Nagpal 1992).

Due to their hydrophobicity, PCBs tend to accumulate in fats, with accumulation noted in livers (Nagpal 1992) because of the relatively high fat content of livers compared to other tissues. They can cause effects on bird embryos, including toxicity and malformations (Vos et al. 2000). Their toxicological effects in mammals include reproductive impairment, immunotoxicity, skeletal abnormalities, endocrine disruption, and negative effects on the population growth rate (Lind et al. 2003, Mos et al. 2006, Tabuchi et al. 2006, Letcher et al. 2010, Jepson et al. 2016, Hall et al. 2018). PCBs have also been associated with neurodevelopmental toxicity in human infants (Winneke et al. 2002). Some PCBs are also carcinogenic to humans (IARC 2018). Major dietary sources of PCBs in humans are fish, meat and dairy products (WHO 2002).

The affinity of PCBs to fats leads to their bioaccumulation in living organisms and their biomagnification in food webs, with the highest levels observed in predatory marine mammals (Alava et al. 2012). Through biomagnification, PCB concentrations in top predators can be a factor of 1000 to 10,000 above their concentrations in benthic invertebrates (Kelly et al. 2007a). PCBs have been identified as a primary contamination threat to high trophic species in the northern hemisphere, and the contaminant of greatest concern to Southern Resident Killer Whales (SRKW; Alava et al. 2012). PCBs have been linked to cancer in California sea lions (Ylitalo et al. 2005). Chronic exposure of marine mammals to PCBs is more problematic than acute exposure, primarily because wildlife are more commonly exposed to low-levels of these contaminants for extended periods of time in their natural settings (Ross and Birnbaum 2003).

Up to seventeen of the 209 PCB congeners are of particular concern because the spatial arrangement of their chlorine atoms is similar to that of 2,3,7,8-TCDD (see section 2.1.2). This leads to dioxin-like toxic

effects of those PCBs (Giesy and Kannan 1998) Dioxin-like PCBs are also referred to as "planar"[1] PCBs, referring to the configuration of chlorine atoms within the molecule. These PCBs behave in the same way as planar PCDDs and PCDFs, in that toxic and biological effects occur through a receptor protein present in most vertebrate tissues (van den Berg et al. 2006). Dioxin-like PCBs, PCDDs and PCDFs express additive interactions among themselves and other pollutants, commonly expressed as the Toxic Equivalence Factor where individual congener toxicity is quantified relative to the most toxic PCDD, 2,3,7,8-TCDD (Giesy and Kannan 1998, Van den Berg et al. 2006). The toxic equivalency patterns in harbour seals from British Columbia and Washington suggested that concentrations of dioxin-like PCBs were greater than that of dioxin-like PCDDs and PCDFs combined (Ross et al. 2004). The remaining PCB congeners are referred to as "non-planar" or "non-dioxin-like" congeners, and have other pathways of toxic effects (Brouwer et al. 1989, Ross and Birnbaum 2003).

2.1.2 PCDDs and PCDFs

There are a total of 75 PCDD congeners and 135 PCDF congeners. Among the most toxic congeners are those with chlorine atoms in the lateral 2, 3, 7, and 8 positions (7 PCDDs and 10 PCDFs) due to their structural resemblance to 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD) (ASTDR 2022). 2,3,7,8-TCDD is the most toxic congener and has been catalogued by the International Agency for Research on Cancer as carcinogenic to humans (IARC 2018).

Like PCBs, PCDDs and PCDFs are highly hydrophobic and tend to bind to sediment and bioaccumulate in the fatty tissues of living organisms, especially congeners that are 2,3,7,8-substituted (Environment Canada 1990). PCDDs and PCDFs are known to cause endocrine disruption in mammals, including marine mammals, meaning that they disrupt hormone function (ASTDR 2022). This leads to effects on growth, development, reproduction and immune function (Srogi 2008). Early stage mortality and associated lesions have been observed in fish (Grimwood and Dobbs 1995).

Although PCDDs and PCDFs have similar hydrophobicities to PCBs, they do not seem to biomagnify in food webs to the same extent as PCBs (CCME 2001a). PCDDs and PCDFs are metabolized to varying degrees in the food chain; for example estimates of the half-life of 2,3,7,8-TCDD are around 58 days in trout , 7 days in harbour seals, and up to 7 years in humans (Niimi 1986, De Swart et al. 1995, Milbrath et al. 2009). Low total PCDD and PCDF concentrations in killer whales suggest a relatively rapid metabolism of dioxin-like contaminants by killer whales and their prey (Ross et al. 2000). Nevertheless, PCDDs and PCDFs are highly persistent in the marine environment where half-lives in sediment may range from several years to more than 100 years (Sinkkonen and Paasivirta 2000), and present a significant risk to lower trophic levels and humans (Lachmuth et al. 2010).

2.2 Potential Sources of PCB, PCDD and PCDF Pollution

2.2.1 Sources of PCBs

PCBs are only generated from anthropogenic activity. They have been used widely as coolants, insulators and lubricants in transformers, capacitors and other electrical equipment (ATSDR 2000), as well as in waxes, adhesives, paints, heat exchange fluids, de-dusting agents, vacuum pump oils, caulking compounds, hydraulic fluids, printing inks, cutting oils, sealants, plasticizers, carbonless copying paper, bridge bearing, fire retardants, cable insulating paper, and flame-proofing (Environment Canada 1997). PCBs were introduced in the 1930s and were mass produced until the 1970s before being banned because they are toxic, persistent and bioaccumulative in the environment (Environment Canada 1997).

[1] Referred to in some sources as "planar".

PCBs were never manufactured in Canada, but were imported from the United States (Environment Canada 2017). The manufacture, import and sale of PCBs was made illegal in 1977, and release to the environment was banned in 1985 (Environment Canada 2017). Regulations under the *Canadian Environmental Protection Act* (1999) prohibit the use of PCBs, their import into Canada and also control the storage and destruction of PCBs in Canada (CEPA 1999). PCBs may still be present in old fluorescent light fixtures, electrical devices and appliances containing PCB capacitors that were made before PCB use stopped. As a signatory of the Stockholm Convention, Canada is required to cease use of PCB containing equipment by 2025 (UNEP 2019).

Before their hazardous nature became known, PCBs were disposed of through burning, dumping and municipal disposal, without any precautions. They were introduced into the environment through vaporisation from plastics, coatings and paints (Nagpal 1992). PCBs continue to be released via leaching from formerly contaminated sites and poorly maintained hazardous waste sites; leaks and spills, for example from the breakdown of aging industrial machinery and household products; and illegal or improper disposal of PCB wastes and PCB-containing consumer products, for example disposal into landfills, incineration and dumping. Once released, PCBs can volatilize and undergo long-range atmospheric transport (Noël et al. 2009).

PCBs were identified as a parameter of concern by BC ENV in 1990 in False Creek, Outer Harbour, Central Harbour and Port Moody Arm (Nijman and Swain 1990), and more recently have been detected at levels of concern in all sub-basins of Burrard Inlet (Rao et al. 2019). Combined sewer overflows (CSOs) and stormwater discharges were identified as pathways for PCBs to enter the Fraser River as early as 1985. PCBs were also found in the leachate from the Lynn Creek landfill (Nijman and Swain 1990).

PCB accumulation had been observed in crabs and mussels in Vancouver Harbour in 1984, especially near ship repair facilities. Sediments in False Creek were also found to be contaminated with PCBs (Nijman and Swain 1990). Many additional studies have found PCB hotspots in urban harbours such as Vancouver Harbour and Victoria Harbour (Morales-Caselles et al. 2017). Studies in harbor seals in BC and Washington State demonstrated that PCBs were highest in Puget Sound and decreased with increasing distance from urban and industrial areas (Ross et al. 2004).

Sun (2020) identified industrialized areas around Burrard Inlet's coastline as major sources of dissolved and particulate PCBs into the Inlet, and subsequently into the Strait of Georgia. Sediment analysis by Grant et al. (2011) had also identified Burrard Inlet as a source of PCBs, with point sources including previously contaminated sites, waste discharges and industrial areas. Identified non-point sources included urban and industrial runoff. Port Moody Arm in particular was identified as containing high levels of PCB contamination. The uniform distribution of PCBs throughout the Strait of Georgia suggests that PCBs have become widely dispersed even after primary source control (Grant et al. 2011).

Sun (2020) suggested that effluents from wastewater treatment plants are not currently expected to be major point sources of PCBs into ambient waters; sources are more likely to be long range transport and atmospheric deposition or soil leaching and spills from industrial zones. Urban runoff is also a leading pathway for PCB pollution; PCBs in old construction materials, paints, transformers and fluorescent light fixtures can volatilize and become airborne, then attach to surfaces, or onto particles on or near the ground and be washed away by rain (Ross et al. 2013). PCBs can be removed from the water column when volatized off of surface water, adsorbed to suspended solids and sediments, or concentrated into biota (ATDSR 2000).

2.2.2 Sources of PCDDs and PCDFs

PCDDs and PCDFs are produced primarily as unintentional by-products of industrial combustion activities and synthetic processes. Trace amounts can also be produced from natural sources such as volcanoes and forest fires (Baars et al. 2004). Historical sources include the chlorinated bleaching process at pulp and paper mills and the manufacture of chlorophenol-based chemicals for wood treatment. Current sources may include the burning of salt-laden wood, the manufacture of certain chlorinated substances including pesticides; incomplete combustion; the incineration of municipal solid and medical waste; the use of pentachlorophenol as a wood preservative; and diesel combustion (ASTDR 2022, Garrett and Ross 2010, Srogi 2008, Ross et al. 2004). Some of these activities release hundreds of PCB, PCDD and PCDF congeners (Van den Berg et al. 2006). Once released, PCDDs and PCDFs typically enter the marine environment through runoff or atmospheric transport and settle in bottom sediments (Jean M. Czuczwa and Ronals A. Hites 1986, Atkinson 1991).

Releases of PCDDs and PCDFs appear to have started with the introduction of chlorine bleaching in the early 1960s and increased with time until stringent federal regulations were put in place in the late 1980s and early 1990s (Macdonald et al. 1992, Yunker et al. 2002). The chlorine bleaching process in pulp and paper mills was identified as a key source of PCDDs and PCDFs to the environment in the 1980s (Garrett and Ross 2010). In 1992, 10 pulp and paper mills in BC discharged chlorinated effluent directly into the marine environment (Macdonald et al. 1992). The congener 2,3,7,8-TCDF was found to be more concentrated near its primary sources, which were pulp mills, but was detected more than 30 km from known sources (Macdonald et al. 1992). Contamination by PCDDs and PCDFs resulted in shellfish harvesting restrictions on the BC coast between 1988 and 1995 (Hagen et al. 1997).

Stringent federal regulations have since significantly reduced the concentration of PCDDs and PCDFs in pulp and paper effluents (Garrett and Ross 2010). In 1990, PCDDs and PCDFs were declared toxic under the *Canadian Environmental Protection Act* due to their persistence in the environment and their risk to human and wildlife health (Environment Canada 1990). In 1992, the Pulp and Paper Mill Regulations under *CEPA* encouraged industry to switch to chlorine-free bleaching technologies, resulting in a 99% reduction in releases of PCDDs and PCDFs to marine waters by 1997 (Government of Canada). Pulp and paper mill effluent is no longer considered a major source of these contaminants into the BC environment (Garrett and Ross 2010).

A more recent source of PCDDs and PCDFs is their atmospheric release during the combustion of wood containing chlorine and salt-laden wood, e.g., ocean transported via log booms or driftwood (Garrett and Ross 2010). The successful implementation of the CCME Canada-Wide Standards for Dioxins and Furans (CCME 2001b), which addressed phase-out activities and pollution prevention strategies for the BC coast pulp and paper boilers burning salt-laden wood, substantially decreased PCDD and PCDF releases to the atmosphere by 60% between 2001-2009 (Government of Canada 2006, Environment Canada 2013).

Despite source controls, in a study of POP concentrations in harbour seals, concentrations of PCDDs and PCDFs were lower in seals from Puget Sound than in seals from the Strait of Georgia, pointing to the persistence of pulp and paper industry pollutants in coastal British Columbia waters (Ross et al. 2004).

PCDDs and PCDFs were not considered in the development of the 1990 Water Quality Objectives for Burrard Inlet (Nijman and Swain 1990). More recently, they have been identified at levels of concern in Outer Harbour, Inner Harbour, Central Harbour, Port Moody Arm and Indian Arm (Rao et al. 2019).

2.2.3 Long-range transport

Historical sources of POPs within a particular area contribute to POP contamination in that area; however, long-range atmospheric transport and deposition of POPs used in other parts of the world also play a role (Ross et al. 2004). PCBs are circulated globally, with atmospheric transport being the most important mechanism for global dispersion (Wania and MacKay 1996). Although PCB manufacture has stopped, PCBs continue to be redistributed in the environment in soil, water and air (ATSDR 2000). PCBs can travel long distances and eventually be deposited in remote environments. As a result, they are found all over the world. Studies have demonstrated that the northeast Pacific Ocean is likely a sink for POPs, following atmospheric transport and deposition, as well as transport via migratory biota (Wania and Mackay 1996, Wilkening et al. 2000).

Atmospheric transport represents the major mechanism for the delivery of POPs into remote environments, with the more volatile congeners traveling greater distances (Wania and Mackay 1996). More highly chlorinated compounds are more likely adsorbed to particles because of their higher hydrophobicity, and thus more rapidly deposited and retained by sediments (Hawker and Connell 1988). The lighter the congener, the further it can be transported from its source: congeners with 0 to 1 chlorine atoms will remain in the atmosphere; those with 1 to 4 chlorine atoms gradually migrate toward the polar latitudes, and those with 4 to 8 chlorine atoms remain close to their source (Wania and Mackay 1996, Beyer et al. 2000).

2.3 Factors Influencing Levels of PCBs, PCDDs and PCDFs in Burrard Inlet

2.3.1 PCBs

PCBs enter the environment as mixtures of various congeners (ATSDR 2000). PCB concentrations in the marine environment are affected by historical and current discharges and inputs, presence of organic matter, atmospheric deposition, biotransformation rates, volatilization and remobilization. The distribution of PCBs in coastal waters has been found to be related to historical point sources in urban areas, and inshore marine areas of industrial zones (Grant et al. 2011). Overall, PCB concentrations in coastal environments have been found to be in decline due to source control under federal and international regulations; however, due to their persistence, PCBs continue to be cycled in the marine environment (Grant et al. 2011, Johannessen et al. 2008b).

PCBs are hydrophobic and preferentially bind to organic particles in the water column, settle into sediments, and may accumulate in fish and marine mammals (ASTDR 2000, Grant et al. 2011). Highly chlorinated PCBs, which have lower water solubility and higher octanol-water partition coefficients ($K_{ow)}$, preferentially bind to particulate organic matter and sediments (ATSDR 2000). Marine sediments are a sink for PCBs. PCB concentrations of marine sediments in the Strait of Georgia are uniformly distributed throughout the basin with older sediments containing higher concentrations of total PCBs (Grant et al. 2011). Total PCBs in sediment also have been found to correlate positively with percent organic carbon (Grant et al. 2011). PCB concentrations in Strait of Georgia sediments were found to be higher in areas where sedimentation rates were low, organic carbon was high, and where the depth of benthic mixing is high, bringing older highly contaminated sediments to the surface (Johannessen et al. 2008b).

Natural events or human activities such as dredging and disposal operations can remobilize pollutants such as PCBs that are bound to contaminated sediment. This process can make them available to biota (Morales-Caselles et al. 2017). The uptake of PCBs by benthic invertebrates depends on the chemical properties of the specific PCB congeners (Dinn et al. 2012) .

In Burrard Inlet, the Inner Harbour (especially Coal Harbour) and Port Moody Arm have been found to contain high concentrations of legacy contaminants such as PCBs, likely due to historical industrial

activities and low sedimentation rates (Johannessen et al. 2015). PCBs have also been detected in Indian Arm (Morales-Caselles et al. 2017). Studies conducted prior to the establishment of the 1990 Water Quality Objectives for Burrard Inlet found PCBs in landfill leachate entering Lynn Creek (Nijman 1993).

2.3.2 PCDDs and PCDFs

PCDDs and PCDFs also bind to particulate matter and settle into sediments, making them available to bottom dwelling organisms (Hagen et al. 1997). The bioavailability of PCDDs and PCDFs depends on the physicochemical properties of the congeners, geochemical factors (e.g. organic content, particle size), and biological factors (CCME 2001a). Release to the atmosphere from the combustion of residential and municipal waste or burning of salt-laden wood have been identified as current sources (Government of Canada 2006), so direct input as well as atmospheric transport and deposition may both be pathways for entry into Burrard Inlet waters (Jean M. Czuczwa and Ronals A. Hites 1986, Atkinson 1991).

The highest concentrations of PCDDs and PCDFs in Burrard Inlet recorded by Morales-Caselles et al. (2017) were in Port Moody Arm and the Inner Harbour.

Although strictly regulated for decades, the concentration of PCBs, PCDDs, and PCDFs in marine sediments is primarily controlled by the revolatilization of contaminated sediments, and sedimentation and mixing rates. When degradation of contaminants is slow or absent (as in the case of PCBs, PCDDs and PCDFs), burial becomes the only process of removal from the sediments (Johannessen et al. 2015). If the sedimentation rates are low, however, and bio-mixing is high, older contaminated sediments are brought to the surface (Johannessen et al. 2008a).

