BURRARD INLET WATER QUALITY PROPOSED OBJECTIVES

Water Quality Assessment and Proposed Objectives for Burrard Inlet: Polybrominated Diphenyl Ethers (PBDEs) Technical Report

August 2021

Tsleil-Waututh Nation
səlilwətaɬ

BRITISH COLUMBIA

The purpose of this initiative is to assess the present state of water quality in Burrard Inlet and to update the Burrard Inlet Water Quality Objectives (BIWQOs) to reflect current knowledge and updated future goals for the health of these waters. It draws upon the 1990 provisional Water Quality Objectives, Tsleil-Waututh Nation's Burrard Inlet Action Plan, the work of the Burrard Inlet Environmental Action Program, water quality monitoring data, current science, Indigenous knowledge and stewardship values, and more. For additional information visit: https://www2.gov.bc.ca/gov/content/environment/air-land-water/water/water-quality/water-quality-objectives.

ISBN: 978-0-7726-8006-8

Citation:

Braig, S., Delisle, K., LeNoble, J., Björklund, K., Noël, M., Rao, A.S. and H.C. Thompson. 2021. Water quality assessment and proposed objectives for Burrard Inlet: polybrominated diphenyl ethers (PBDEs) technical report. Tsleil-Waututh Nation and Province of BC.

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

Acknowledgements

The production of this document and the progress of this initiative were a result of work and guidance by the following:

- Tsleil-Waututh Nation (TWN) and the BC Ministry of Environment and Climate Change Strategy (ENV), who co-led a coordination team. The members of the coordination team were Anuradha Rao (contractor to TWN), Diane Sutherland (ENV), Patrick Lilley (Kerr Wood Leidal, consultant to TWN) and Sarah Dal Santo (TWN),
- Multi-agency advisory bodies: Burrard Inlet Water Quality Technical Working Group and Roundtable (representatives of First Nations; local, provincial and federal governments; health authorities; industry; academics and NGOs).
- Staff, specialists and consultants including:
    - Adrienne Hembree, Andrew George, Bridget Doyle, Carleen Thomas, Ernie George, Gabriel George, Graham Nicholas, John Konovsky, Stormy MacKay (TWN) and Allison Hunt (Inlailawatash)
    - Kevin Rieberger and Melany Sanchez (ENV), and Dan Stein (BC MOH)
    - Daniel Brown, Grace O'Connell, Jack Lau, Larissa Low (Kerr Wood Leidal)

We would also like to acknowledge financial support from: BC Ministry of Environment and Climate Change Strategy, Government of Canada, Sitka Foundation and Vancity Credit Union, and in-kind contributions from Roundtable members.

CHAPTER SUMMARY

This chapter presents proposed water quality objectives for several polybrominated diphenyl ethers (PBDEs) in Burrard Inlet. These proposed objectives were developed using up-to-date research on relevant values and potential effects, sources and factors influencing PBDE levels, benchmark screening, and historic and recent monitoring data for Burrard Inlet.

From the 1970s until the early 2000s, PBDEs were used extensively as flame retardant additives in various materials, such as plastics, rubber, electronics, furniture, building materials, and textiles (Abbasi et al., 2019). Global production levels of PBDEs were estimated at 70,000 metric tonnes per year in the late 1990s, 49% of which originated in North America (Grant et al., 2011). In 1999, PBDEs were declared toxic under the *Canadian Environmental Protection Act* (CEPA). Penta- and octa-BDE commercial mixtures were voluntarily phased out in Canada in the early 2000s, followed by regulatory restrictions in both Canada and the USA (Canadian Gazette, 2006; ECCC, 2008). Several PBDEs are also listed on the Stockholm Convention on Persistent Organic Pollutants (POPs), a global treaty that requires its parties to eliminate or reduce the release of listed chemicals into the environment (Abbasi et al., 2019).

Even though some PBDE formulations have been banned or are no longer in use in Canada, they can be transported via atmospheric and oceanic currents from areas outside of Canada. Studies that examined the net flux of PBDEs in the Strait of Georgia indicated that PBDE inputs and movement appear to be from the atmosphere to seawater (Noël et al. 2009), and from seawater into sediments (Dinn et al. 2008). In addition to atmospheric inputs, major sources of PBDEs into Burrard Inlet include effluent from the Lions Gate Wastewater Treatment Plant (WWTP), combined sewer overflows, and PBDEs translocated from outside the Inlet via movement of fish and other biota (Burd et al., 2019).

The analysis for PBDEs in Burrard Inlet was limited to sediment and tissue because sampling for PBDEs in marine water samples was not conducted in any of the key data sources vetted for this study. PBDEs were classified and assessed by homologue groups and specific congeners were further assessed based on detection frequency in sediment and tissue. Homologue groups were prioritized if concentrations exceeded relevant guidelines or human health-based screening values. These prioritized congeners and corresponding homologue groups include: BDE-209 (deca-BDE), BDE-47 (tetra-BDE), BDE-49 (tetra-BDE), BDE-99 (penta-BDE), BDE-207 (nona-BDE), BDE-100 (penta-BDE), and BDE-17 (tri-BDE), BDE-66 (tetra-BDE), BDE-153 (hexa-BDE), BDE-154 (hexa-BDE) and BDE-203 (octa-BDE). Total PBDEs in sediment were also assessed because there is an available BC guideline.

Screening of data against benchmarks indicated that total PBDE concentrations exceeded the BC Working Water Quality Guideline of 1 ng/g (dry weight) in most sediment samples. Sites with the highest PBDE sediment concentrations were generally in the Inner Harbour, Outer Harbour, and Port Moody Arm. PBDEs within each homologue group were summed for comparison to the most conservative tissue benchmarks available. Results for fish tissue data showed that total concentrations of tetra-, penta-, and hexa-BDEs were above the homologue-specific benchmark value for almost all sites sampled. PBDE levels in fish tissues were highest in the Outer and Inner Harbour sites, with some elevated concentrations in the Port Moody area. For mussel tissue, PBDE levels were highest in the Outer and Central Harbours. PBDE levels in mussels were below the homologue-specific benchmark values protective of both fish health and consumers of aquatic biota.

The following have been proposed as water quality objectives for Burrard Inlet. In addition to these numeric objectives, an overall objective is for a decreasing trend in PBDE concentrations in all media.