2.4 1990 Provisional Water Quality Objectives for PCBs

Objectives were set in 1990 for PCBs in sediments and fish tissue in all sub-basins except Indian Arm (see Table 1) because of the occurrence of PCBs in combined sewer overflows and stormwater discharges. The objectives proposed for the marine waters of Burrard Inlet were those which had been proposed for the Fraser River at the time (Nijman and Swain 1990). Objectives for PCBs were only set for the water column for Lynn Creek. The term 'PCBs' was defined as the sum of Aroclor 1242, 1254 and 1260 (Nijman 1993). Aroclor is the trade name for PCB mixtures used between the 1930s and 1970s. Aroclors 1016, 1242, 1254, and 1260 were the most widely used PCB mixtures.

Water quality objectives were not established in 1990 for PCDDs or PCDFs.

Table 1: 1990 Provisional Water Quality Objectives for total PCBs (sum of Aroclor 1242, 1254 and 1260)

Sub-basin	False Creek	Outer Harbour	Inner Harbour	Central Harbour	Port Moody Arm	Indian Arm
Sediment	0.03 µg/g dry weight maximum[1]					N/A
Fish Tissue	0.5 µg/g wet weight maximum[2]					N/A

[1] The maximum value should not be exceeded in bottom surface sediments taken in any part of the sub-basin except in the initial dilution zone of effluents. The average of at least three replicate sediment samples taken from the same site should be used to check the objective

[2] Applies only to muscle tissue, not the whole fish or organs, of any fish of any species caught in any part of the sub-basin, including the initial dilution zones of effluents.

3. WATER QUALITY ASSESSMENT

3.1 Benchmarks Used in this Assessment

Benchmarks were used to screen available data for potential acute and chronic effects and to inform the derivation of proposed objectives for PCB, PCDD and PCDF levels in Burrard Inlet. Based on the available literature, aquatic life is the value most sensitive to PCB, PCDD and PCDF levels in marine water and sediments, particularly higher trophic level species (Ross et al. 2000, Tabuchi et al. 2006, Desforges et al. 2018, Hall et al. 2018). Finfish and shellfish consumption by humans are the most sensitive values for PCB, PCDD and PCDF levels in tissue.

Canadian guidelines for the protection of these values were used as screening benchmarks where available. The following potential sources of screening benchmarks were considered:

1. BC Approved Water Quality Guidelines (BC ENV 2019);
2. BC Working Sediment Quality Guidelines (BC ENV 2020, for consistency with ECCC 2021 and based on Alava et al. 2012 and CCME 2001a);
3. ECCC Recommended Environmental Quality Guidelines for the Protection of Southern Resident Killer Whales (SRKW) and Their Prey (ECCC 2021, BC working guidelines are generally consistent with these);
4. BC Working Tissue Quality Guidelines (BC ENV 2020, for consistency with ECCC 2021, based on Alava et al. 2012, and CCME 2001a);
5. Calculated tissue screening values protective of human health (Thompson and Stein 2021);
6. Benchmarks from other jurisdictions.

The screening benchmarks used for the data assessment in this report are summarized in Table 2.

The benchmarks listed above are orders of magnitude more conservative than the 1990 provisional water quality objectives for PCBs in Burrard Inlet. Additional benchmarks exist (e.g. ECCC screening criteria for Disposal at Sea); however, only the most protective benchmarks relevant to the identified values to protect in Burrard Inlet (Rao et al. 2019) were used in the data assessment.

Water

BC has approved guidelines for total PCBs (0.1 ng/L), as well as for PCBs 105 (0.09 ng/L), 126 (0.00025 ng/L), 169 (0.06 ng/L) and 77 (0.04 ng/L) in water (BC ENV 2019, from MOELP 1992). The guideline for total PCBs (0.1 ng/L) is consistent with the recommended Environmental Quality Guidelines for the protection of SRKW and their prey, prepared as part of the SRKW Contaminants Technical Working Group convened by Environment and Climate Change Canada (ECCC 2021). These are intended to be protective of freshwater and marine aquatic life, and wildlife consumers of fish and shellfish. Studies at the time were limited to small mammals such as mink, as well as seabirds (Nagpal 1992), so it is unclear whether these levels would be protective of marine mammals. Congeners 105, 126, 169 and 77 are the most common dioxin-like PCBs, so the Province of BC calculated guidelines for those congeners separately. The United States Environmental Protection Agency (US EPA) has a criterion for total PCBs in water of 0.064 ng/L, including the sum of all congeners, isomers, homologs or Aroclor analyses[2] (US EPA

[2] Aroclor is a commercial name for PCBs produced by Monsanto. The Aroclor numbers refer to the percentage of chlorine added; the proportion of individual congeners vary across products. Aroclor-based analyses are an outdated and unreliable technique for estimating total PCBs based on a subset of congeners in a particular mixture. PCBs weather and change structure in the environment, meaning that Aroclor-based estimates can differ significantly from actual PCB concentrations. High resolution analyses of total PCB concentrations are a preferred method (Sather et al. 2001). Detection limits for PCBs in water are low, so Aroclor-based analyses for water would be particularly inadequate; a more robust analytical method would be high

2002). The US EPA also has a criterion for 2,3,7,8-TCDD in marine water of 5×10^{-6} ng/L (US EPA 2002). These criteria are intended to be protective of human consumption of organisms, and based on an incremental lifetime cancer risk of 10^{-6} (US EPA 2002). These benchmarks were the best available at the time of writing.

Sediment

The BC Working Sediment Quality Guideline (WSQG) for total PCBs was updated in 2020 to 0.0037 µg/kg dry weight (BC ENV 2020). This update was to maintain consistency with recommended Environmental Quality Guidelines for the protection of SRKW and their prey, prepared as part of the SRKW Contaminants Technical Working Group (ECCC 2021). Those Environmental Quality Guidelines adopt the recommendation from Alava et al. (2012) for the geometric mean total PCB concentrations in sediments that is estimated to result in total PCB concentrations in Chinook salmon below Canada's Tissue Residue Guideline for Wildlife Consumers of Fish or below toxicity threshold concentrations in 95% of male and female Northern and Southern Resident Killer Whales.

The BC WSQGs for total PCDDs and PCDFs in sediment (BC ENV 2020) adopts the environmental quality guidelines from the Canadian Council for Ministers of the Environment (CCME 2001a). It is intended to be protective of freshwater and marine aquatic life. To develop these guidelines, a safety factor of 10 was applied to the threshold effects level (TEL) and the probable effects level (PEL), which were derived largely from studies on field-collected sediments that measured concentrations of 2,3,7,8-substituted PCDDs and PCDFs and were assessed on a toxic equivalency basis using toxic equivalent factors for fish (CCME 2001a).

Tissue

BC's approved guideline for PCBs in fish tissue is 0.1 µg/g (short term maximum; BC ENV 2019, from MOELP 1992) to protect fish-consuming birds and animals; however, this value is not protective of high trophic level species such as SRKW. In 2020, BC updated their working guideline for total PCBs in wildlife dietary species to 0.00029 µg/g wet weight (BC ENV 2020), adopted from Alava et al. (2012) to be protective of marine mammals, particularly killer whales. The SRKW Contaminants TWG recommended PCB tissue guideline (ECCC 2021) also adopted the guideline calculated by Alava et al. (2012), Alava et al. (2012) calculated that value from the geometric mean total PCB concentration in Chinook salmon required to be below toxicity threshold concentrations in 95% of male and female Northern and Southern Resident Killer Whales[3]. That research did not separate dioxin-like and non-dioxin-like PCBs.

CCME (2001a) published tissue residue guidelines for total PCDDs, PCDFs and dioxin-like (planar) PCBs that are protective of wildlife consumers of aquatic biota (7.1×10^{-7} µg/g and 4.75×10^{-6} µg/g for mammals and birds, respectively).

Because the derivation of the above tissue benchmarks did not involve a risk assessment appropriate for coastal Indigenous consumers, who consume many times more seafood than the average Canadian (e.g. Mos et al. 2004), human-health based screening values for fish and shellfish tissue were derived from Health Canada toxicological reference values (TRVs) and risk assessment methodologies (Health Canada 2010a,b,c, 2012a; Richardson 1997, Richardson and Stantec 2013).

resolution gas chromatography-mass spectrometry methods to measure total PCBs, individual congeners or total planar PCBs and toxic equivalency (Peter Ross, *pers. comm.* January 2022).

[3] Since toxicity threshold effect concentrations of total PCBs in killer whales do not exist, Alava et al. 2012 used the toxicity effect concentrations for harbor seals and bottlenose dolphins. All studies involved free-ranging or captive fed marine mammals, wherein PCBs represented the dominant concern and the contaminants which best correlated with observed effects (Alava et al 2012).

The risk-based approach used to calculate human health-based tissue screening values for fish and shellfish tissue considers: the contaminant *receptors* (subsistence fisher, recreational fisher, the general BC population, pregnant woman, child and toddler), *exposure* to the contaminant (how much fish the receptors consume), and the contaminant *toxicity* (what is known about the contaminant and how it affects different receptors). Receptor characteristics were defined from Richardson and Stantec (2013), exposure was calculated through fish ingestion rates from Richardson (1997) and Health Canada (2010c), and toxicity was defined through TRVs prescribed by Health Canada (2010a).

Screening values are defined as threshold values against which contaminant concentrations in fish 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 2010b). Screening values provide general guidance to environmental managers when site-specific information is not available and represent a suggested safe level of contaminant in fish tissue based on a conservative estimate of a person's fish consumption per day; they do not provide advice regarding consumption limits or constitute a fishing advisory. Exceedances of a screening value may indicate that further investigation to assess human health risk at a particular site is warranted; however, exceeding a screening value does not imply an immediate risk to human health.

Tissue screening values were calculated using equations from Health Canada (2012a). The calculations used for PCBs, PCDDs and PCDFs are summarized in Appendix A; further details about the methodology are described by Thompson and Stein (2021). The screening value used as a benchmark for tissue is the most conservative, as calculated for the most sensitive receptor. An allocation factor of 0.2 was used in the calculation to reflect the percentage of PCBs, PCDDs and PCDFs assumed to come from country foods (in this case, wild seafood). Three screening values were selected to capture a range of potential fishers. The most conservative value is protective of a toddler from a subsistence fisher population while the less conservative values correspond with adult subsistence fishers and adult recreational fishers. Three benchmarks were selected as this would provide more reference points for the data assessment. Although some PCDDs are carcinogenic, the most recent Health Canada guidance on these substances (Health Canada 2021) considers carcinogenic effects but uses non-carcinogenic effects (developmental toxicity) as the most sensitive endpoint.

TRVs were available for non-dioxin-like PCBs, and for 2,3,7,8-TCDD so distinct tissue screening values were derived for a) non-dioxin-like PCBs and b) PCDDs, PCDF and dioxin-like PCBs.

The total dioxin-like toxicity of complex mixtures of PCBs, PCDDs and PCDFs in tissues is calculated by way of a toxic equivalency factor (TEF) approach that quantifies the toxicity of individual congeners relative to the most toxic of the PCBs, PCDDs and PCDFs, namely 2,3,7,8-TCDD (van den Berg et al. 2006), provided in Table 3[4]. In this method, the concentration of each PCDD, PCDF and PCB in a given mixture is multiplied by its respective TEF, and the resulting concentrations are summed to estimate a total 2,3,7,8-TCDD toxic equivalence (TEQ) concentration (Health Canada 2021). That concentration is then measured against the benchmark for PCDDs, PCDFs and dioxin-like PCBs, which was derived using the tolerable daily intake (TDI) for 2,3,7,8-TCDD. See Appendix A for details.

[4] This approach does not apply to sediments (van den Berg et al. 2006).

Table 2: Screening Benchmarks for PCB, PCDD and PCDF in Water, Sediment, and Tissue Used in this Assessment

Sample Type	Screening Benchmark	Value	Reference
Water			
Total PCBs in Marine Water (planar + non-planar)	0.1 ng/L	Freshwater and marine aquatic life, and wildlife consumers of fish and shellfish	BC ENV 2019 (MOELP 1992)
Total PCBs in Marine Water (planar + non-planar)	0.064 ng/L	Human consumption of finfish and shellfish	US EPA 2002
Dioxin-like (planar) PCBs in Water	PCB 105: 0.09 ng/L PCB 126: 0.00025 ng/L PCB 169: 0.06 ng/L PCB 77: 0.04 ng/L	Freshwater and marine aquatic life, and wildlife consumers of fish and shellfish	BC ENV 2019 (MOELP 1992)
2,3,7,8-TCDD in Marine Water	5.1×10^{-6} ng/L	Human consumption of finfish and shellfish	US EPA 2002
Sediment			
Total PCBs in Sediment (planar + non-planar)	3.7×10^{-6} µg/g dry weight	Marine aquatic life (Resident Killer Whales)	BC ENV 2020, from Alava et al. 2012
Total PCDDs and PCDFs in Sediment	Lower ISQG (threshold effects level): 8.5×10^{-7} µg/g dry weight Upper SWQG (probable effects level): 2.15×10^{-5} µg/g dry weight	Marine aquatic life	BC ENV 2020, from CCME 2001a
Tissue			
Total PCBs in Tissue (planar + non-planar)	0.00029 µg/g wet weight	Marine aquatic life (Resident Killer Whales)	BC ENV 2020, from Alava et al. 2012
Total non-dioxin-like (non-planar) PCBs in Tissue	0.0004 µg/g wet weight (toddler subsistence fisher screening value) 0.0007 µg/g wet weight (adult subsistence fisher screening value) 0.0014 µg/g wet weight (adult recreational fisher screening value)	Human consumption of finfish and shellfish	Thompson and Stein 2021 Health Canada 2021
Total PCDDs, PCDFs and dioxin-like (planar) PCBs in Tissue (summed using TEFs as per Table 3)	8.0×10^{-8} µg/g wet weight (toddler subsistence fisher screening value) 1.6×10^{-7} µg/g wet weight (adult subsistence fisher screening value) 3.2×10^{-7} µg/g wet weight (adult recreational fisher screening value)	Human consumption of finfish and shellfish	Thompson and Stein 2021 Health Canada 2021

Table 3: Toxic equivalency factors (TEFs) for PCDDs, PCDFs, and Dioxin-Like PCBs (Health Canada 2021, from van den Berg et al. 2006)

Substance	CAS No.	TEF
Polychlorinated Dibenzo-p-dioxins		
2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD)	1746-01-6	1
1,2,3,7,8-Pentachlorodibenzo-p-dioxin (PeCDD)	40321-76-4	1
1,2,3,4,7,8-Hexachlorodibenzo-p-dioxin (HxCDD)	39227-28-6	0.1
1,2,3,6,7,8-Hexachlorodibenzo-p-dioxin (HxCDD)	57653-85-7	0.1
1,2,3,7,8,9-Hexachlorodibenzo-p-dioxin (HxCDD)	19408-74-3	0.1
1,2,3,4,6,7,8-Heptachlorodibenzo-p-dioxin (HpCDD)	35822-46-9	0.01
Octachlorodibenzo-p-dioxin (OCDD)	3268-87-9	0.0003
Polychlorinated Dibenzofurans		
2,3,7,8-Tetrachlorodibenzofuran (TCDF)	51207-31-9	0.1
1,2,3,7,8-Pentachlorodibenzofuran (PeCDF)	57117-41-6	0.03
2,3,4,7,8-Pentachlorodibenzofuran (PeCDF)	57117-31-4	0.3
1,2,3,4,7,8-Hexachlorodibenzofuran (HxCDF)	70648-26-9	0.1
1,2,3,6,7,8-Hexachlorodibenzofuran (HxCDF)	57117-44-9	0.1
1,2,3,7,8,9-Hexachlorodibenzofuran (HxCDF)	72918-21-9	0.1
2,3,4,6,7,8-Hexachlorodibenzofuran (HxCDF)	60851-34-5	0.1
1,2,3,4,6,7,8-Heptachlorodibenzofuran (HpCDF)	67562-39-4	0.01
1,2,3,4,7,8,9-Heptachlorodibenzofuran (HpCDF)	55673-89-7	0.01
Octachlorodibenzofuran (OCDF)	39001-02-0	0.0003
Non-ortho Substituted PCB Congeners		
PCB 77	32598-13-3	0.0001
PCB 81	70362-50-4	0.0003
PCB 126	57465-28-8	0.1
PCB 169	32774-16-6	0.03
Mono-ortho Substituted PCB Congeners		
PCB 105	32598-14-4	0.00003
PCB 114	74472-37-0	0.00003
PCB 118	31508-00-6	0.00003
PCB 123	65510-44-3	0.00003
PCB 156	38380-08-4	0.00003
PCB 157	69782-90-7	0.00003
PCB 167	52663-72-6	0.00003
PCB 189	39635-31-9	0.00003

3.2 Data Sources

Data for PCB, PCDD, and PCDF levels in Burrard Inlet were gathered from several studies and monitoring programs. A summary of the datasets used for this assessment is presented in Table 4. Although other datasets containing sampling data may exist, the priority datasets were found to be the best available data for assessing the status of PCB, PCDD, and PCDF in Burrard Inlet within the constraints of the project.

Maps showing the distribution of sampling sites for each of the studies or monitoring programs are provided in Figure 1 through Figure 4.

Table 4: Studies and Monitoring Programs Contributing Data Used for the Assessment

Source	Study/Monitoring Program, Years	No. of Unique Observations	No. of Sites	Sampling Frequency	Sample Type
BC ENV	Monitoring Data for Burrard Inlet, 1991-2020	12 PCB in water 39 PCB in sediment 4 PCDD and PCDF in sediment 30 PCB in fish tissue	2 water 15 sediment 12 tissue	Irregular	Total PCBs in sediment and tissue PCDD and PCDF homologs and congeners in tissue
Environment Canada	Canada Disposal at Sea Program, 2009-2017	20 PCB in sediment	8	Irregular	Total PCBs in sediment
Ocean Wise	PollutionTracker, 2015–2018	15 PCB in sediment 14 PCDD and PCDF in sediment 13 PCB in mussel tissue 13 non-dioxin-like PCB in mussel tissue 13 dioxin-like PCB in mussel tissue	15 sediment 8 tissue	3 sediment samples and 1 composite of 50–200 mussels per site on a single day in October 2015, December 2015, April 2016, October 2018, and/or November 2018	PCB, PCDD, and PCDF homologs and congeners in sediment and tissue PCB homologs and congeners in sediment and tissue
Metro Vancouver	Burrard Inlet Ambient Monitoring Program, 2007–2016	105 PCB in sediment 70 PCDD and PCDF in sediment 70 PCB in fish tissue 35 non-dioxin-like PCBs in fish tissue 67 dioxin-like PCB, PCDD, and PCDF in fish tissue	7	5 water samples/year, at both top and bottom of water column, regular Reported as maximum values and mean of 5 samples in 30 days 5 sediment samples/2 years, regular Tissue samples in 2007 and 2012	Total PCBs in sediment and tissue, total PCDD in tissue PCB, PCDD, and PCDF homologs and/or congeners in sediment and tissue

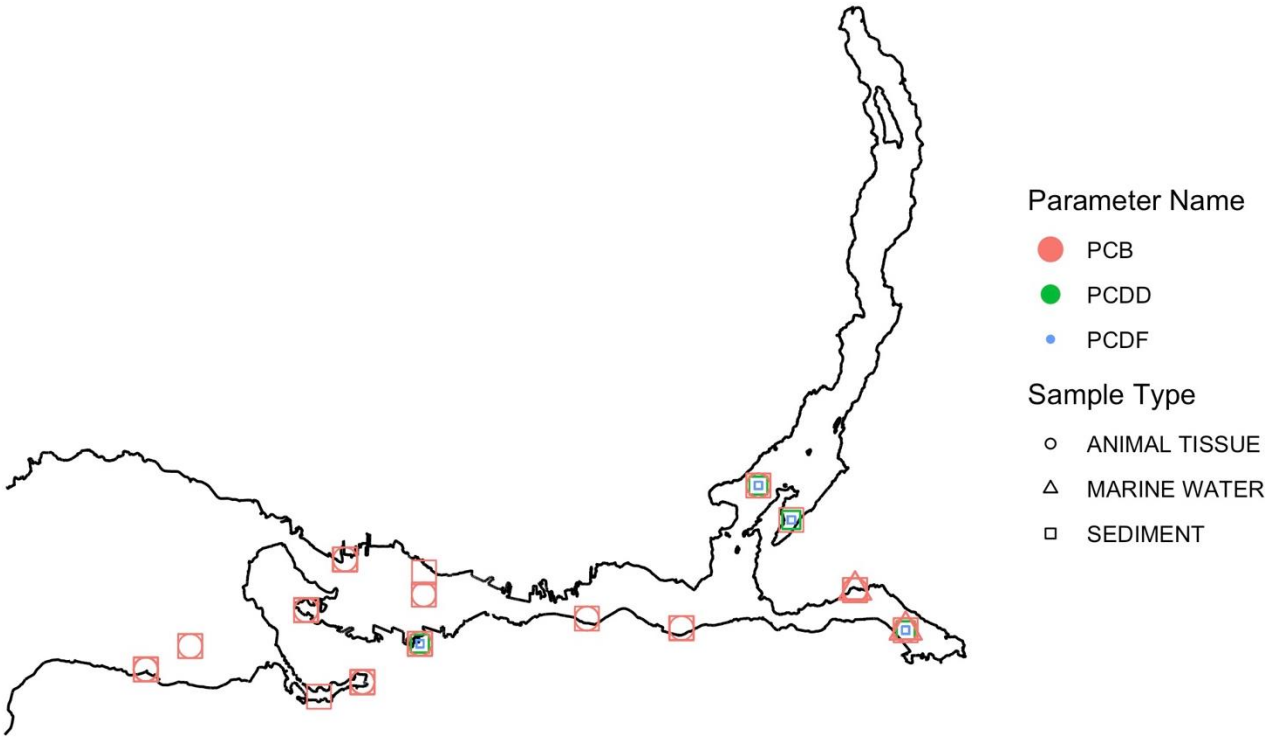

Figure 1: BC ENV sampling stations for PCBs, PCDDs, and PCDFs in Burrard Inlet (1991 to 2020)

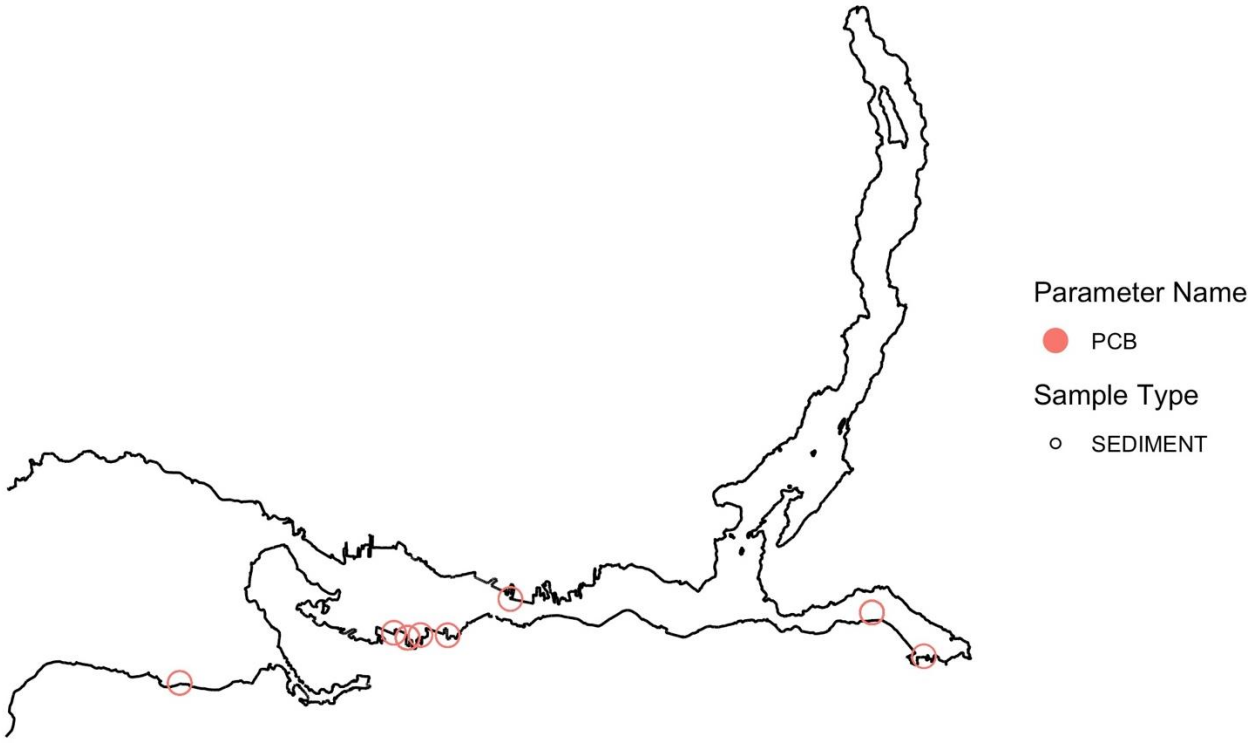

Figure 2: Environment Canada Disposal at Sea sampling stations for PCBs in Burrard Inlet (2009 to 2017)

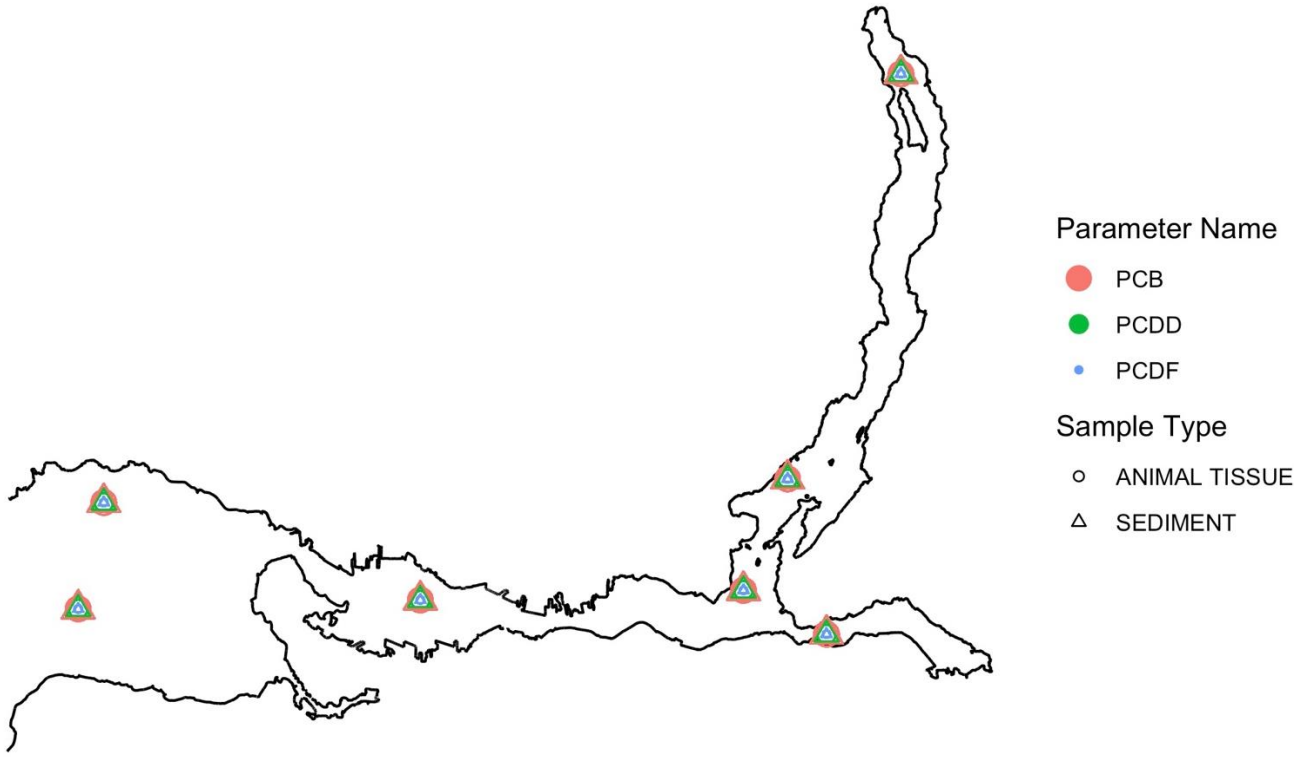

Figure 3: Metro Vancouver sampling stations for PCBs, PCDDs, and PCDFs in Burrard Inlet (2007 to 2016)

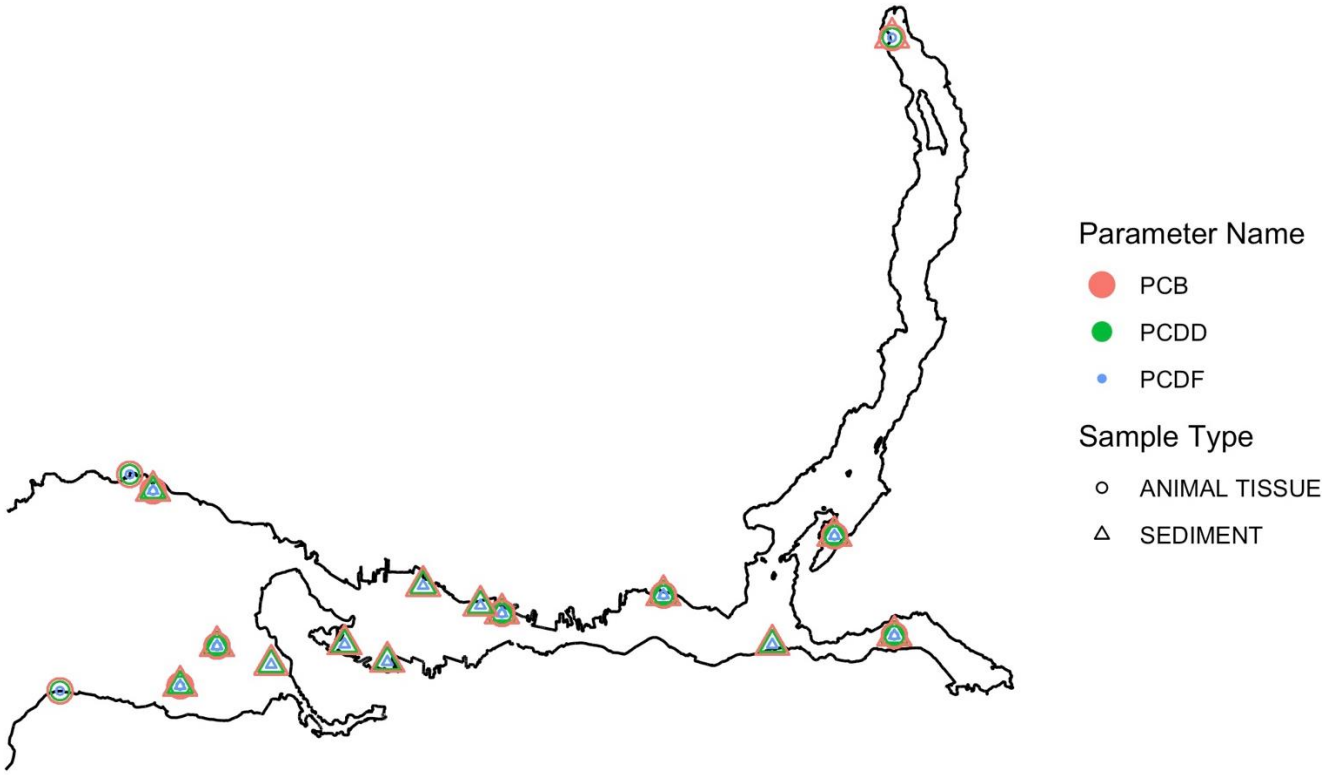

Figure 4: PollutionTracker sampling stations for PCBs, PCDDs, and PCDFs in Burrard Inlet (2015 to 2018)

3.3 Assessment Results

Ambient monitoring data were compared to screening benchmarks, and temporal and spatial observations are presented by sub-basin, where appropriate. Data from monitoring programs for specific sources of PCBs such as outfalls were outside of the scope of this assessment; however, a comparison of the screening benchmarks (section 3.1) and proposed objectives (section 4.1) to the results of these other monitoring programs would be valuable to develop an understanding of contributing sources.

Because of variation in the sampling and analytical methods and distribution of sites, results from each monitoring program are discussed separately. Programs that collect samples at sites close to the shore are expected to produce different results compared to programs that collect samples offshore and at depth for ambient conditions. Therefore, there are limitations on comparing results between the monitoring programs.

The screening benchmarks used in this assessment are for total PCBs and PCDDs/PCDFs. In some monitoring programs, however, specific congeners and/or homologs were also measured. To compare monitoring data to the screening benchmarks, total PCB and/or PCDD/PCDF measurements were prioritized. Where total concentrations were not measured by the laboratory, homolog and/or congener concentrations were summed to enable comparison with the benchmarks. Where both homologs and congeners were measured, summed totals of homologs were prioritized for the comparison over summed totals of congeners because there were far fewer non-detects among the homolog analytes. The summed totals were calculated twice; once including homolog or congener values that were below detections limits at the detection limits, and once excluding values that were below detection limits. The comparison illustrated that these two methods produced results within 0.1% to 15% of each other and did not impact the screening results in comparison to the screening benchmarks. Therefore, in the data assessment, only summed totals excluding values that were below detection limits are discussed.

Samples that were below detection limits were excluded from the evaluation of mean and maximum concentrations at the sample locations. Key observations for detection frequency, exceedances, and maximum observed PCB and PCCD/PCDF concentrations are described by monitoring program. Overall summaries of status and observations for sediment and tissue are provided alongside the rationale for the proposed water quality objectives.

Data for constituents that impact PCB, PCDD, and PCDF toxicity and bioavailability were also collected in the majority of these monitoring programs. An assessment of potential bioavailability or toxicity due to environmental conditions was outside of the scope of this assessment, however; additional analyses would be required for confirmation.

PCBs, PCDDs and PCDFs in Marine Water: No Data Above Detection Limits

Among priority monitoring programs, data for PCBs in Burrard Inlet marine waters was limited to 12 data points from 1991 and 1993, all of which were below detection limits. In addition, no monitoring of PCDDs and/or PCDFs has been conducted for marine water samples.

Total PCBs in Sediment and Tissue

BC ENV

1991–2020 – BC ENV water quality objectives attainment monitoring samples for total PCBs collected between 1991 and 2020 were above detection limits for 17 (57%) of 30 sediment samples and 7 (23%) of 30 tissue samples. Detection limits were 0.02 µg/g for sediment and between 0.01 µg/g and 0.02 µg/g for English Sole fish tissue. Because the detection limits are three to five orders of magnitude above the

screening benchmarks, there were challenges with interpreting the findings against the screening benchmarks. Therefore, only data that were above detection limits are compared for this monitoring program with detectable concentrations interpreted as exceedances of the benchmarks. It is recognized that with such high detection limits, however, there is a high risk that existing concentrations in environmental samples that were below detection limits are actually higher than the benchmarks.