Proposed Benchmarks and Objectives for PBDEs in Burrard Inlet (all sub-basins)[1]; proposed WQOs are bolded and highlighted

Homologue	Congener(s)	Water[1] (ng/L)	Sediment[2] (ng/g dry weight)	Tissue		
				Fish Tissue (ng/g wet weight)[3]	Wildlife Diet (ng/g wet weight food source)[4]	Screening value for human consumption of finfish and shellfish (ng/g wet weight)[5]
Tri-BDE	Total	**46**		**120**	-	
Tetra-BDE	Total	**24**		88	**44**	
	BDE-47					**4**
Penta-BDE	Total	**0.2**		**1**	3 (mammal) 13 (birds)	70
Hexa-BDE	Total	**120**		420	**4**	
	BDE-153					7
Hepta-BDE	Total	**17**		-	**64**	
Octa-BDE	Total	**17**		-	**63**	105
Nona-BDE	Total	-		-	**78**	
Deca-BDE	Total	-		-	**9**	
	BDE-209					246
Total PBDEs[5]			**1**			

[1] Based on a minimum of 5 surface samples in 30 days collected during the wet season.
[2] Based on a minimum of 1 composite sample composed of at least 3 replicates.
[3] The concentration not expected to cause adverse effects to the fish, based on a whole-body composite sample consisting of at least 5 individual fish. See Rao et al. (in prep.) for additional guidance.
[4] The concentration in whole food not expected to cause adverse effects to wildlife consumers, based on a whole-body composite sample consisting of at least 5 individual fish or 25 bivalves. See Rao et al. (in prep.) for additional guidance.
[5] The concentration not expected to cause adverse effects to human consumers, based on a composite sample consisting of at least 5 individual fish or 25 bivalves. See Rao et al. (in prep.) for additional guidance.

[1] From BC Working Water Quality Guidelines (ENV, 2020; adopted from ECCC 2013b and Alava et al. 2016) and calculated screening values from Health Canada (2010c), calculated by Thompson and Stein (2021).

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ACRONYMS

BC	British Columbia
BDE	Brominated Diphenyl Ethers
CCME	Canadian Council of Ministers of the Environment
CEPA	Canadian Environmental Protection Agency
CFIA	Canadian Food Inspection Agency
CSO	Combined Sewer Overflow
CSSP	Canadian Shellfish Sanitation Program
DDT	Dichlorodiphenyltrichloroethane
DL	Detection limit
DW	Dry weight
ECCC	Environment and Climate Change Canada
ENV	Ministry of Environment and Climate Change Strategy
FEQG	Federal Environmental Quality Guideline
LW	Lipid weight
MLD	Million litres per day
PBDEs	Polybrominated Diphenyl Ethers
PCBs	Polychlorinated Biphenyls
PCDEs	Polychlorinated Diphenyl Ethers
PCNs	Polychlorinated Naphthalenes
POPs	Persistent Organic Pollutants
TOC	Total Organic Carbon
TWN	Tsleil-Waututh Nation
WW	Wet weight
WWTP	Wastewater Treatment Plant

1. INTRODUCTION

This chapter presents proposed water quality objectives for several polybrominated diphenyl ethers (PBDEs) in Burrard Inlet. Tsleil-Waututh Nation identifies PBDEs as primary pollutants of concern in their Burrard Inlet Action Plan (TWN, 2017). The chapter includes relevant background information and an assessment of the current status of PBDEs in sediment and biota in Burrard Inlet, and a rationale for PBDE objective development. Recommendations for future monitoring as well as management options to help achieve these objectives are also included.

2. BACKGROUND

PBDEs are a class of anthropogenic organobromine compounds comprised of 209 possible congeners, each having between one and ten bromine atoms (ECCC, 2013b). PBDEs include mono-brominated diphenyl ethers (mono-BDEs), di-BDEs, tri-BDEs, tetra-BDEs, penta-BDEs, hexa-BDEs, hepta-BDEs, octa-BDEs, nona-BDEs and deca-BDEs. The lower brominated compounds (tetra- to hexa-BDEs), as well as the resulting environmental decomposition products, are usually more volatile, water soluble, and bioaccumulative than the higher brominated compounds (ECCC, 2013b). Furthermore, debromination of highly brominated PBDE congeners (e.g., BDE-209) in the environment can potentially lead to the formation of lighter congeners that are more toxic, more mobile and readily accumulate in aquatic food webs (Stapleton et al., 2004; Stapleton et al., 2006; Thomas et al., 2005; Ross et al., 2013). Studies have demonstrated that BDE-209 debrominates into lighter congeners in fish (Stapleton et al., 2004b, 2006a) and seals (Thomas et al., 2005).

PBDEs are persistent, bioaccumulative, and toxic to both humans and the environment. They can undergo long-range global atmospheric transport and biomagnify in the food chain, causing potential health effects in apex predators (ECCC, 2013b).

From the 1970s until the early 2000s, PBDEs were used extensively as flame retardant additives in various materials, such as plastics, rubber, electronics, furniture, building materials, and textiles (Abbasi et al., 2019). Global production levels of PBDEs were estimated at 70,000 metric tonnes per year in the late 1990s, 49% of which originated in North America (Grant et al., 2011). The commercial penta-BDE congener was used in 90% of residential furniture, car upholstery, sound insulation, and wood imitation products, while deca-BDE was used in cabinets, pipes and fittings, as well as automotive parts and electronic components (ECCC, 2013b). In 1999, PBDEs were declared toxic under the *Canadian Environmental Protection Act* (CEPA), with penta- and octa-BDE commercial mixtures being voluntarily phased out in Canada in the early 2000s, followed by regulatory restrictions in both Canada and the USA (Canadian Gazette, 2006; Environment Canada, 2008). Several PBDEs are also listed on the Stockholm Convention on Persistent Organic Pollutants (POPs), a global treaty which requires its parties to eliminate or reduce the release of listed chemicals into the environment (Abbasi et al., 2019).

PBDE flame retardants are reaching a steady state or are declining in biota over time and in the vicinity of the coastal and marine environment of BC, as observed in harbour seal pups (Ross et al., 2013; Alava, 2019). PBDE concentrations increased exponentially between 1984 and 2003, but appear to be in decline following the implementation of regulations in the early 2000s (Noël and Ross 2018). PBDEs are among the POPs of greatest concern to high trophic level marine animals in BC, such as resident killer whales, due to their increasing biomagnification in food webs (Alava, 2019). The Strait of Georgia has been identified as a regional sink for POPs such as PBDEs (Johannessen et al. 2008; Grant et al. 2011). Food web bioaccumulation models suggest that PBDEs in resident killer whales exceed toxic thresholds throughout their coastal BC habitats (Alava et al. 2012a, 2016).

2.1 Values and Potential Effects

The most sensitive values guiding water quality objectives for PBDEs are aquatic life, wildlife and human consumption of shellfish and finfish. The goal of the WQOs is to maintain PBDEs levels below values which would be toxic to aquatic life, wildlife and humans who consume seafood at rates relevant to coastal Indigenous peoples.

The potential toxic effects of PBDEs on aquatic life are diverse and specific to the life history and trophic level of the receptor organism as is discussed in the following sections.