The following key points summarize the BC ENV monitoring results:

- Total PCB concentrations in sediment samples exceeded detection limits, and therefore the screening benchmark (3.7×10^{-6} µg/g dry weight), at six monitoring stations. The highest total PCB concentration recorded was 3 µg/g dry weight measured in the Inner Harbour at Vancouver Wharves (Station E207816) in 1991. This total PCB concentration far exceeds the second highest total PCB concentration in the database also measured at Vancouver Wharves (0.0589 µg/g dry weight in 2020). Data are too limited to assess temporal trends in sediment PCB concentrations. An illustration of total PCB concentrations above detection limits in the BC ENV sediment samples is provided in Figure 5.
- Total PCB concentrations in English Sole fish tissue exceeded detection limits, and therefore the screening benchmark (0.00029 µg/g wet weight) at four monitoring stations. The highest total PCB concentration recorded was 0.65 µg/g wet weight measured in the Indian Arm at Indian Arm Cable Crossing (Station 300080) in 1992. No tissue samples were collected at Vancouver Wharves, where the highest PCB concentrations in sediment were recorded, in the 1990s but samples collected in 2003 were all below detection limits (0.2 µg/g wet weight). An illustration of total PCB concentrations above detection limits in the BC ENV tissue samples is provided in Figure 6.

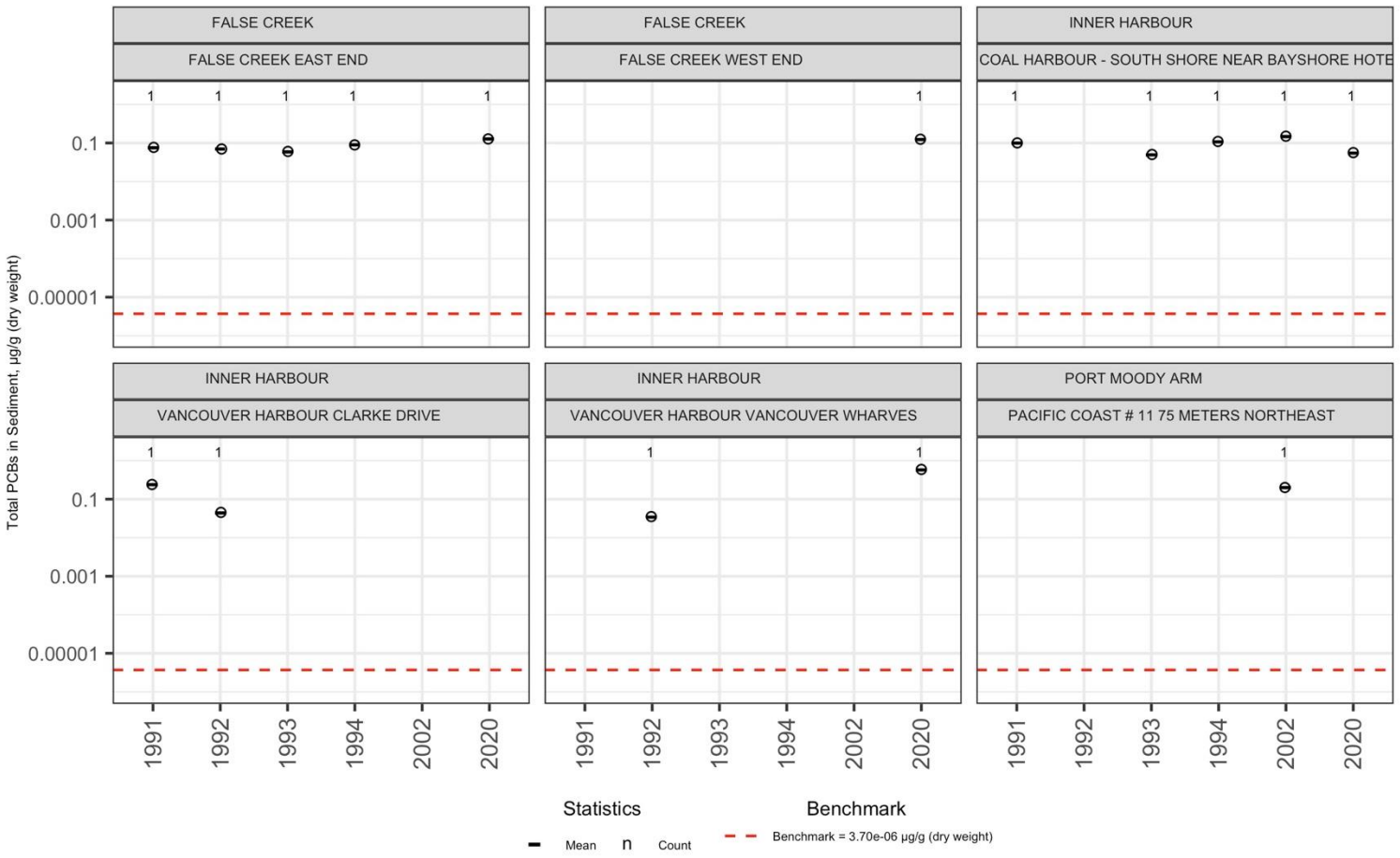

Figure 5: Total PCB concentrations in BC ENV sediment samples (1991 to 2020) in µg/g dry weight (log scale, excluding non-detects below 0.02 µg/g)

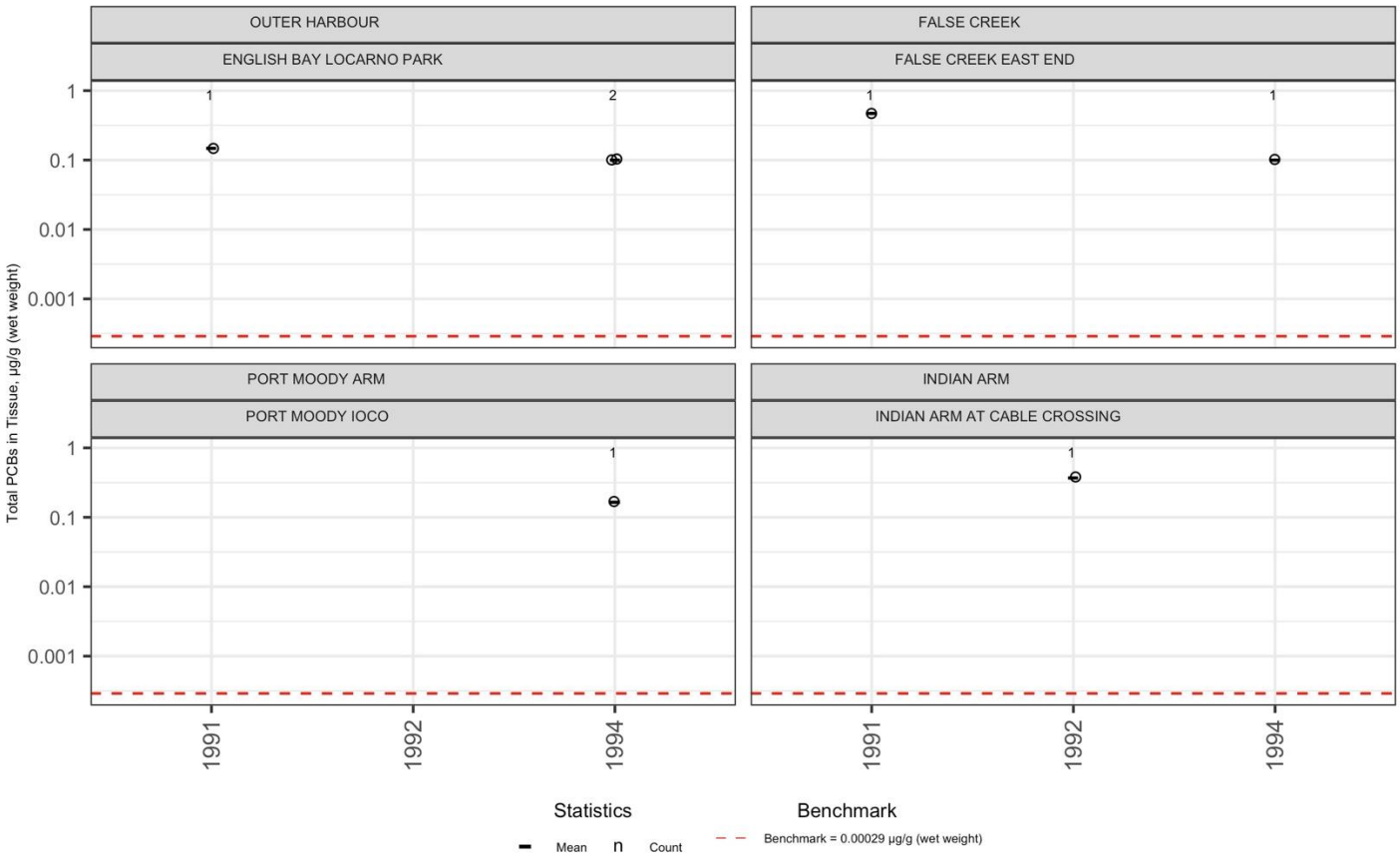

Figure 6: Total PCB concentrations in BC ENV English sole fish tissue samples (1991 to 1994) in µg/g wet weight (log scale, excluding non-detects)

ECCC Disposal at Sea

2009–2017 – ECCC Disposal at Sea monitoring samples for total PCBs collected between 2009 and 2017 were above detection limits for 6 (30%) of 20 sediment samples. Detection limits were between 0.01 µg/g and 0.1 µg/g dry weight. Because the detection limits are between four and five orders of magnitude above the screening benchmarks, there were challenges with interpreting the findings against the screening benchmarks. Therefore, only data that were above detection limits are compared for this monitoring program with detectable concentrations interpreted as exceedances of the benchmarks.

The following key points summarize the ECCC Disposal at Sea monitoring results:

- Total PCB concentrations in sediment samples exceeded detection limits, and therefore the screening benchmark (3.7×10^{-6} µg/g dry weight), at two monitoring stations in 2013 and 2016. The highest total PCB concentration recorded was 0.086 µg/g dry weight measured in Port Moody Arm in 2013 and the second highest total PCB concentration recorded was 0.083 µg/g dry weight measured in Inner Harbour in 2016. An illustration of total PCB concentrations above detection limits in the ECCC Disposal at Sea sediment samples is provided in Figure 7.

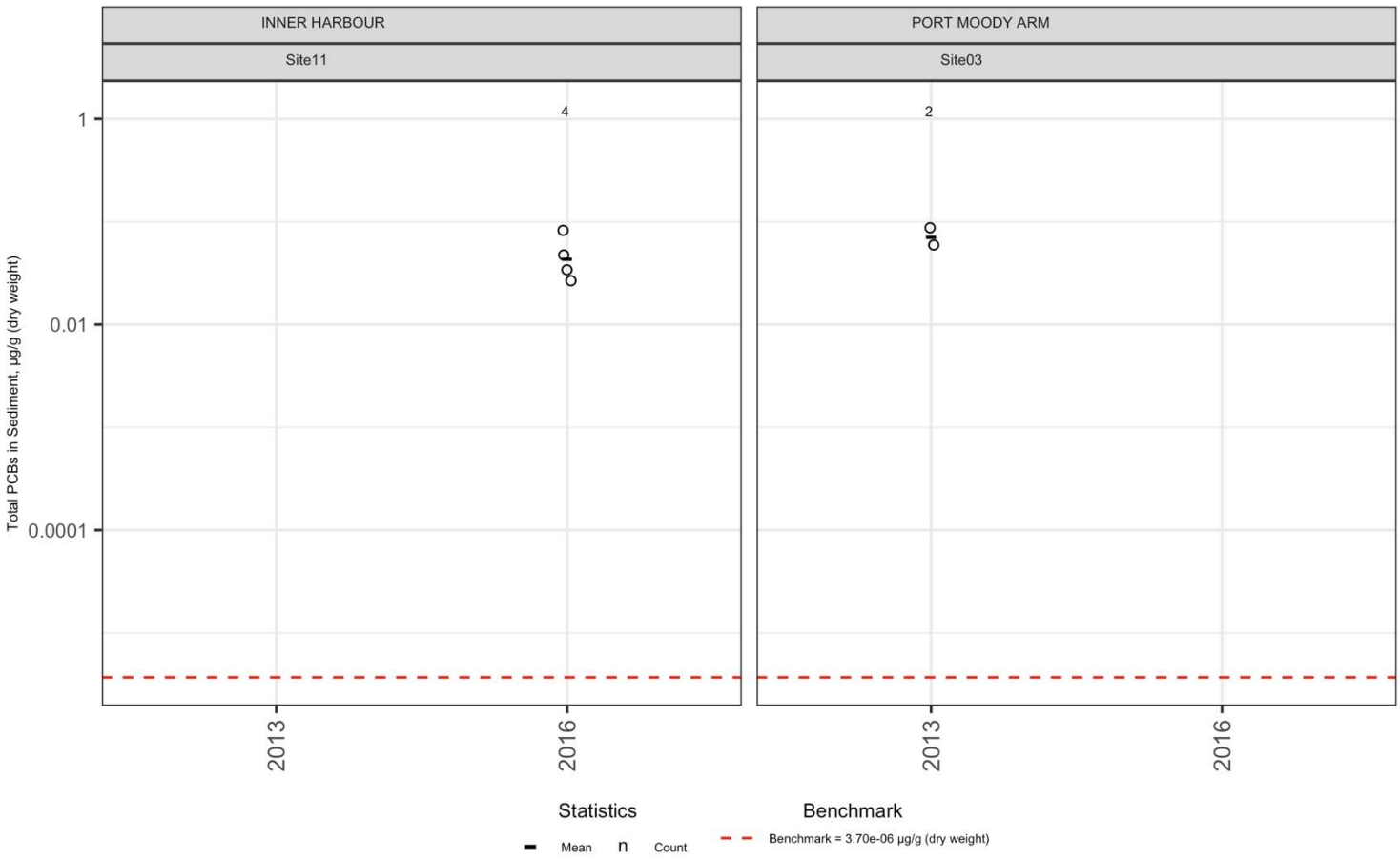

Figure 7: Total PCB concentrations in ECCC Disposal at Sea sediment samples (2009 to 2017) in µg/g dry weight (log scale, excluding non-detects)

Metro Vancouver

2007–2016 – As part of the Burrard Inlet Ambient Monitoring Program, Metro Vancouver has monitored total PCB concentrations in sediment every 2 to 3 years (Figure 8) since 2008. Total PCB concentrations in English Sole tissue (whole body, muscle, and liver) samples were measured in 2007 and 2012 (Figure 9). Between 2007 and 2016, total PCB concentrations were above detection limits in all (100%) 105 sediment samples, and 54 (77%) of 70 tissue samples. Detection limits for total PCB concentrations were 0.03 µg/g dry weight for sediment samples and were 0.03 µg/g wet weight for total PCB concentrations in tissue measured in 2007. Detection limits for PCB homolog groups were lower and varied from 1.2×10^{-8} µg/g to 5.2×10^{-4} µg/g wet weight. Homologs were summed together to determine total PCB concentrations in tissue samples in 2012. Because of the lower detection limits and the resulting greater detection frequency, greater emphasis has been placed on the Metro Vancouver monitoring data compared to the BC ENV monitoring data. Some data from the Metro Vancouver monitoring program could not be compared with benchmarks, however, because the detection limits exceeded the benchmarks.

The following key points summarize the Metro Vancouver monitoring program results:

- Total PCB concentrations exceeded the screening benchmark for total PCBs (3.7×10^{-6} µg/g dry weight) in all sediment samples in this monitoring program. The highest total PCB concentration

recorded was 0.12 µg/g dry weight measured in the Inner Harbour in 2008. An illustration of total PCB concentrations in the Metro Vancouver sediment samples is provided in Figure 8.

Total PCB concentrations in English Sole fish tissue have exceeded the screening benchmark (0.00029 µg/g wet weight) at all seven monitoring stations. The highest total PCB concentration recorded was 0.28 µg/g wet weight measured in liver samples from Central Harbour in 2012. The highest total PCB concentration measured in muscle tissue was 0.04 µg/g wet weight from Indian Arm in 2007. Generally total PCB concentrations were highest in liver samples and higher in whole body samples compared to muscle samples. An illustration of total PCB concentrations above detection limits in the Metro Vancouver samples is provided in Figure 9. While the liver samples had the highest concentrations of these contaminants, they are likely of lesser relevance for human consumption compared to muscle and whole body samples.