2.1.1 ***Ecotoxicology/Effects on Marine Biota***

2.1.1.1 Algae, Bivalves & Fish

In microalgae, the adverse effects of PBDE exposure include inhibition of growth, damage of cellular membranes and/or photosynthetic systems, as well as oxidative stress (Zhao et al., 2019). Damage to photosynthetic functions can also cause a shortage of cellular energy, resulting in further blockage of cell division.

Although there are limited studies on the effects of PBDE exposure in bivalves, Espinoza et al. (2019) indicated that when blue mussels (*Mytilus galloprovincialis)* were fed microalgae with increasing concentrations of PBDEs, immune function was affected, and PBDEs acted as a stressor that could compromise the general welfare of the animal.

In fish, studies have demonstrated that exposure to PBDEs can cause alteration of behaviour, growth, reproductive, hepatic and renal functions, as well as immune and endocrine impairment (Horri et al., 2018). The thyroid is the main target organ for PBDE-related toxicity in fish, but PBDEs may also interfere with steroidogenesis gene expression, sex hormone levels, and the gonad gland (Yu et al., 2015). Accumulation of PBDEs varies between fish species. For example, wild Chinook salmon (*Oncorhynchus tshawytscha*) typically accumulate higher percentages of BDE-47, 49 and 99, while wild carp (*Cyprinus carpio*) typically accumulate higher proportions of BDE-47 and 100 (Roberts et al., 2011).

2.1.1.2 Marine Mammals

PBDEs can biomagnify in food webs causing potential health effects in apex predators, such as marine mammals (Kelly et al., 2007; Kelly e al., 2008; Alava et al., 2016). Marine mammals are particularly sensitive to the effects of PBDEs that accumulate in their bodies over their lifespan. PBDEs are released into the milk of lactating females, or are passed via transplacental transfer to offspring, which can affect development in critical early-life stages (Frouin et al., 2010 and Desforges et al., 2011). In vitro cytotoxic studies of PBDE exposure in mammalian cell lines show that PBDEs can induce programmed cell death and cell cycle arrest, leading to the inhibition of cell division (Zhao et al., 2019). Potential PBDE-induced health effects in marine mammal species are discussed more specifically in the following sections.

Both seals and whales use Burrard Inlet, including species at risk (e.g., killer whales). The need for development of PBDE guidelines protective of these top predators (Alava et al., 2016) is also addressed further in this report (Section 3.4).

2.1.1.3 Seals

Hall et al. (2003) investigated the effects of total PBDE exposure in 54 post weaned grey seal (*Halichoerus grypus*) pups and 55 first year juveniles from the Farn Islands, United Kingdom. The results of the field study indicated a link between thyroid hormones and exposure to PBDEs during the first year of life, which suggests that these pollutants are potential endocrine disruptors in grey seals (Hall et al., 2003). Frouin et al. (2010) collected blood samples from adult seals and pups in the St. Lawrence estuary, Canada, and examined in vitro effects of BDE-47, 99, and 153 on cultured blood cells. Results indicated the potential for PBDEs to disrupt the immune system in seals, although the mechanisms of action were unclear (Frouin et al., 2010)

Despite evidence indicating that PBDEs can interfere with thyroid hormones and the immune system in seals (Hall et al. 2003; Hall and Thomas, 2007; Frouin et al. 2010), the effects of PBDEs on the health of marine mammals are poorly understood and remain largely unknown at this time (Alava et al., 2016).

Interactions between different pollutants in the marine environment can also affect the toxicity of PBDEs in marine mammals. In a study of free-ranging harbour seals, elevated serum T3 (thyroid hormone) levels were significantly related to higher blubber concentrations of a contaminant mixture that included PBDEs, PCBs, and dichlorodiphenyltrichloroethane (DDT), while another study reported an association between blubber concentrations of PBDEs and PCBs and thymic atrophy and splenic depletion in stranded and by-caught harbour porpoises (Shaw et al., 2009). The results of these two studies suggest that PCB-PBDE interactions may have possible synergistic effects in marine mammals (Shaw et al., 2009).

2.1.1.4 Killer Whales

In the Recovery Strategy for the transient, northern, and southern resident killer whale (*Orcinus orca*) populations in Canada, PBDEs are listed as a priority contaminant of concern (Department of Fisheries and Oceans, 2007).

Killer whales can be exposed to PBDEs in Burrard Inlet from the consumption of contaminated prey items (salmon and/or seals). Salmon and seals may be contaminated with PBDEs that are either accumulated within or outside of Burrard Inlet. Since killer whales are long-lived top predators, they can accumulate high levels of PBDEs in their tissues, which can potentially result in endocrine disruption and affect neurological and reproductive development (Ross, 2006; Alava et al., 2016). Even the lowest detected PBDE concentration in blubber from a southern resident killer whale (2500 ng/g lipid weight) is at a level associated with endocrine disruption in grey seals (Krahn et al., 2007).

2.1.2 *Toxicity/Potential Effects on Humans*

PBDE exposure in humans can occur through air, water, soil, sediments, indoor dust, and food, with dietary and dust exposure being considered the most important sources (National Collaborating Center for Environmental Health, 2013). Watanabe (2003) reported that the greatest source of PBDEs in humans was from fish consumption, based on studies in the Baltic Sea and Japan.

The potential for bioaccumulation, including increased potential following debromination of PBDEs (Stapleton et al., 2004; Stapleton et al., 2006; Thomas et al., 2005; Ross et al., 2013), is of concern for human consumers of finfish and shellfish, particularly for subsistence consumers such as coastal Indigenous Peoples (TWN, 2017).

The effects of different PBDEs congeners in humans are not well known, but the potential for endocrine disruption is particularly concerning, since effects may occur at extremely low doses (National Collaborating Center for Environmental Health, 2013). Critical health endpoints determined from studies on rats and mice include neurobehavioral and neurodevelopmental effects (US EPA, 2017).

2.2 Potential Sources of PBDE Pollution

PBDEs are released to air, water and soil during their manufacture and use. In the air they can be present as particles, settling as dust onto land and into water. PBDEs are not easily water soluble; they partition to particulate matter and tend to settle. As a result, high PBDE levels do not tend to be found in water samples (Pohl et al. 2017).

Even though some PBDE formulations have been banned or are no longer in use in Canada, PBDEs can be transported via the atmosphere from areas outside of Canada that continue to use them. This global transport of PBDEs creates continuous inputs into the local Burrard Inlet marine environment (Noël et al., 2009; Grant et al., 2011; Alava et al., 2016). PBDEs can also be present in manufactured items made from recycled plastics that originally contained PBDEs (e.g., pens, staplers, and wire spools) (Government of Canada, 2018).