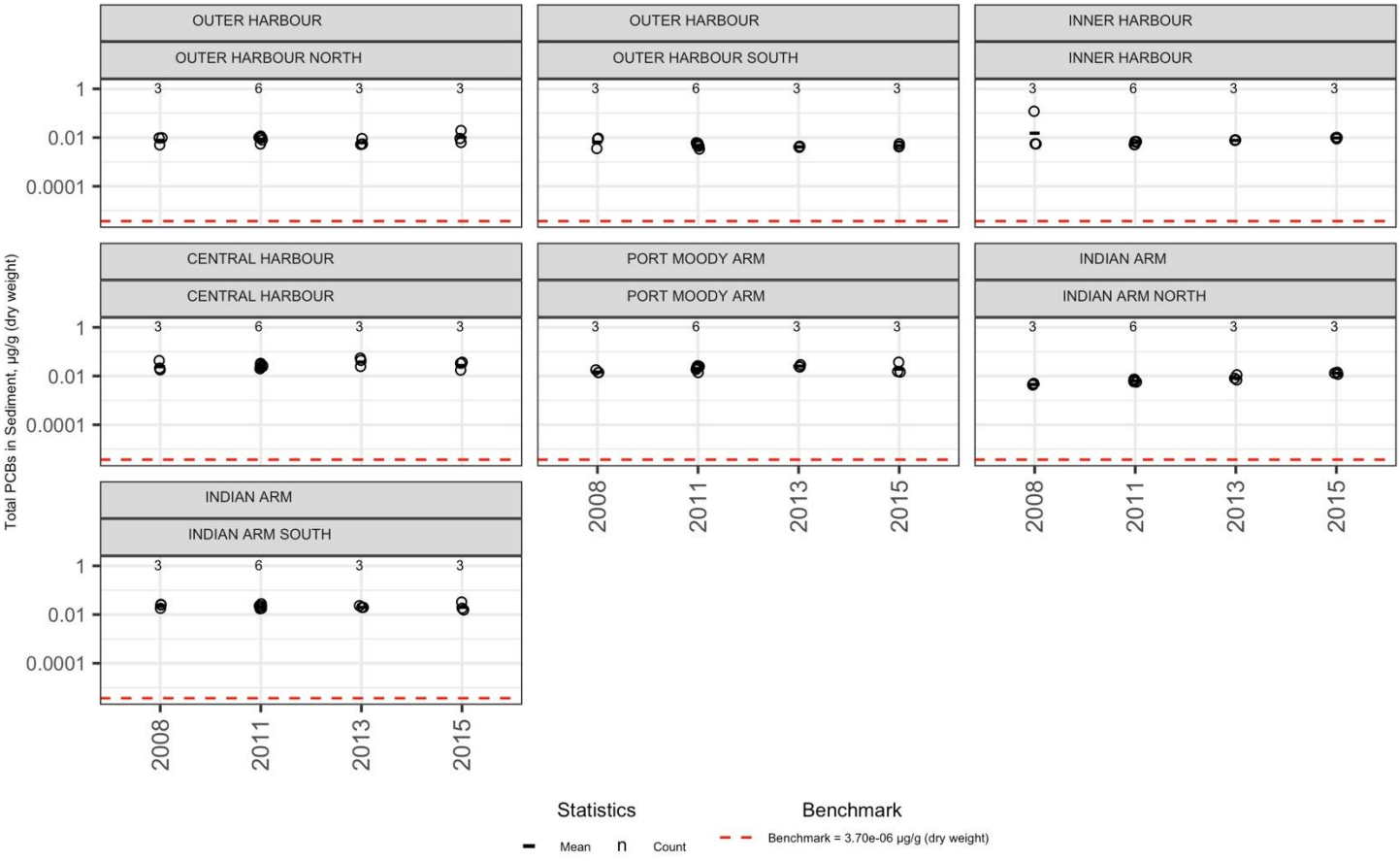

Figure 8: Total PCB concentrations in Metro Vancouver sediment samples (2008 to 2015) in µg/g (log scale)

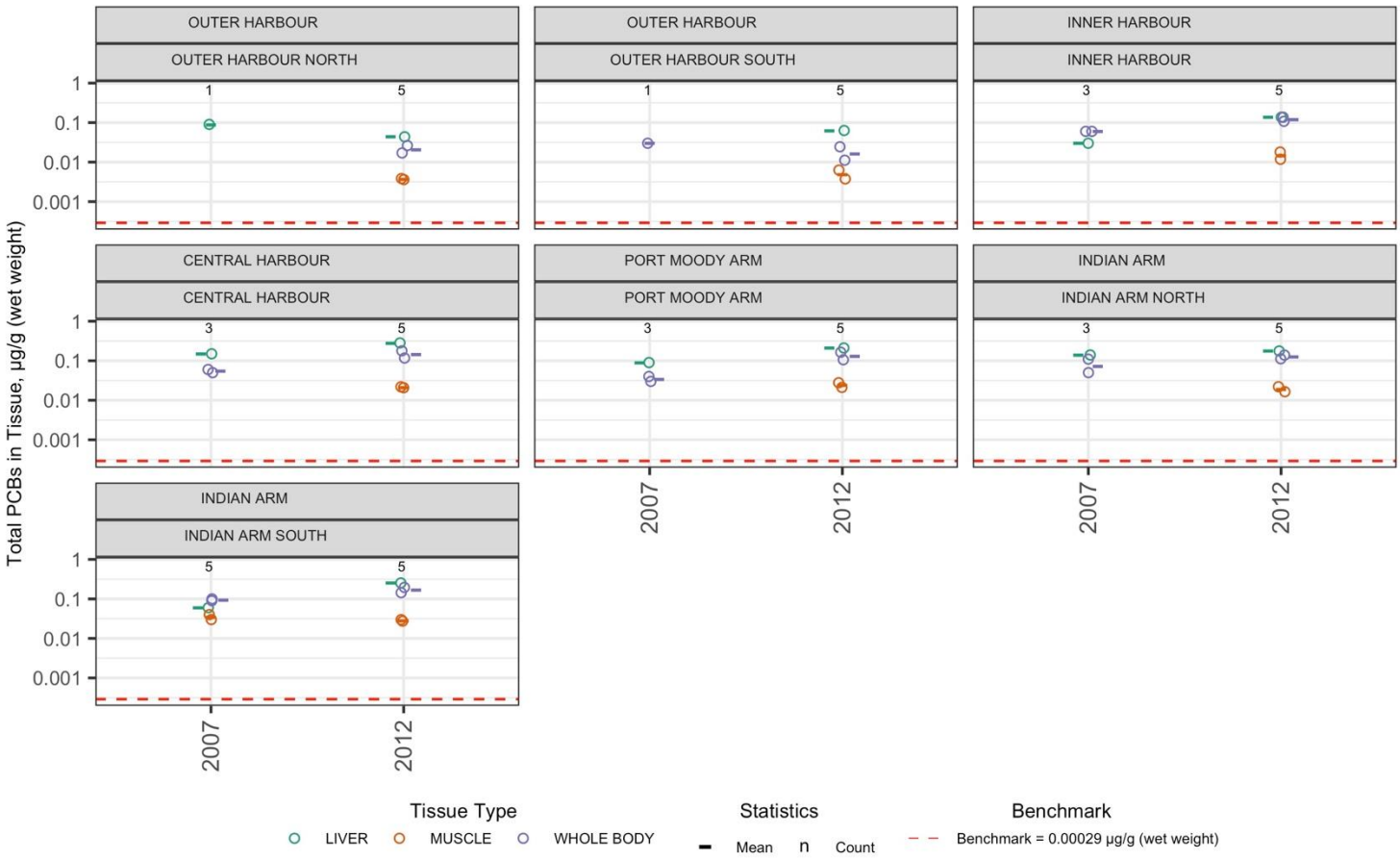

Figure 9: Total PCB concentration in Metro Vancouver English sole fish tissue samples (2007 to 2012) in µg/g (log scale, excluding non-detects)

Ocean Wise PollutionTracker Program

2015–2018 – PollutionTracker measured PCB congener concentrations in 2015, 2016 and 2018. Congener concentrations were summed together to determine total PCB concentrations. All PCB congener concentrations in sediment samples were above detection limits. Some PCB congener concentrations in tissue samples were below detection limits and were filtered from the dataset prior to summing the total PCB concentrations. The calculated median difference between the summed PCB congener totals including and excluding non-detects was 0.28%. Because the data are generally well above the screening benchmarks, it was concluded that excluding the non-detects from the calculated PCB totals was appropriate as this approach did not impact the overall findings for the screening assessment.

The following key points summarize the PollutionTracker monitoring program results:

- Total PCB concentrations in sediment consistently exceeded the screening benchmark for total PCBs (3.7×10^{-6} µg/g dry weight) in this monitoring program. The highest total PCB concentration recorded was 0.06 µg/g dry weight measured in the Inner Harbour in 2015. An illustration of total PCB concentrations in the PollutionTracker sediment samples is provided in
- Figure 10.
- Total PCB concentrations in blue mussel tissue have exceeded the screening benchmark (0.00029 µg/g wet weight) at all monitoring stations. The highest total PCB concentration recorded was

0.00068 µg/g wet weight measured in Inner Harbour in 2018. Total PCB concentrations that were above detection limits in the PollutionTracker samples are illustrated in Figure 11.

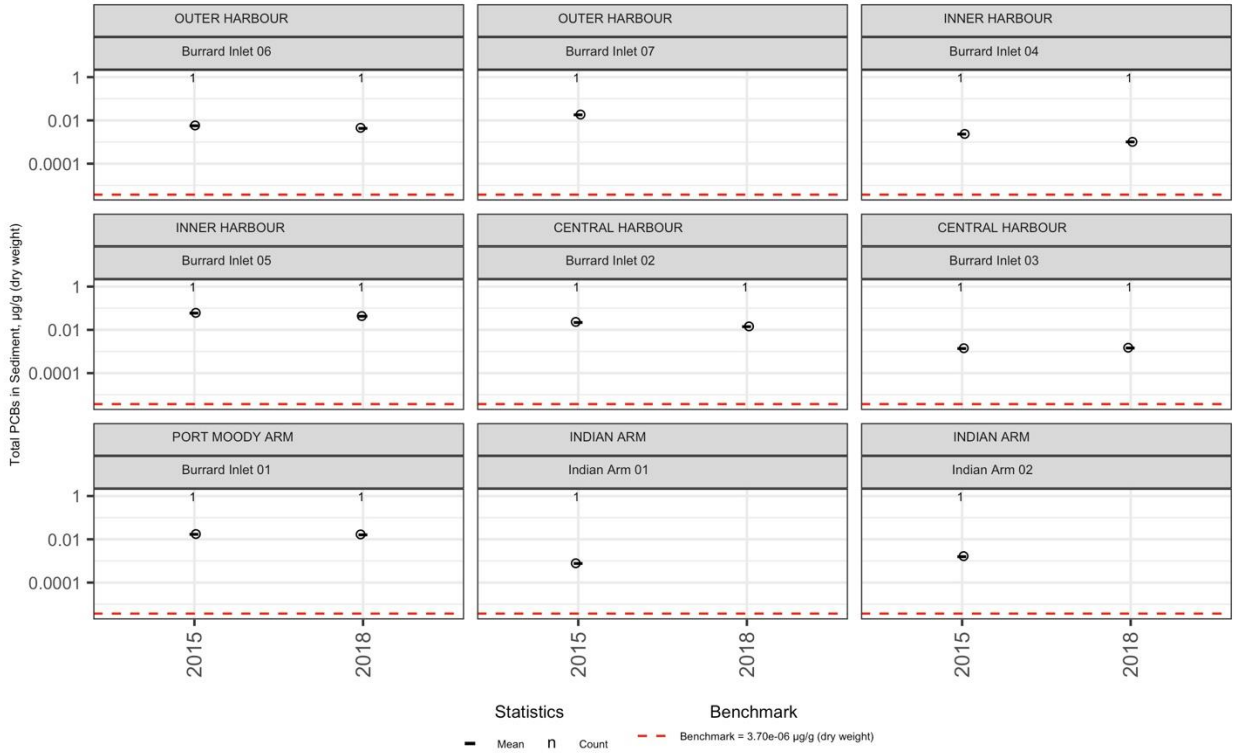

Figure 10: Total PCB concentrations PollutionTracker sediment samples (2015, 2016 and 2018) in µg/g (log scale)

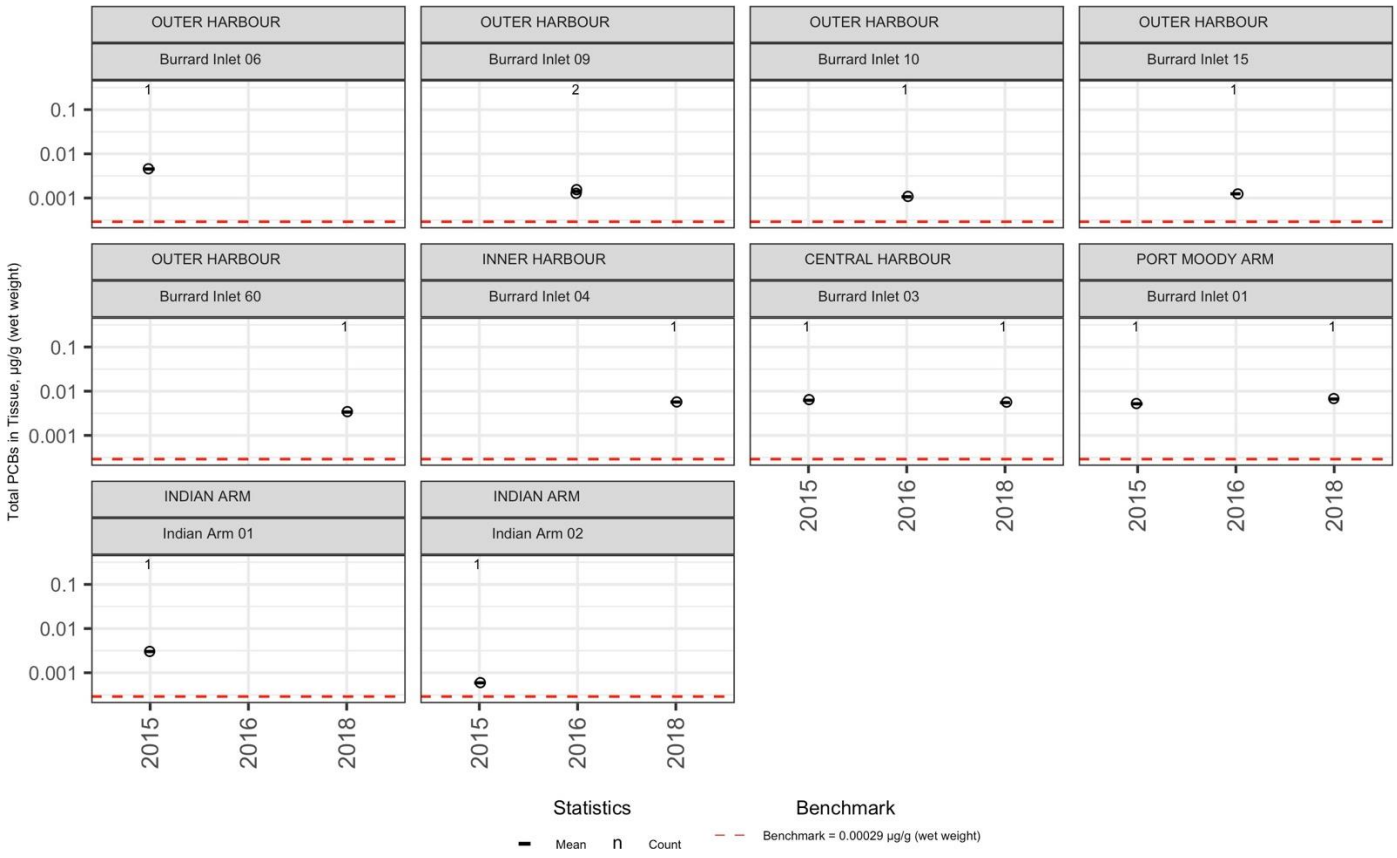

Figure 11: Total PCB concentrations in PollutionTracker blue mussel tissue samples (2015, 2016, and 2018) in µg/g (log scale, excluding non-detects)

Total PCDDs and PCDFs in Sediment

BC ENV

1991–2020 – BC ENV water quality objectives attainment monitoring samples for total PCDDs and PCDFs collected in 2020 were above detection limits in all (100%) 4 sediment samples though specific detection limits were not provided in the data transfer.

The following key points summarize the BC ENV monitoring results:

- Total PCDD and PCDF concentrations in sediment samples exceeded the screening benchmarks (TEL = 8.5×10^{-7} µg/g dry weight; PEL = 2.15×10^{-5} µg/g dry weight), at all four monitoring stations where samples were collected in 2020. The highest total PCDD and PCDF concentration recorded was 0.0044 µg/g dry weight measured in the Inner Harbour at Clarke Drive (Station E207818). An illustration of total PCDD and PCDF concentrations in the BC ENV sediment samples is provided in Figure 12.