Studies that examined the net flux of PBDEs in the Strait of Georgia indicated that PBDE inputs and movement appear to be from the atmosphere to seawater (Noël et al. 2009), and from seawater into sediments (Dinn et al. 2008 and Alava et al., 2016). This supports the notion that air and water may be delivering a major portion of PBDEs to the aquatic food web (Alava et al., 2016).

Densely populated harbours and inlets such as Burrard Inlet have some of the highest levels of PBDEs, in comparison to more remote areas in BC. In addition to atmospheric inputs of PBDEs into Burrard Inlet, major sources include effluent from the Lions Gate Wastewater Treatment Plant (WWTP), combined sewer overflows, and translocation of PBDEs (Burd et al., 2019). These sources are discussed in the following sections.

2.2.1 *Atmospheric Deposition*

Atmospheric deposition of PBDEs to Burrard Inlet originates from both local and global sources. Locally, PBDE-associated dust particles from urban areas surrounding Burrard Inlet may be deposited into the marine environment (Anderson et al., 2006). Globally, prevailing winds from the west and southwest spread air across the Pacific Ocean. The rapid movement of these westerly air masses over the Pacific Ocean delivers pollutants from Asia to North America in a matter of days (Noël et al., 2009).

Noël et al. (2009) studied the difference between local and background sources of 205 PBDE congeners in coastal British Columbia. Air samples were collected from a remote site on western Vancouver Island and from a near-urban site in the Strait of Georgia over the course of a year. The study concluded that deposition of PBDEs at the remote site amounted to 42% (10.4 mg/ha/year) of that at the near-urban site (Noël et. al., 2009). In addition, over 12,000 ten-day back trajectories[2] indicated that 98% of the air masses in coastal BC over the sampling period originated from Asia, compared to only 2% from North America, suggesting that Asian sources may account for a significant percentage of PBDEs in coastal British Columbia air (Noël et al., 2009).

2.2.2 *Sewage Outfalls*

WWTPs can potentially discharge effluent that has been contaminated with PBDEs from old consumer electronics, furniture, treated fabrics, and other products that may still contain PBDEs. Once PBDEs are in the sewage waters, due to their high hydrophobicity and low vapour pressure, they can strongly adsorb to suspended organic material within the sewage sludge. However, PBDE residues are also possible within effluent waters (Lee et al., 2018), and over large volumes can represent a significant contribution to the Inlet. High levels of PBDEs have been found within WWTP effluents and sludge in other jurisdictions (Lee et al., 2018). A study of PBDEs in sediments and benthic invertebrates at Vancouver's Iona Island sewage treatment plant's outfall found that PBDE accumulation in benthic invertebrates was controlled in part by total organic carbon (TOC) content in sediment. It was therefore

[2] Back trajectory analyses use interpolated measured/modeled meteorological fields to estimate the most likely central path over geographical areas that provided air to a receptor at a given time. The method essentially follows a parcel of air backward in hourly steps for a specified length of time (Colorado State University, 2019)

concluded that TOC removal during sewage treatment could lead to higher benthic bioaccumulation of PBDEs near the outfall, emphasizing a need for source control and PBDE-specific treatment prior to discharge (Dinn et al., 2012). Biosolids, which are produced from treated sewage solids, are regulated for land application under the *BC Organic Matter Recycling Regulation* and used as an ingredient in landscaping soil applied in parks, habitat restoration projects and landscaping within Metro Vancouver (Metro Vancouver 2020a).

The Lions Gate WWTP, located near the Lions Gate Bridge, east of the Capilano River is a potential source of PBDE contamination in Burrard Inlet. The WWTP receives residential and commercial wastewater from North Vancouver, treating a total of 30,419 million litres (ML) in 2017, with an average daily flow of 83 million litres per day (MLD) for 2016 and 2017 (Metro Vancouver, 2017). It discharges primary treated effluent into the turbulent First Narrows area of Burrard Inlet through an outfall and diffuser located just west of the Lions Gate Bridge in West Vancouver. The outfall is about 184 m offshore, and the average diffuser depth is approximately 20 metres. Effluent dispersal is influenced by tides and currents and can disperse within Burrard Inlet before entering the Strait of Georgia (Metro Vancouver, 2017).

Currently, the Lions Gate WWTP is using a primary treatment system, where wastewater is passed through several tanks and filters that separate the water from solids and organic matter, producing a sludge. Biosolids produced from the treated sludge are often used by Metro Vancouver to create a fertilizer called Nutrifor™ (Metro Vancouver, 2019). Within the Burrard Inlet watershed, these biosolids are used as an ingredient in landscaping soil, making up approximately 10% of the dry weight. Metro Vancouver biosolids are also mixed into a soil product being used in the Fraser Valley to reclaim land disturbed by gravel extraction (Metro Vancouver, 2020a). A portion of PBDEs contained within these biosolids could be transported via surface water or groundwater runoff into Burrard Inlet; however, organic matter can slow the transport of contaminants with high K_{ow} such as PBDEs and prevent them from partitioning into infiltrating water (Gorgy et al., 2013). In a leaching column test in Kamloops, most PBDEs in biosolids-amended soil were retained in the soil after application to agricultural areas, with 1% found in the final leachate (Gorgy et al., 2011).

Upgrades to tertiary treatment are underway at the Lions Gate WWTP, with completion scheduled for 2024. It is anticipated that upgrades will reduce PBDEs in the effluent but increase the amount of biosolids (Kim et al., 2013).

2.2.3 *Stormwater and Combined Sewer Overflows*

Combined sewer overflows (CSOs) occur during wet weather events when there is insufficient combined sewer capacity to handle additional stormwater inflows. As a result, excess combined sewage and stormwater flow directly into Burrard Inlet, rather than being treated at a WWTP (Metro Vancouver, 2017). An assessment of potential localized sources of PBDEs around the Niagara River, Ontario suggested that CSOs could be a source of PBDE contamination into the river (Samara et al., 2006). Preliminary analyses of PBDEs in CSO discharges by Metro Vancouver in 2019 indicate CSOs around Burrard Inlet could also be contributing to PBDEs in the marine environment (Metro Vancouver, 2020b).

2.3 Factors Influencing PBDE Levels in Burrard Inlet

The physicochemical properties of PBDEs and environmental processes acting on these pollutants determine the fate of PBDEs in the marine environment. PBDE concentrations in sediment are strongly influenced by the location of sources (Johannessen et al., 2008). PBDEs will exist in both vapour and particulate phases in the atmosphere, where the particulate phase will be removed via wet and dry deposition (Pohl et al., 2017). Once PBDEs are in marine waters, they will adsorb strongly to any suspended sediments.

Sediments function as a sink for high molecular weight PBDEs such as BDE-209, which bind preferentially to particles and may not be detected in water or biota samples (Johannessen et al., 2008; Ross et al., 2009).