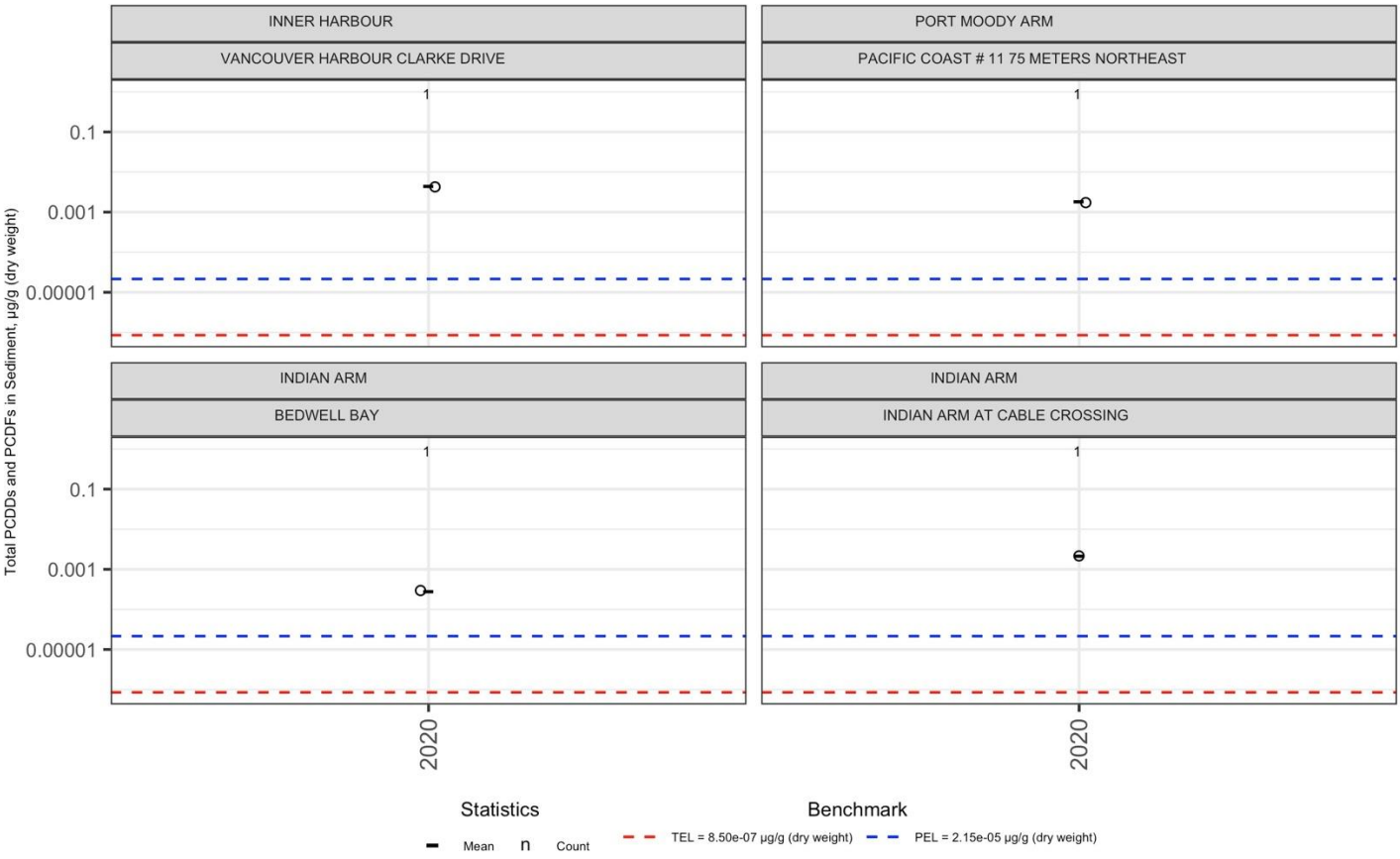

Figure 12: Total PCDD and PCDF concentrations in BC ENV sediment samples (2020) in µg/g (log scale)

Metro Vancouver

2007–2016 – As part of the Burrard Inlet Ambient Monitoring Program, Metro Vancouver has monitored PCDD and PCDF homolog and congener concentrations in sediment every 2 to 3 years (Figure 8) since 2008. Homolog concentrations were summed together to determine total PCDD and PCDF concentrations. All PCDD and PCDF homolog concentrations in sediment samples were above detection limits. Some PCDD of PCDF homolog concentrations in sediment samples were below detection limits and were filtered from the dataset prior to summing the total PCDD and PCDF concentrations; however, this had a negligible impact on the calculated totals with the greatest difference between including and excluding the non-detects calculated to be 0.22%.

The following key points summarize the Metro Vancouver monitoring program results:

- Total PCDD and PCDF concentrations exceeded the TEL and PEL screening benchmark for total PCDD and PCDFs (8.5×10^{-7} µg/g and 2.15×10^{-5} µg/g, respectively) in sediment in this monitoring program. The highest total PCDD and PCDF concentration recorded was 0.00416 µg/g dry weight measured in Indian Arm South in 2013. An illustration of total PCDD and PCDF concentrations in the Metro Vancouver sediment samples is provided in Figure 13.

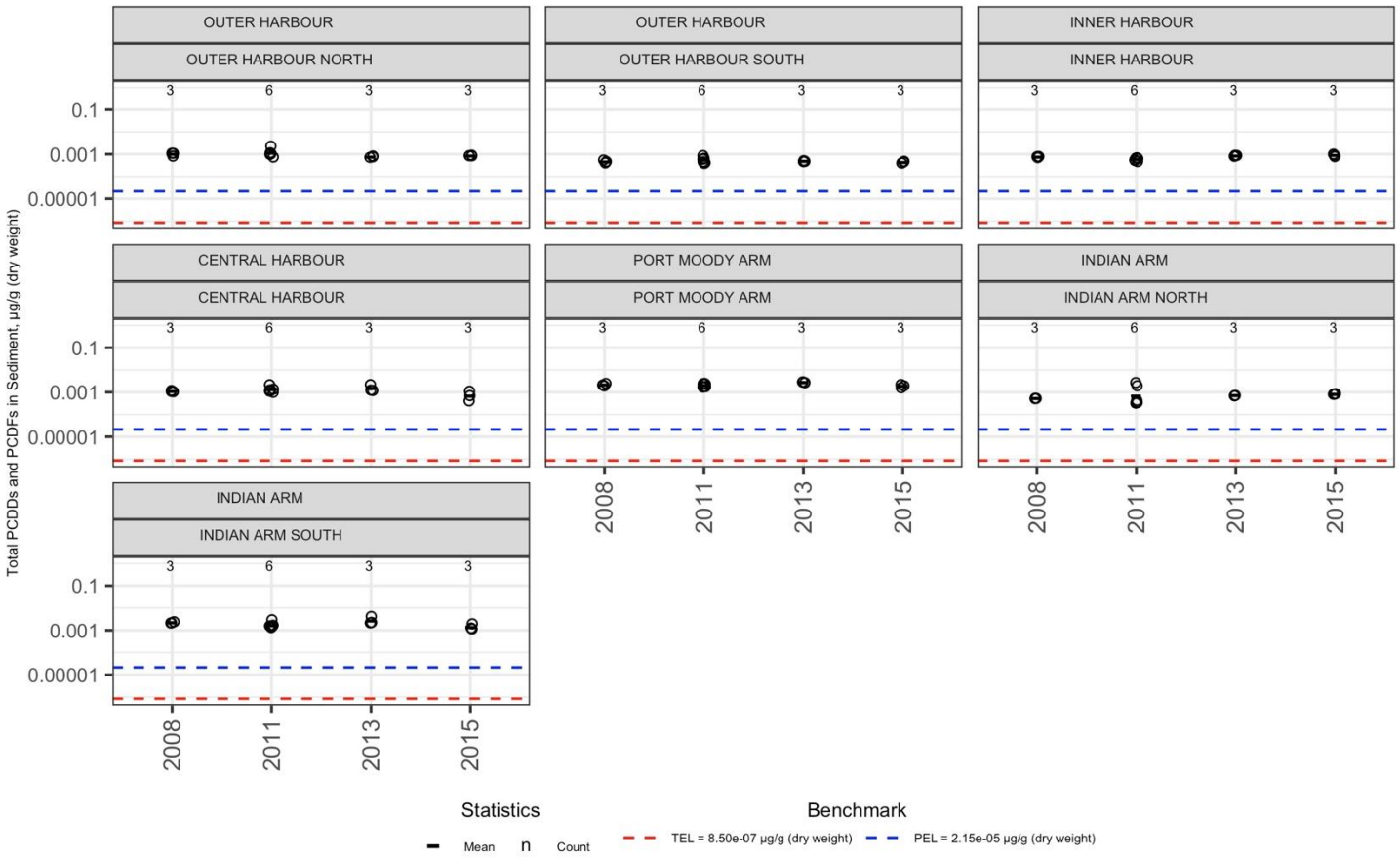

Figure 13: Total PCDD and PCDF concentrations in Metro Vancouver sediment samples (2008 to 2015) in µg/g (log scale)

Ocean Wise PollutionTracker Program

2015–2018 – PollutionTracker measured PCDD and PCDF congener concentrations in 2015 and 2018. Congener concentrations were summed together to determine total PCDD and PCDF concentrations. All PCDD and PCDF homolog concentrations in sediment samples were above detection limits. Detection limit values were not listed for PCDDs and PCDFs data from this monitoring program.

The following key points summarize the PollutionTracker monitoring program results:

- Total PCDD and PCDF concentrations exceeded the TEL and PEL screening benchmark for total PCDD and PCDFs (8.5×10^{-7} µg/g and 2.15×10^{-5} µg/g, respectively) in sediment in this monitoring program. The highest total PCDD and PCDF concentration recorded was 0.0026 µg/g measured in Port Moody Arm in 2015. An illustration of total PCDD and PCDF levels in the PollutionTracker sediment samples is provided in Figure 14.

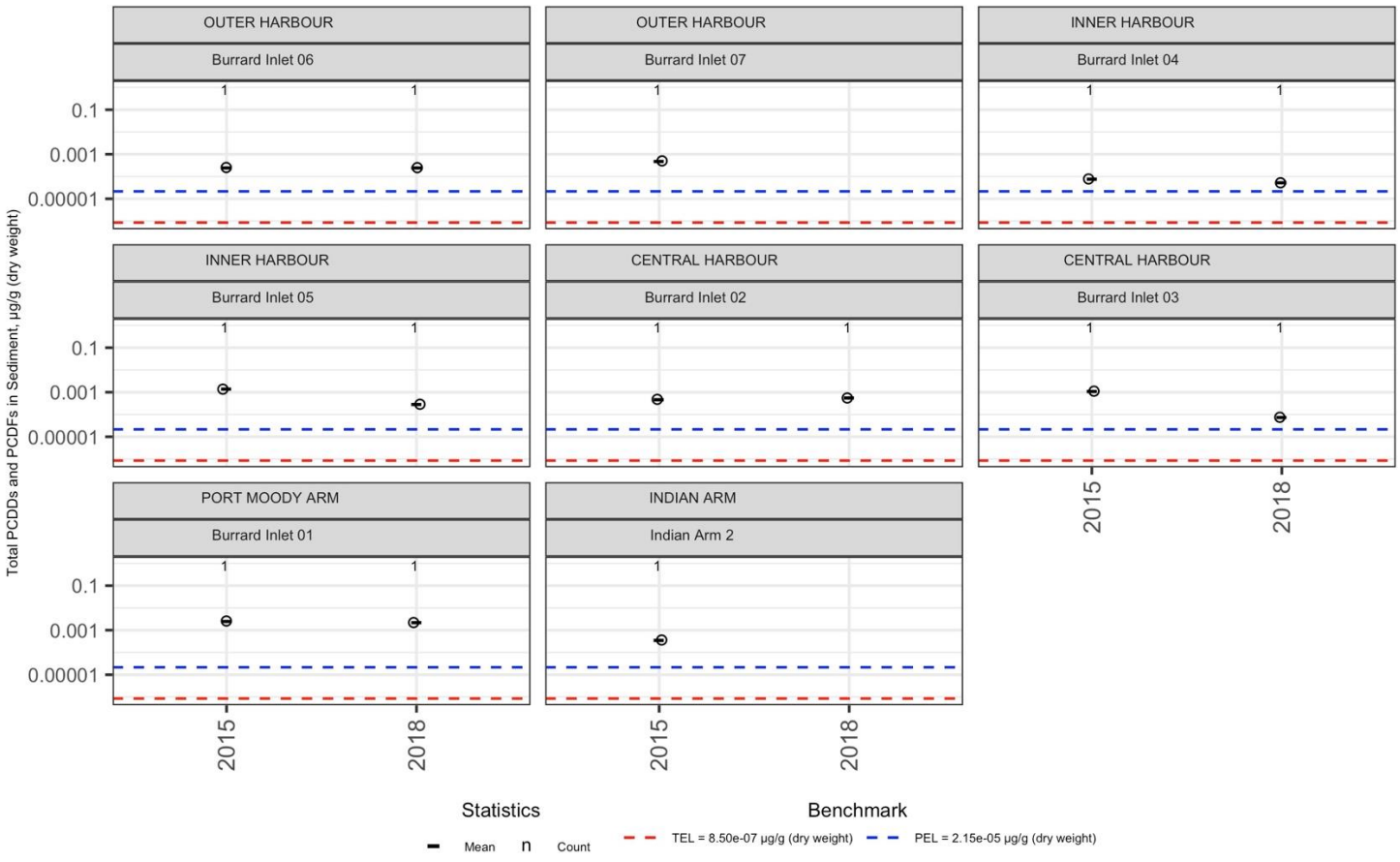

Figure 14: Total PCDD and PCDF concentrations in PollutionTracker sediment samples (2015, 2016 and 2018) in µg/g (log scale)

Non-Dioxin-Like PCBs in Tissue

Because TRVs for human health were available for non-dioxin-like PCBs specifically, rather than for total PCBs, a data analysis was conducted for this subset of PCBs to inform further conversations on human health risk assessment.

Metro Vancouver

2007–2016 – As part of the Burrard Inlet Ambient Monitoring Program, Metro Vancouver has monitored non-dioxin-like PCB congeners in English Sole tissue (whole body, muscle, and liver) samples in 2007 and 2012 (Figure 15). Detection limits for congeners ranged from 1.3×10^{-8} µg/g to 3.0×10^{-2} µg/g wet weight and concentrations above detection limits were only found in 2012. Including or excluding data below detection limits in the calculation of total non-dioxin-like PCBs had a negligible impact on the totals.

The following key points summarize the Metro Vancouver monitoring program results:

- Non-dioxin-like PCB concentrations calculated excluding non-detects for English Sole fish tissue have exceeded the screening benchmarks in samples collected at all monitoring stations. The highest total non-dioxin-like PCB concentration recorded was 0.260 µg/g wet weight excluding non-detects measured in liver samples from Central Harbour in 2012. The highest total non-dioxin-like PCB concentrations measured in muscle tissue was 0.275 µg/g wet weight excluding non-detects from Indian Arm South in 2012. Generally total non-dioxin-like PCB concentrations were highest in liver

samples and higher in whole body samples compared to muscle samples. An illustration of total non-dioxin-like PCB concentrations excluding non-detects in the Metro Vancouver samples is provided in Figure 15. While the liver samples had the highest concentrations of these contaminants, they are likely of lesser relevance for human consumption compared to muscle and whole body samples.

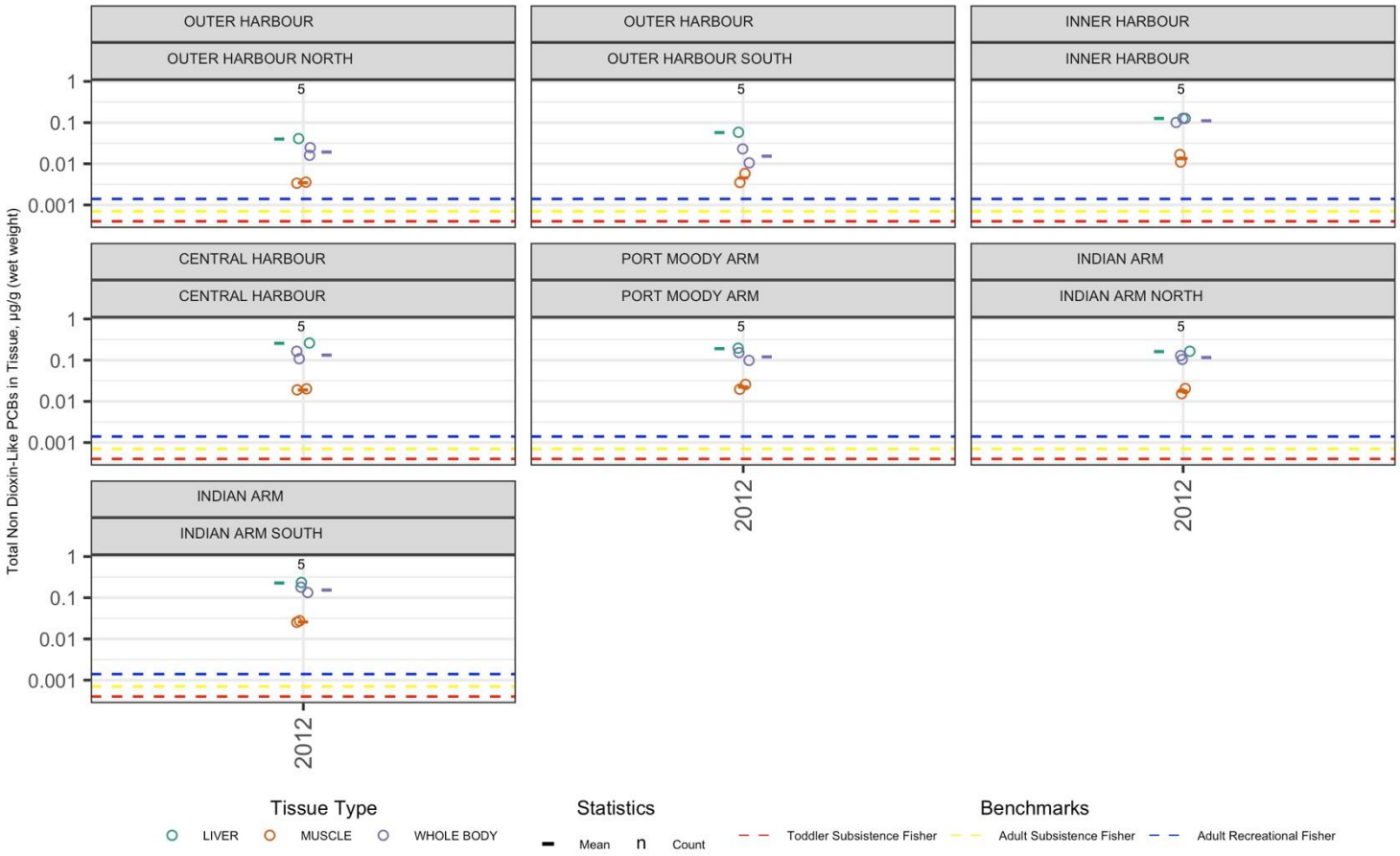

Figure 15: Total Non-Dioxin-Like PCBs concentrations in Metro Vancouver English sole fish tissue samples (2007 to 2012) in µg/g (log scale, excluding non-detects)

Ocean Wise PollutionTracker Program

2015–2018 – PollutionTracker has monitored non-dioxin-like PCBs congeners in blue mussel tissue samples in 2015, 2016, and 2018 (Figure 16). Detection limits for the congeners ranged from 4.83×10^{-8} µg/g wet weight to 1.16 µg/g wet weight. Including or excluding detection limits in the calculation of total non-dioxin-like PCBs had a negligible impact on the totals relative to the screening benchmarks.