The commercial octa- and deca-BDEs can absorb light up to 325 nm, indicating that these compounds may be susceptible to photodegradation at environmental wavelengths. In contrast, di- and tetra-BDEs were seen to absorb minimal light wavelengths of more than 300 nm, suggesting that the lower brominated PBDEs (e.g., penta-BDE commercial mixtures) are less susceptible to photolysis than octa-BDE and deca-BDE commercial mixtures (Pohl et al., 2017). Lower brominated PBDEs will also be more volatile, water soluble, and bioaccumulative than higher brominated compounds, which are heavier and disperse less in the environment (Watanabe, 2003).

Deca-BDE, a higher brominated congener, makes up approximately 80% of the total PBDEs found in the marine sediment of southern British Columbia (Ross et al., 2009; Burd et al., 2019). Other PBDE congeners seen in marine sediments are BDE-207, BDE-47, BDE-99, BDE-49, BDE-100, and BDE-17 (Grant et al., 2011 and Morales-Caselles et al., 2017). The reservoir of PBDEs in marine sediments can potentially pose a risk to all trophic levels, particularly since any de-bromination products are also toxic and persistent in the food chain (Burd et al., 2019).

2.4 1990 Provisional Water Quality Objectives for PBDEs

Provisional Water Quality Objectives were not developed for PBDEs in 1990.

3. WATER QUALITY ASSESSMENT

In this report, PBDEs were classified and assessed by homologue groups and specific congeners were further assessed based on detection frequency in sediment and tissue. Homologue groups were prioritized if concentrations have exceeded relevant benchmarks or human health-based screening values. These prioritized congeners and corresponding homologue groups include: BDE-17 (tri-BDE), BDE-47 (tetra-BDE), BDE-49 (tetra-BDE), BDE-66 (tetra-BDE), BDE-99 (penta-BDE), BDE-100 (penta-BDE), BDE-153 (hexa-BDE), BDE-154 (hexa-BDE), BDE-203 (octa-BDE), BDE-207 (nona-BDE), and BDE-209 (deca-BDE). Total PBDEs in sediment were also assessed because there is an available BC guideline. With the exception of BDE-17, these priority PBDE congeners are also part of the Environment Canada and Health Canada final Screening Assessment Reports (SARs) for PBDEs, that indicate that these congeners have or may have an immediate or long-term harmful effect on the environment or its biological diversity, and meet the criteria of the *Canadian Environmental Protection Act* (CEPA, 1999).

3.1 Benchmarks Used in this Assessment

Benchmarks were used to screen available data for potential effects and to inform the derivation of proposed objectives for PBDEs levels in Burrard Inlet. Based on the available literature and benchmarks, human health, wildlife and aquatic life are the values most sensitive to PBDEs.

Canadian guidelines for the protection of these values were used as screening benchmarks, where available. Potential sources of screening benchmarks were prioritized as follows:

1. The BC Ministry of Environment and Climate Change Strategy (ENV) has adopted the Environment and Climate Change Canada (ECCC) Federal Environmental Quality Guidelines

(FEQGs) for BDE homologue groups in water, fish tissue, and wildlife diet as Working Water Quality Guidelines (WWQG, ENV, 2020; ECCC, 2013b).

2. For sediment, ENV has adopted a benchmark recommended by Alava et al. (2016), intended to be more protective of higher trophic levels, as a BC Working Water Quality Guideline (ENV, 2020). If no benchmarks were available from the above sources, then guidelines or benchmarks available from other sources or jurisdictions were used.

The screening benchmarks chosen for the data assessment in this report are summarized in Table 1.

Table 1. Benchmarks for PBDEs Used in this Assessment (from BC Working Water Quality Guidelines (ENV, 2020; adopted from ECCC, 2020; ECCC, 2013b and Alava et al., 2016) and calculated screening values from Health Canada (2010c; Thompson and Stein, 2021)

Homologue	Priority Congeners	Congener	Water (ng/L)	Sediment (ng/g dry weight)	Fish Tissue (ng/g wet weight)[1]	Wildlife Diet[2] (ng/g wet weight food source)	Screening value for human consumption of finfish and shellfish (ng/g wet weight)[3]
Tri-BDE	BDE-17	Total	46		120	-	
Tetra-BDE	BDE-47, -49, -66	Total	24		88	44	
Tetra-BDE		BDE-47					4
Penta-BDE	-	Total	0.2		1	3 (mammal) 13 (birds)	70
Penta-BDE	BDE-99	BDE-99	4		1	3	
Penta-BDE	BDE-100	BDE-100	0.2		1	-	
Hexa-BDE	BDE-153, BDE-154	Total	120		420	4	
Hexa-BDE		BDE-153					7
Hepta-BDE	-	Total	17		-	64	
Octa-BDE	BDE-203	Total	17		-	63	105
Nona-BDE	BDE-207	Total	-		-	78	
Deca-BDE	BDE-209	Total	-		-	9	
Deca-BDE		BDE-209					246
Total PBDEs				1			

[1] The concentration of substance in whole body fish tissue (wet weight) which is not expected to cause adverse effects to the fish.

[2] The concentration of substance in whole food (wet weight) which is not expected to cause adverse effects to the wildlife consumers.

[3] Calculated screening value for which PBDE concentrations in tissue can be compared and assessed for potential risks to human health. This is a single benchmark for all tissue types (e.g., fish muscle, bivalves, crustaceans) as data are not available to resolve to the level of objectives for different tissue types. See Appendix A for calculation details.

Federal fish tissue guidelines are not based on fish tissue burdens, but estimated from PBDE concentrations detected in water, together with the degree to which fish are known to accumulate PBDEs from water (ECCC, 2013b and Morales-Caselles et al., 2017).

In the absence of relevant guidelines for human consumption of fish and shellfish tissue, a risk-based approach was used to calculate human health-based tissue screening values for fish and shellfish tissue (Thompson and Stein, 2021). The approach considers: the contaminant *receptors* (people who are exposed to the contaminant, in this case subsistence/Indigenous, recreational, and general BC populations, with screening values calculated for the most sensitive life stage within each population), *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 toxicological reference values (TRVs) prescribed by Health Canada (2010a) or other international agencies (i.e., United States Environmental Protection Agency and the World Health Organization).

Tissue screening values are defined as conservative threshold values against which contaminant concentrations in fish tissue can be compared and assessed for potential risks to human health (Thompson and Stein, 2021). Fish and shellfish tissue 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 and represent a suggested safe level of a 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 (Thompson and Stein, 2021).

Tissue screening values were calculated using equations from Health Canada (2012) as described in Thompson and Stein (2021) (see Appendix A). An allocation factor of 0.2 was used in the calculation to reflect the fraction of PBDEs assumed to come from country foods (in this case, wild seafood). The screening value used as a benchmark for tissue is the most conservative, as calculated for the most sensitive receptor, i.e., a toddler from a subsistence fisher or Indigenous population.

3.2 Data Sources

Data on PBDE levels in Burrard Inlet were gathered from several recent and historic studies and monitoring programs between 2003 and 2016 and are listed in Table 2. Data was taken from Grant et al. (2011), Metro Vancouver's Burrard Inlet Ambient Monitoring Program, Morales-Caselles (2017), Noël and Ross (2017), and Ocean Wise Conservation Association's *PollutionTracker* program. Although other research has been conducted on PBDEs in the Strait of Georgia (e.g. Alava et. al., 2012), this report focuses on datasets specific to Burrard Inlet. Other studies were referenced to describe the health effects of PBDEs, but the derivation of benchmarks from primary research papers was beyond the scope of this report.

Table 2. Studies and Monitoring Programs for Sediment and Tissue Data Used for this Assessment

Source	Study/Monitoring Program	Year(s)	No. of Obs.	No. of Sites	Sampling Frequency	Parameters Sampled
Ross et al. (2013) & Nöel and Ross (2017)	Declining concentrations of persistent PCBs, PBDEs, PCDEs, and PCNs in harbour seals (*Phoca vitulina*) from the Salish Sea & Biological effects of emerging pollutants in marine food web: harbour seals as coastal sentinels	2003 & 2015	Not indicated	4 sites in the Salish Sea, 2 sites in Washington State, USA and 14 harbour seal pups live captured from Vancouver Harbour, Burrard Inlet.	2003 and 2015	PBDEs plus PCBs (polychlorinated biphenyls), PCDEs (polychlorinated diphenyl ethers), PCNs (polychlorinated naphthalenes)/PBDEs
Grant et al. (2011)	Georgia Strait sediment sampling	2003 to 2007	Not indicated	2 sediment	2003 and 2007	PBDEs
Metro Vancouver's Burrard Inlet Ambient Monitoring Program	Ambient sediment program	2008 to 2015	4251 sediment 1282 tissue	7 sediment 7 tissue	2008, 2009, 2013 and 2015	PBDEs
Morales-Caselles (2017)	Georgia Strait sediment sampling	2011	Not indicated	6 sediment	2011	PBDEs
Ocean Wise	*PollutionTracker*	2015 to 2016	532 sediment 402 tissue	15 sediment 9 tissue	2015 and 2016	PBDEs

3.2.1 *Ross et al. (2013) and Nöel and Ross (2017)*

Ross et al. (2013) studied congener-specific PCBs, PBDEs, polychlorinated diphenylethers (PCDEs), and polychlorinated naphthalenes (PCNs) in blubber biopsies from free-ranging harbour seal pups inhabiting four sites in the Salish Sea[3] in 2003. Blubber biopsies were collected from live-captured harbour seal pups from two locations in British Columbia (Hornby Island and the Fraser River estuary in Vancouver) and from two locations in Washington State, USA (Smith Island and Gertrude Island) (Figure 1). The harbour seals that were captured were all healthy, nearly weaned, free-ranging individuals between the ages of three to five weeks; therefore, the body burdens measured reflected the contaminants acquired

[3] The Salish Sea includes the southwestern portion of British Columbia and the northwestern portion of the state of Washington.

via transplacental transfer and from their mothers' milk (Ross et al., 2013). Results from this study are discussed further in Section 3.3.2.

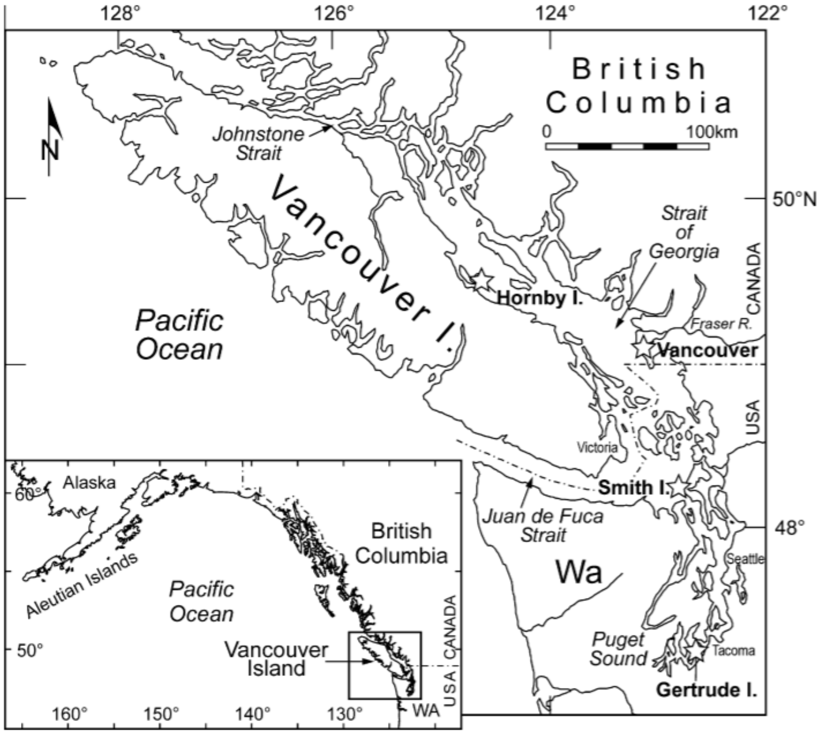

Figure 1. Harbour seal live-capture sites: Hornby Island and Vancouver, British Columbia, and Smith Island (Juan de Fuca Strait, Washington State, USA) and Gertrude Island (Puget Sound, Washington State, USA) (Ross et al., 2013).

In the Nöel and Ross (2017) study, a total of six four-week-old harbour seal pups were live captured from Vancouver Harbour in Burrard Inlet. A single skin/blubber biopsy, fur, and blood were collected from each pup (Nöel and Ross, 2017). Samples were analyzed for several emerging pollutants, including PBDEs. Results are discussed in Section 3.3.2.

3.2.2 ***Grant et al. (2011)***

Grant et al. (2011) collected sediment samples from 41 sites in the Strait of Georgia between 2003 and 2007, with two sites located in Burrard Inlet (indicated by the red circle in Figure 2). Samples were analyzed for PBDEs and results are discussed in Section 3.2. The Burrard Inlet sites included Port Moody Arm (Site 2) and a site in the Central Harbour (Site 3).