The following key points summarize the PollutionTracker monitoring program results:

- Non-dioxin-like PCB concentrations calculated excluding non-detects for blue mussel tissue have exceeded the toddler subsistence screening benchmark (0.0004 µg/g wet weight) in samples collected at all monitoring stations. Non-dioxin-like PCB concentrations calculated excluding non-detects for blue mussel tissue have exceeded the adult subsistence and adult recreational screening benchmark (0.0007 µg/g and 0.0014 µg/g, wet weight respectively) in samples collected at most monitoring stations. The highest total non-dioxin-like PCB concentration recorded was 0.00630 µg/g

wet weight excluding non-detects measured in Port Moody Arm in 2018. An illustration of total non-dioxin-like PCB concentrations excluding non-detects in the PollutionTracker samples is provided in Figure 16.

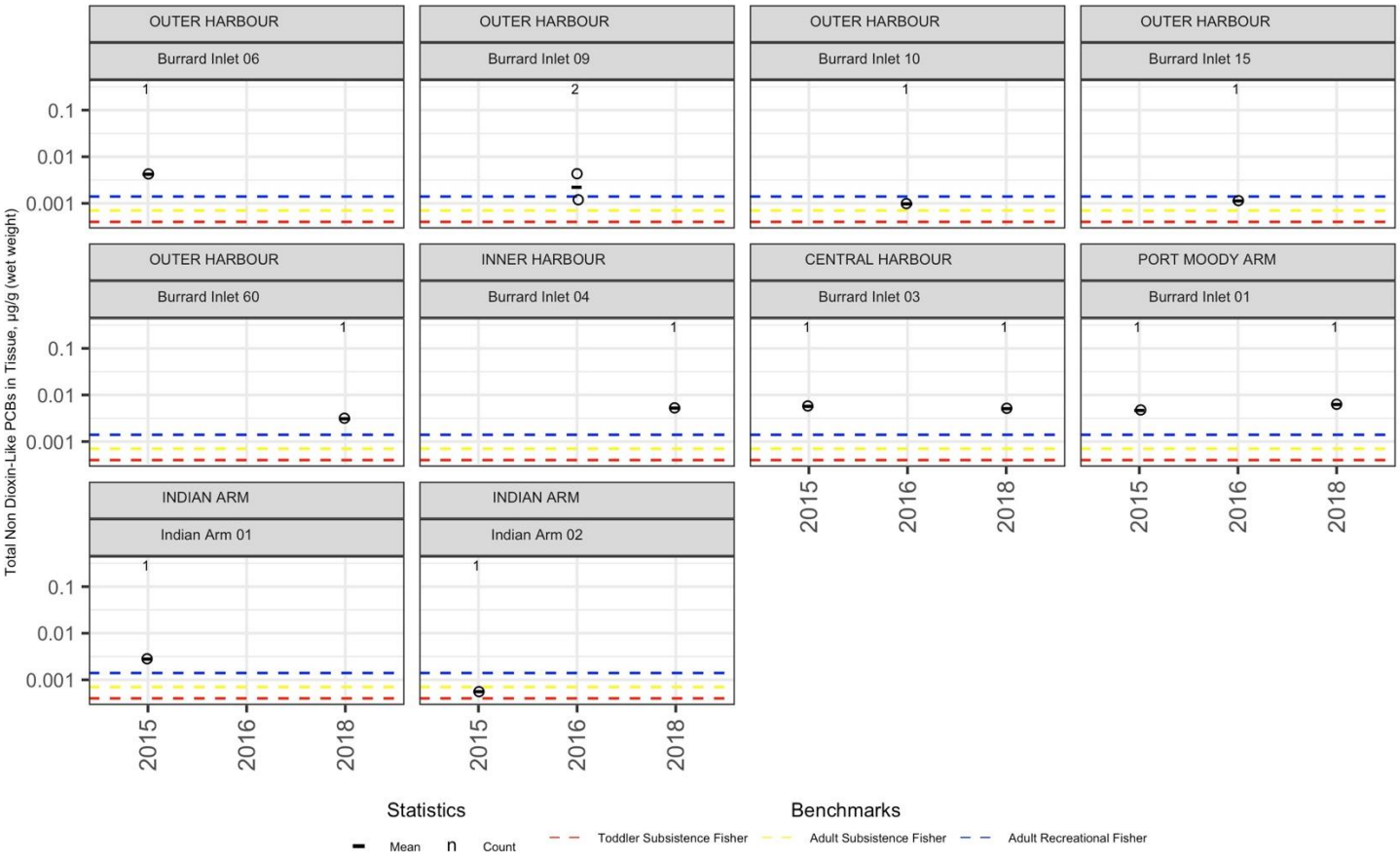

Figure 16: Total Non-Dioxin-Like PCB concentrations in PollutionTracker blue mussel tissue samples (2015, 2016, and 2018) in µg/g (log scale, excluding non-detects)

2,3,7,8-TCDD Equivalency of Total PCDDs, PCDFs, and Dioxin-Like PCBs in Tissue

Metro Vancouver

2007–2016 – As part of the Burrard Inlet Ambient Monitoring Program, Metro Vancouver has monitored dioxin-like PCBs, PCDDs, and PCDFs congeners with equivalency to 2,3,7,8-TCDD in English Sole tissue (whole body, muscle, and liver) samples in 2007 and 2012 (Figure 17). Detection limits for congeners ranged from 1.3×10^{-8} µg/g wet weight to 3.0×10^{-2} µg/g wet weight. Including or excluding detection limits in the calculated equivalencies to 2,3,7,8-TCDD had a negligible impact on the calculated totals relative to the screening benchmarks.

The following key points summarize the Metro Vancouver monitoring program results:

- Total 2,3,7,8-TCDD equivalency levels calculated excluding non-detects for English Sole fish tissue have exceeded the screening benchmarks for adults (≥1.6×10^{-7} µg/g wet weight) in samples collected at all monitoring stations. The highest total 2,3,7,8-TCDD equivalency level recorded was 5.7×10^{-6} µg/g wet weight excluding non-detects measured in liver samples from Indian Arm in 2007. The highest total 2,3,7,8-TCDD equivalency level measured in muscle tissue was 7.5×10^{-7} µg/g wet

weight excluding non-detects from Indian Arm in 2007. Generally total 2,3,7,8-TCDD equivalency levels were highest in liver samples and higher in whole body samples compared to muscle samples. An illustration of total 2,3,7,8-TCDD equivalency levels excluding non-detects in the Metro Vancouver samples is provided in Figure 17. While the liver samples had the highest concentrations of these contaminants, they are likely of lesser relevance for human consumption compared to muscle and whole body samples.

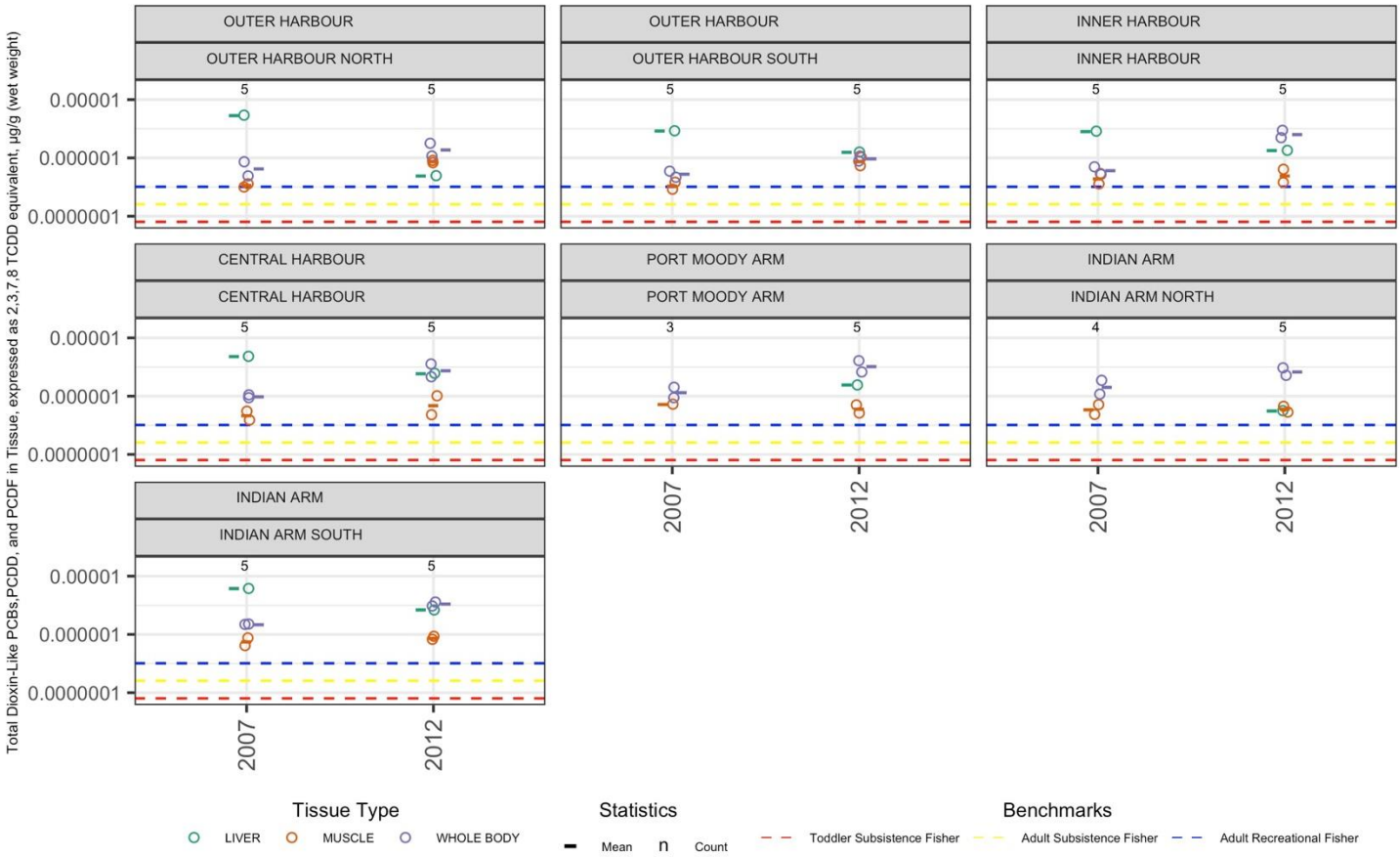

Figure 17: 2,3,7,8-TCDD Equivalency Levels of Total Dioxin-Like PCBs, PCDDs, and PCDFs in Metro Vancouver English sole fish tissue samples (2007 to 2012) in µg/g (log scale, excluding non-detects)

Ocean Wise PollutionTracker Program

2015–2018 – PollutionTracker has monitored dioxin-like PCBs, PCDDs, and PCDFs congeners with equivalency to 2,3,7,8-TCDD in blue mussel tissue samples in 2015, 2016, and 2018 (Figure 18). Detection limits for the congeners ranged from 4.83×10^{-8} µg/g wet weight to 1.16 µg/g wet weight. Including or excluding detection limits in the calculated equivalencies to 2,3,7,8-TCDD had a negligible impact on the calculated totals relative to the screening benchmarks.

The following key points summarize the PollutionTracker monitoring program results:

- Total 2,3,7,8-TCDD equivalency levels calculated excluding non-detects for blue mussel tissue have exceeded the toddler subsistence screening benchmark (8.0×10^{-8} µg/g wet weight) in all samples collected at all monitoring stations. The highest total 2,3,7,8-TCDD equivalency level recorded was 5.11×10^{-6} µg/g wet weight excluding non-detects measured in Outer Harbour in 2016. An illustration

of total 2,3,7,8-TCDD equivalency levels excluding non-detects in the PollutionTracker samples is provided in Figure 18.

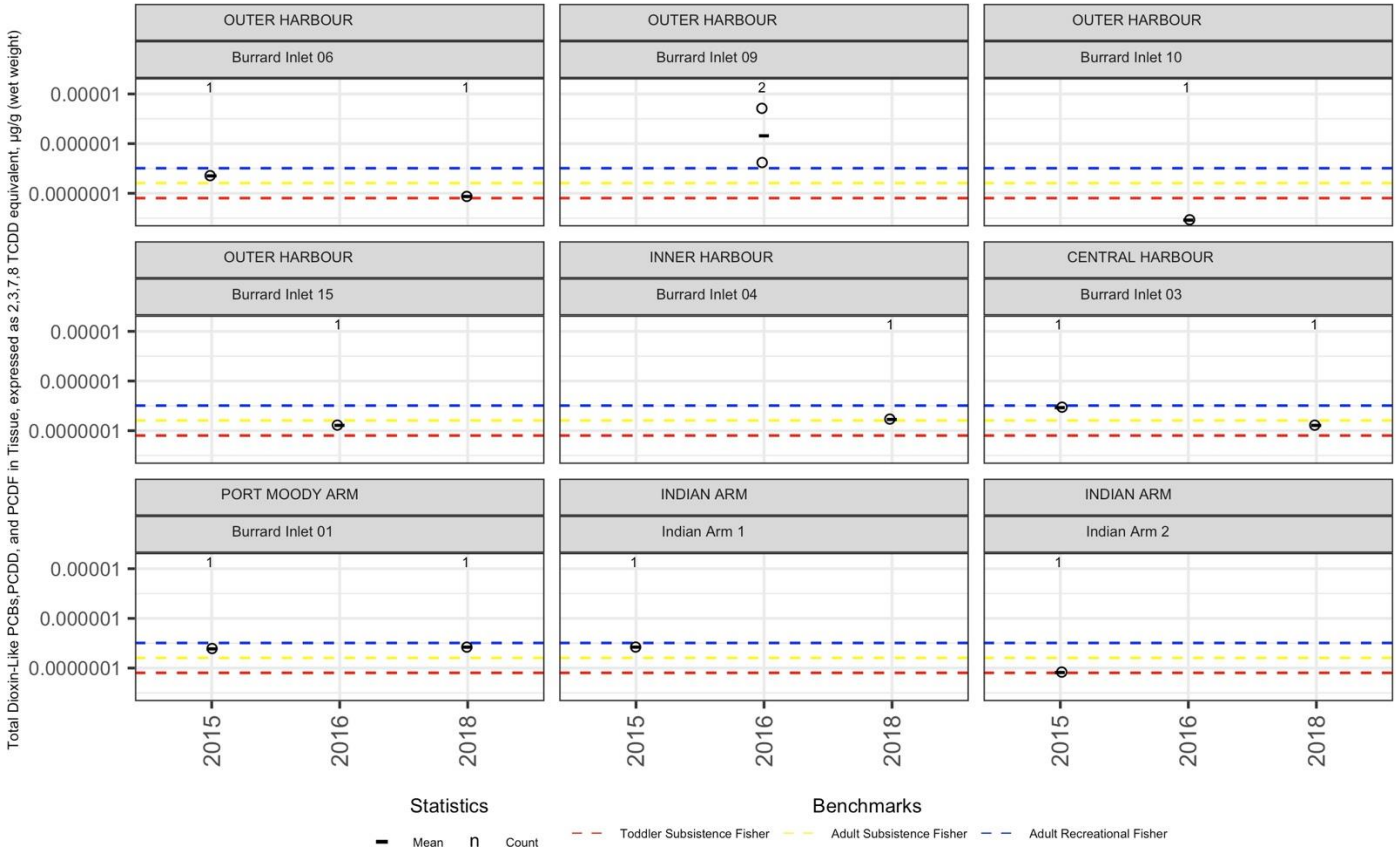

Figure 18: 2,3,7,8-TCDD Equivalency Levels of Total Dioxin-Like PCBs, PCDDs, and PCDFs in PollutionTracker blue mussel tissue samples (2015, 2016, and 2018) in µg/g (log scale, excluding non-detects)

3.4 Knowledge Gaps and Research Needs

Data on PCBs, PCDDs and PCDFs in the waters of Burrard Inlet is limited or lacking. The few data points that exist for PCBs in marine water are out of date and likely based on sampling methodology that was insufficient for comparison against current benchmarks. Data should be collected and compared against existing benchmarks to determine whether these contaminants are at levels of concern in the water column.

Because of differences in sampling methodologies, reporting, and detection limits between years and programs, it is difficult to interpret whether temporal and spatial differences are a result of actual differences in PCB, PCDD and PCDF levels in the environment, or a result of improvements or differences in the sampling methodology and/or analysis.

Lower laboratory detection limits are needed to better understand the inputs of PCBs, PCDDs and PCDFs to the marine environment and their potential effects.

Existing data on sediment and tissue samples collected from Burrard Inlet are limited and do not allow for an assessment of temporal trends in detected concentrations. As PCBs have been banned for several decades, and emissions of PCDDs and PCDFs have been mitigated in many cases, their occurrence in

Burrard Inlet could be expected to decline. Knowledge of temporal trends are important for setting achievable objectives and understanding the need for additional mitigative measures to further reduce the occurrence of PCBs, PCDDs, and PCDFs in Burrard Inlet. In addition, mitigative measures could become less effective if spatial differences, pollution hotspots and potential sources are not identified.

An understanding of contaminant levels in preferred food species are needed to enable a human health risk assessment and maximum recommended daily intake of seafood.

4. PROPOSED OBJECTIVES FOR PCBS, PCDDS AND PCDFS IN BURRARD INLET

4.1 Proposed Objectives

Proposed objectives for PCBs, PCDDs and PCDFs are presented in Table 5.