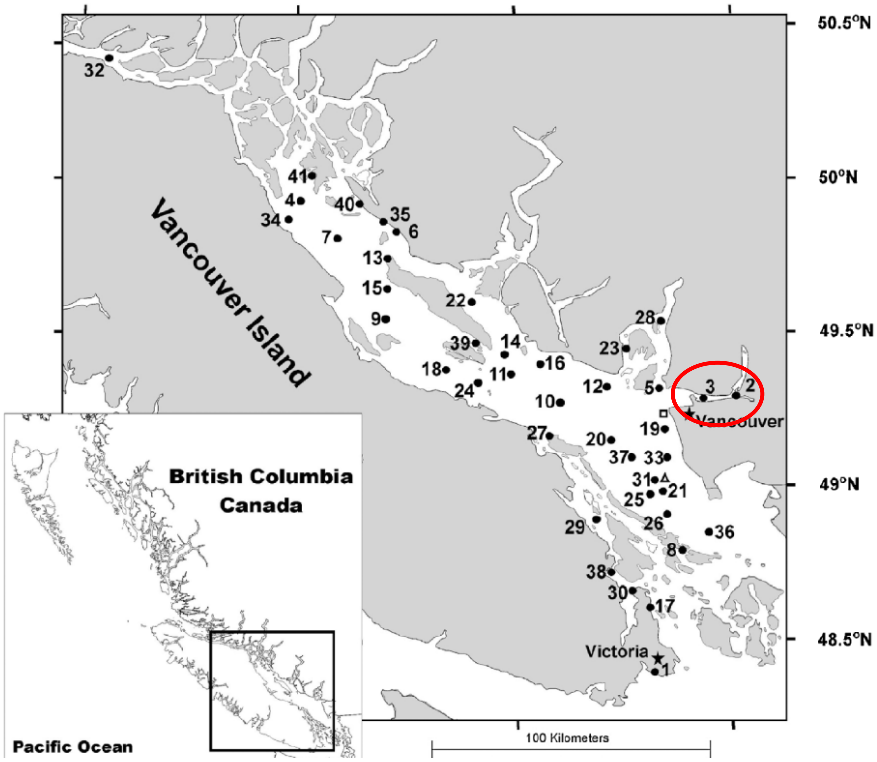

Figure 2. Surficial sediment sampling locations (Grant et al, 2011; Burrard Inlet sites indicated by red circled area)

3.2.3 *Metro Vancouver*

Metro Vancouver conducts an ongoing ambient monitoring program to sample the immediate environment surrounding, but not directly affected by, any discharge from the Lions Gate WWTP into Burrard Inlet. Seven sites were sampled and analyzed for PBDEs in English sole whole body, muscle, and liver in 2007, while only muscle and whole body were sampled in 2012. Sediment was sampled in 2008, 2011, 2013, and 2015 and analyzed for PBDEs. Sampling results are discussed in Section 3.3.

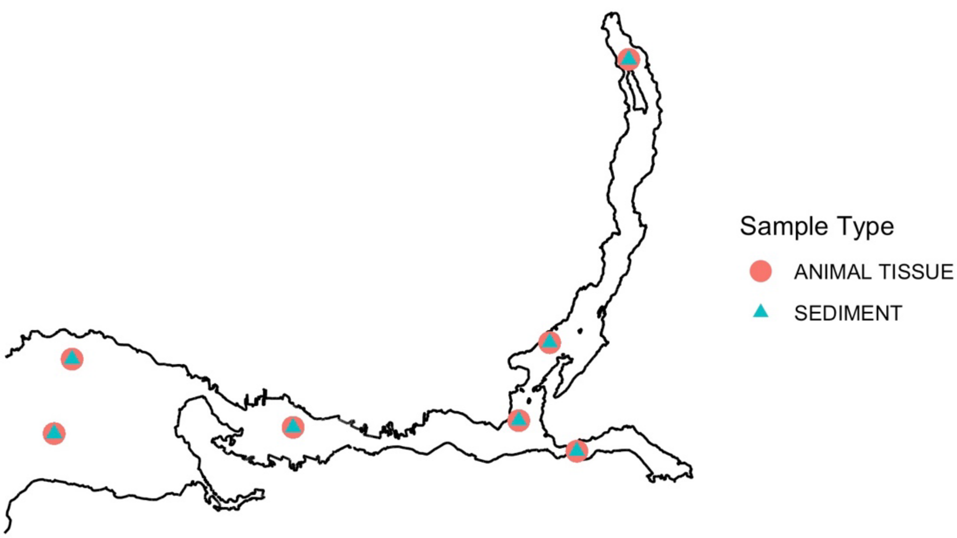

Figure 3. Metro Vancouver sampling stations in Burrard Inlet (2007 to present)

3.2.4 ***Morales-Caselles et al. (2017)***

Twelve surficial sediment samples were collected from three near urban areas in the Salish Sea, British Columbia, with six sites in Burrard Inlet (S1 to S6). These six sites are indicated in Figure 3 below (circled in red). The sites included English Bay (S1), Coal Harbour (S2), Neptune Bank (S3), Indian Arm (S4), Moody Arm (S5), and east Terminal (S6). Results from the analysis of PBDEs are discussed in Section 3.3.

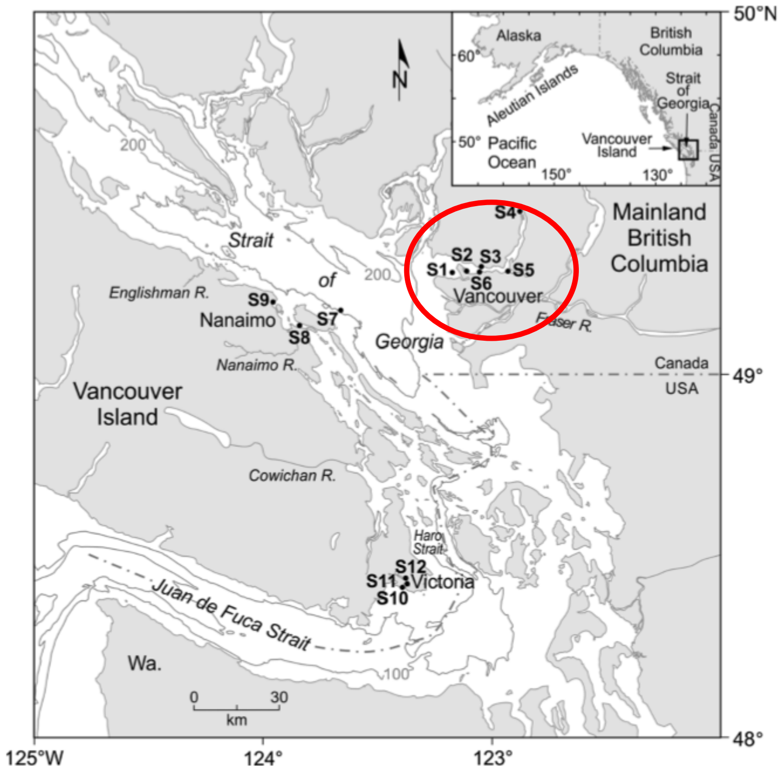

Figure 4. Surficial sediment samples collected. Red circled area indicates Burrard Inlet sites S1 to S6 (adapted from Morales-Caselles et al., 2017)

3.2.5 ***PollutionTracker***

As part of the Phase 1 *PollutionTracker* program*,* analysis of PBDEs in Burrard Inlet sediment and mussels was conducted in partnership with the Vancouver Fraser Port Authority, Metro Vancouver, and the Tsleil-Waututh Nation. Fifteen surface sediment and nine mussel samples (*Mytilus sp.)* were collected from the Outer, Inner and Central Harbours, Port Moody Arm, and two sites in Indian Arm in 2015 and 2016 (Figure 4). Results are discussed in Section 3.3.