Table 5: Proposed Water Quality Objectives for PCBs, PCDDs and PCDFs

Sub-basin	Outer Harbour	False Creek	Inner Harbour	Central Harbour	Port Moody Arm	Indian Arm
Water						
Total PCBs in Water (interim screening benchmark)	*6.2 x 10^{-4} ng/L single sample maximum[1]*					
2,3,7,8-TCDD[7] in Marine Water (interim screening benchmark)	*5.1 x 10^{-6} ng/L single sample maximum[2]*					
Sediment						
Total PCBs in Sediment	3.7 x 10^{-6} µg/g dry weight single-sample maximum[3,4]					
Total PCDDs and PCDFs in Sediment	8.5 x 10^{-7} µg/g dry weight single sample maximum[,4,5]					
Tissue[5]						
Total PCBs in Tissue	2.9 x 10^{-4} µg/g wet weight single-sample maximum[3,6]					
Total PCDDs, PCDFs, and Dioxin-like (planar) PCBs in Tissue	8.0 x 10^{-8} µg/g wet weight single-sample maximum[6,8]					
All						
All PCBs, PCDDs and PCDFs in all media	Decreasing concentrations					

[1]Calculated using method from F. Gobas (*pers. comm.* 2021); not currently proposed as an objective, pending improvements in laboratory detection limits.
[2]Adopted from US EPA (2002); not currently proposed as an objective, pending improvements in laboratory detection limits.
[3]From BC ENV 2020, adopted from Alava et al. 2012.
[4]Based on at least 1 composite sample consisting of at least 3 replicates.
[5]From BC ENV 2020, adopted from CCME 2001a. Expressed on a toxic equivalency basis using toxic equivalent factors for fish (CCME 2001a).
[6] Applies to all tissue types. Based on at least 1 composite sample consisting of at least 5 fish or 25 bivalves. See Rao et al. (in prep) for additional details.
[7]2,3,7,8-tetrachlorodibenzo-*p*-dioxin, the most toxic of the PCDDs and PCDFs.
[8]PCDDs, PCDFs, and dioxin-like PCBs are assessed by converting their concentrations to units of 2,3,7,8-TCDD toxic equivalence (TEQ) using toxic equivalency factors (TEFs) applied to human health. These TEFs are published in Van den Berg et al. 2006 and included in Health Canada 2021. The sum of the TEQs is then compared to the objective. TEFs are provided in Table 3.

4.2 Rationale

Water

The two benchmarks available for total PCBs in marine water (section 3.1) are not proposed as objectives, because they do not consider chronic effects, and are inconsistent with the proposed sediment objective, which is more protective of higher trophic level species. A proposed interim benchmark for total PCBs in water was back-calculated from the proposed sediment objective as follows (from Frank Gobas, Simon Fraser University, *pers. comm.*, 2021 and 2022):

PCB concentration in sediment = 0.0037 µg/kg dw for sediments with a 1% organic carbon content

⇨ PCB concentration = 0.0037 ÷ 0.01 = 0.37 µg/kg organic carbon

Divide this by the organic carbon/water partition coefficient of PCB ≈ $10^{6.6}$ x 0.35 = 1.4 x 10^{6} L/kg organic carbon:[5]

⇨ Equilibrium concentration of PCBs in water = 0.37 µg/kg ÷ 1.4 x 10^{6} L/kg = 2.6 x 10^{-7} µg/L

This is the dissolved concentration of PCBs in water, which can be expressed in terms of a total (unfiltered) concentration of PCBs in water by incorporating sorption of PCBs to particulate organic carbon in the water. To do this, we can approximate the fraction of the PCBs that are dissolved and not associated with particulate matter in the water column as:

⇨ Fraction of PCB dissolved in water = 1 ÷ (1 + $10^{6.6}$ × 0.35 × 1x10^{-6}) = 0.26 (using 1.0 mg/L particulate organic carbon in water based on the average of available monitoring data for Burrard Inlet.[6])

⇨ This means that the total concentration of PCBs in water at equilibrium with the concentration of PCB in sediment at the sediment quality guideline is approximately:

2.6x10^{-7} µg/L / 0.26 = 6.2x10^{-7} µg/L

The lowest available benchmark for total PCBs in water (6.4.10^{-5} ug/L, US EPA 2002) is about 100 times greater than this calculated value of 6.2x10^{-7} ug/L. Back-calculating the water objective from the sediment objective, as done above, aims to avoid situations where the water objective is not exceeded while the sediment objective is exceeded.

In the absence of local guidelines for PCDDs or PCDFs in marine water, an interim benchmark for the dioxin 2,3,7,8-TCDD in marine water of 5.1×10^{-9} µg/L from the US EPA (2002) is recommended, which is intended to be protective of human consumption of seafood. The US EPA (2002) evaluated 2,3,7,8-TCDD as a non-threshold (carcinogenic) contaminant. At this time, there is no Canadian equivalent calculation for the derivation of a non-carcinogenic screening value for 2,3,7,8-TCDD in marine water.

[5] Assuming the log K_{ow} of PCBs is adequately represented by 6.6 and the factor of 0.35 represents the correlation between the organic carbon/water partition coefficient (K_{oc}) and the K_{ow}, from Seth et al. (1999). Mackay et al. (2006) list a log K_{ow} for Aroclor 1254 of 6.1 to 6.8. PCB mixtures in fish and killer whales tend to resemble a PCB mixture similar to Aroclor 1254 to 1260, i.e. a heavier mixture, as the lower chlorinated PCBs are more quickly eliminated (F. Gobas, *pers. comm.* July 2022). Mackay (2001: 47) recommends the use of an estimated log K_{ow} for total PCBs of 6.6.

[6] Measured carbon levels vary widely, so a conservative approach was used to arrive at a numeric value. Average particulate organic carbon (OC) of 1 mg/L was calculated by matching the samples in the available data for Burrard Inlet that had both dissolved and total OC measurements, calculating the particulate OC (= total OC – dissolved OC), and averaging all calculated particulate OC values that were greater than zero. An alternative approach, given the complexity of the data, would be to assume that 50% of the total OC is particulate OC; because the average total OC in the available data for Burrard Inlet is 2 mg/L, this approach results in the same value of 1 mg/L particulate OC.

The BC guidelines for specific dioxin-like PCBs in water (PCB 105: 0.09 ng/L, PCB 126: 0.00025 ng/L, PCB 169: 0.06 ng/L, PCB 77: 0.04 ng/L), from the BC approved water quality guidelines (BC ENV 2019) are not proposed as objectives, as they would be inconsistent with the proposed objective for total PCBs in marine water. An objective for total PCBs in marine water provides a fuller picture of toxic risk including both dioxin-like and non-dioxin-like effects. The concentrations of dioxin-like PCBs in water could potentially be analyzed within toxic equivalency analyses, using the proposed objective for 2,3,7,8-TCDD.

The values for PCBs, PCDDs and PCDFs in water are proposed as interim single sample maximum benchmarks rather than objectives because data are limited for these contaminants, and because laboratory methods are currently unable to detect PCBs, PCDDs and PCDFs at such low levels. A subcommittee of the Southern Resident Killer Whales contaminants technical working group is developing updated PCB guidelines for sediment, water, and fish tissue that are expected to differ from those specified in ECCC (2021) to be more protective of high trophic level marine mammals. The water quality guideline under development by this subcommittee will likely also be very low and undetectable by current laboratory methods (A. Tillmanns, BC ENV, *pers. comm.*, June 2022). As described in section 2.2.1, stormwater and CSO discharges are pathways for PCB entry into Burrard Inlet; therefore, interim benchmarks in water are proposed with the intent to convert these benchmarks into objectives once laboratory detection limits are improved. These interim single sample maximum benchmarks should be used to guide future monitoring of levels of PCBs, PCDDs and PCDFs in discharges, and to determine if they are in ambient waters at levels that could be of concern to marine aquatic life or to human consumers of seafood. A qualitative objective for PCBs, PCDDs and PCDFs is proposed for water (as well as for sediment and tissue) of decreasing concentrations.

Because PCBs, PCDDs and PCDFs have low water solubility, objectives for these substances in water are less practical for ambient monitoring compared to sediment and tissue objectives.

Sediment

Because PCBs are hydrophobic and preferentially bind to organic matter and sediments rather than remaining in the water column, objectives for sediment are generally more relevant than objectives for water (Brewis 2010).

The proposed objective for total PCBs in sediment (3.7×10^{-6} µg/g dry weight) is based on research that recommended guidelines protective of higher trophic level aquatic species, in particular Northern and Southern Resident Killer Whales (Alava et al. 2012). This objective is consistent with the BC Working Water Quality Guidelines (BC ENV 2020) and the Environmental Quality Guidelines recommended by the SRKW Contaminants Technical Working Group (ECCC 2021).

The proposed sediment objective for total PCDDs and PCDFs in sediment (8.5×10^{-7} µg/g dry weight) is based on the threshold effects level (TEL) outlined by CCME (2001a), which has been adopted by BC ENV (2020) as a working sediment quality guideline. This objective is not protective of higher trophic levels such as marine mammals, however. Therefore, the tissue objective should be used in conjunction to the sediment objective to evaluate the potential for adverse effects on aquatic wildlife and humans (CCME 2001a).

Tissue

The proposed objective for total PCBs in tissue (0.00029 µg/g wet weight) is protective of Northern and Southern Resident Killer Whales, and is the most conservative of the tissue benchmarks that include non-dioxin-like PCBs. Attaining this level would also be protective of human consumption of seafood based on the calculation of tissue screening values, as summarized in Table 2 and described in more

detail in Appendix A. A single objective is proposed for simplicity; however, if this screening value is exceeded, further analysis against the tissue screening values as derived from Health Canada TRVs would be necessary for human health risk assessment, and to help in determining safe consumption limits. As with water, an objective for total PCBs in tissue provides a full picture of toxic risk, as it includes both dioxin-like and non-dioxin-like PCBs.

It has been recommended that quantitative analysis of dioxin-like compounds include all chemicals showing similar biological and dioxin-like activity (Srogi 2008). This is consistent with the methodology used by Health Canada (2021) to set TRVs for these compounds. Hence, a single objective (8.0×10^{-8} µg/g wet weight) is proposed for the sum of dioxin-like PCBs, PCDDs and PCDFs for tissue. This proposed objective is based on the most conservative tissue screening value, protective of consumption of finfish and shellfish by toddlers from a subsistence fishing population. Data analysis to determine whether this objective has been attained should be carried out by converting the concentrations of all PCDDs, PCDFs and dioxin-like PCBs in any given mixture to units of 2,3,7,8-TCDD TEQs using the TEFs provided in Table 3 (from van den Berg et al. 2006 and included in Health Canada 2021). The sum of the TEQs is then compared to the objective, which was derived using the TDI for 2,3,7,8-TCDD.

Although data were available for liver, muscle and whole body tissue, and the data assessment indicated that liver samples had the highest concentrations of these contaminants, the tissue objectives are proposed to apply to all tissue types for greatest relevance to consumption by humans and by wildlife. This approach is consistent with the approach taken in the technical reports for other contaminants considered in the update to the Water Quality Objectives for Burrard Inlet.

All

The proposed objectives are frequently well below the measured concentrations of PCBs, PCDDs and PCDFs in Burrard Inlet, and this may be partly due to sources outside of the Burrard Inlet basin, as well as persistence in the marine environment. While attaining these objectives is anticipated to be challenging, it was deemed important to propose them because of their known toxicities and as aspirational goals to protect the values defined for Burrard Inlet, for example protection of aquatic life and protection of seafood consumption by humans (Rao et al. 2019).

A qualitative objective is proposed for a decreasing trend in the concentrations of all PCB, PCDD and PCDF congeners in all media over time.

These objectives may be updated in the near future if the guidelines developed by the subcommittee of the Southern Resident Killer Whales contaminants technical working group are more protective than the proposed benchmarks for water, and the proposed objectives for sediment and tissue.

5. MONITORING RECOMMENDATIONS

Monitoring recommendations help refine existing monitoring programs and inform future assessments to determine whether the objectives are attained. The following are recommendations for future monitoring of PCBs, PCDDs and PCDFs in Burrard Inlet:

- Establish consistent methodologies for water column, sediment, and tissue sampling, including consistent reporting of total PCBs, total PCDDs and PCDFs, and 2,3,7,8-TCDD TEQs.
- Incorporate analysis of dioxin-like PCBs into monitoring programs, along with analyses of other PCBs, PCDDs and PCDFs;
- Carry out sufficient monitoring to determine the locations of PCB, PCDD and PCDF hotspots in Burrard Inlet to focus remediation efforts;

- Sample PCBs, PCDDs and PCDFs in water to update the data set and understand whether they are present in the water column, to update the objectives in water;
- Analyze PCBs, PCDDs and PCDFs in age-dated sediment cores to assess temporal trends and anticipated decline in occurrence.;
- Identify and monitor specific point sources and dispersal by currents to take into account distribution of PCB congeners in Burrard Inlet (Sun 2020). Also monitor the vicinity of areas known to be sources of PCBs, such as near stormwater and combined sewer outfalls and near permitted discharges. Agencies that are monitoring specific sources of PCBs, PCDDs, and PCDFs (e.g. ongoing CSO and treatment plant outfall monitoring and sediment effects surveys by Metro Vancouver) should continue to link this work to remediation of the potential effects of those sources;
- Ensure that material proposed to be dredged from Burrard Inlet is tested for PCBs, PCDDs and PCDFs, and concentrations compared to the updated WQOs for these substances;
- Sample tissue in species that are preferred for human consumption, in particular for species preferred by TWN members, and compare results to those in more frequently monitored species (e.g. blue mussels) to determine if the latter are suitable indicator species.

6. MANAGEMENT OPTIONS

The following initiatives are planned or underway that will help reduce PCB, PCDD and PCDF levels in Burrard Inlet:

- International treaties such as the Stockholm Convention, signed by more than 150 countries, which aims to end all use of PCBs by 2025, and required reductions in emissions of dioxins and dioxin-like substances (UNEP 2019);
- Canadian measures, such as regulation of the storage, handling, transport, and destruction of PCBs, and the planned virtual elimination of PCDDs and PCDFs through federal and provincial legislation such as the *Canadian Environmental Protection Act* (1999) the federal *Toxic Substances Management Policy* (Government of Canada 1995), the CCME *Policy for the Management of Toxic Substances*, and the BC Contaminated Sites Regulation and Hazardous Waste Regulation (BC ENV 2021);
- The *Pulp and Paper Mill Effluent Chlorinated Dioxins and Furans Regulations (CEPA 1999),* which control the release of PCDDS and PCDFs, restricting the use of elemental liquid chlorine in pulp and paper mills, and other improved pollution control standards for pulp and paper plants and wood treatment facilities (*CEPA* 1999). These have resulted in the virtual elimination of PCDDs/PCDFs releases to water from this source (Government of Canada 2006). The CCME Canada-Wide Standards for Dioxins and Furans include emission limits or activity phase-out, implementation timelines, and pollution prevention strategies (CCME 2001b). In 2009, CCME reported that the Canada-wide standards were successfully implemented and achieve the desired outcome of reducing the annual release of dioxins and furans to the atmosphere by 60% (ECCC 2013).

The following management options that have the potential to further reduce PCB, PCDD and PCDF contamination in Burrard Inlet are recommended for consideration, although this is not an exhaustive list of tools and actions:

- Continue implementation of provincial or federal laws and approaches to identify, reduce the use of, or phase out and properly dispose of toxic materials, particularly those that contain chemicals that persist or bioaccumulate in the marine environment (*CEPA* 1999, Toxic

Substances Management Policy, CCME Canada-Wide standards, BC Hazardous Waste Regulation, BC *Pesticide Control Act)* ;

- Clean-up of PCB-contaminated sites, and ensure appropriate disposal of existing PCBs;
- Mitigation of all landfill leachates;
- Education of individuals and organizations about responsible handling of materials that may contain PCBs, PCDDs, or PCDFs, for example:
    - Not burning garbage, plastic, treated wood or painted wood;
    - Recycling and disposing of waste responsibly (Government of Canada 2006);
- Given that PCBs are frequently detected in stormwater, increased implementation of stormwater source controls, including green stormwater infrastructure such as swales, rain gardens, and tree trenches is needed (King County 2020).

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APPENDIX A: CALCULATIONS FOR SCREENING VALUES FOR HUMAN FISH CONSUMPTION

Human health based tissue screening values for PCBs, PCDDs and PCDFs were calculated from the following equation (see Thompson and Stein [2021] for details) and listed in the tables below. Tolerable daily intakes (TDIs) were obtained from Health Canada (2021). For non-dioxin-like PCBs, the TDI was the provisional oral TDI used to derive a toxicological reference value based on a critical health endpoint of immunotoxicity (decreased antibody response). For PCDDs, PCDFs and dioxin-like PCBs, the TDI was the oral TDI for 2,3,7,8-TCDD used to derive a toxicological reference value based on critical health endpoints of developmental toxicity, in particular decreased sperm production, altered sexual behaviour in male offspring, decrease of ventral prostate weight, and anogenital distance in male offspring. The total dioxin-like toxicity of mixtures of PCDDs, PCDFs and dioxin-like PCBs in tissues is calculated by way of a toxic equivalency factor (TEF) approach that quantifies the toxicity of individual congeners relative to 2,3,7,8-TCDD, explained above in section 3.1.

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

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 non-dioxin-like PCBs

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)
Subsistence Fishers	Toddler	94	0.01	16.5	100%	0.2	0.0004
	Adult	220	0.01	76.5	100%	0.2	0.0007
Recreational Fishers	Adult	111	0.01	76.5	100%	0.2	0.0014

Table A2. Tissue screening value calculation for PCDDs, PCDFs and dioxin-like PCBs

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)
Subsistence Fishers	Toddler	94	0.0000023	16.5	100%	0.2	0.0000000008
	Adult	220	0.0000023	76.5	100%	0.2	0.00000016
Recreational Fishers	Adult	111	0.0000023	76.5	100%	0.2	0.00000032