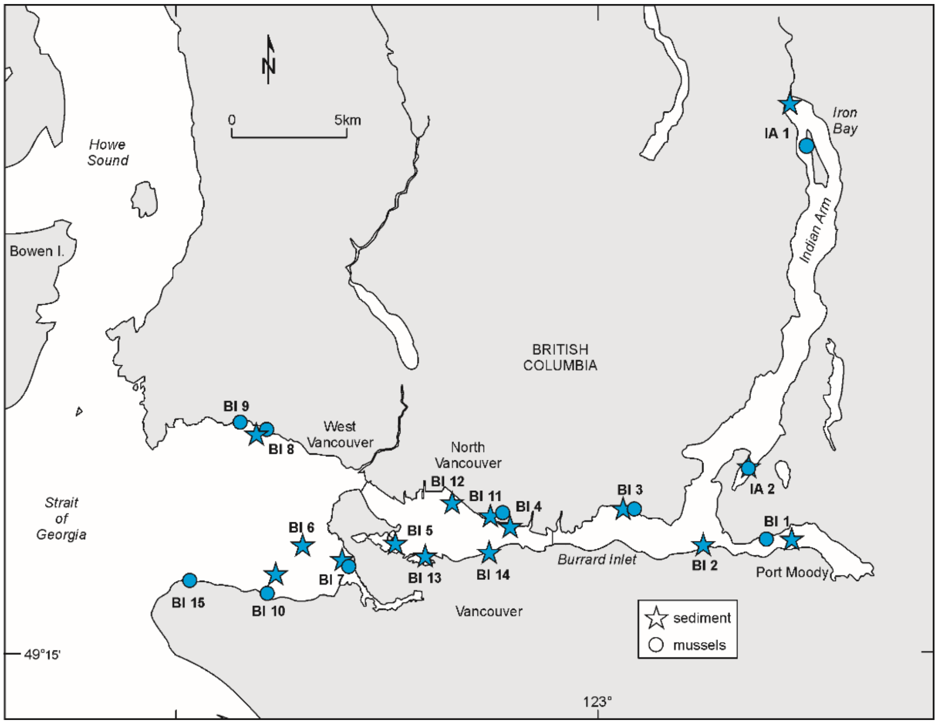

Figure 5. PollutionTracker sampling stations in Burrard Inlet (Ocean Wise, 2019)

3.2.6 ***Quality Assurance/Quality Control Procedures***

The PBDEs were analyzed according to US EPA method 1614A, Brominated Diphenyl Ethers in Water, Soil, Sediment, and Tissue by HRGC/HRMS (US EPA, 2010) in the studies by Morales-Caselles et al. (2017), Grant et al. (2011), Ross et al. (2013), and in Metro Vancouver's Burrard Inlet Ambient Monitoring Program for sediment and tissue (2015, 2016). The recommended procedure for blank correction is that "a result is significantly above the blank level, and the level in the blank may be subtracted, if the result is greater than two standard deviations above the average of results of analyses of 10 or more blanks for a sample medium" (US EPA, 2010: 52). Further, the detection limit is determined by the procedure in Title 40 of the U.S. Code of Federal Regulations (US EPA, 2012: appendix B), as follows. The detection limit can be estimated using one of the following: (a) The concentration value that corresponds to an instrument signal/noise in the range of 2.5 to 5; (b) The concentration equivalent of three times the standard deviation of replicate instrumental measurements of the analyte in reagent water; (c) That region of the standard curve where there is a significant change in sensitivity, i.e., a break in the slope of the standard curve; or (d) Instrumental limitations. The referenced studies did not report which approach was used to determine the method detection limit.

Morales-Caselles et al. (2017) reported the total concentration of PBDEs (66 congeners were analyzed) in sediment samples. Non-detects were replaced by the detection limit followed by blank correction, and those congeners that were non-detects in the set of samples were eliminated. Grant et al. (2011) reported concentrations of the top six PBDE congeners, and total PBDEs in sediments. Concentrations of the top six congeners were above the detection limit in all samples. Undetectable values of other PBDEs were replaced by a random number between zero and the limit of detection before principal component analysis (PCA); however, it is not known whether this was also performed for calculation of total PBDEs. Ross et al. (2013) reported total concentrations for PBDEs in seals, calculated as the sum of the concentrations of the peaks that were detectable in at least 70% of the seal samples from all sites. Where congeners were undetectable, the detection limit was substituted. Congeners that were detected in less than 70% of the samples were not included in calculations performed by Ross et al. (2013). However, total PBDE concentrations reported in Ross et al. (2013) were not part of the data assessment in this study. In the Metro Vancouver sediment and tissue monitoring reports (2015, 2016), congener concentrations less than the detection limit were equalled to zero when total PBDE concentration or means were calculated for the biennial reports. The concentrations of individual congeners in each sediment or tissue sample were also reported; concentrations below the detection limit were reported as <DL, congener specific in tissue samples and <DL, congener and sample specific in sediment samples.

For this data assessment, concentrations of total PBDEs in sediments, monitored by Metro Vancouver (2015, 2016), were calculated for each subsample (three to six per year and sampling location) by adding up all analyzed homologues and/or congeners. Analyzed constituents differed slightly between sampling occasion: total PBDEs were reported in 2011; homologues were reported in 2015; and congeners were reported in 2008, 2011, 2013, and 2015. Data on blanks were not reported with the Metro Vancouver sediment raw data, and reported analytical data were not blank corrected. In 2015, the analytical laboratory reported overall good compliance with quality control objectives; 90% compliance for all quality control types excluding surrogate spikes and 95% including surrogate spikes.

In the PollutionTracker program, data were blank-corrected and, when the concentration measured in the blank was equal to or greater than the sample concentration, the sample concentration was reported as zero. However, for this data assessment, total PBDE concentrations in sediments were calculated without correcting for blanks. To evaluate the impact of not blank-correcting PBDE concentrations, the calculated data were compared to the values published by Ocean Wise (Ocean Wise 2020), which were blank corrected. The average absolute difference is 0.09 ng/g dw and the average relative difference is 11%. Blank correcting has a larger effect when the measured concentration is low (<0.2 ng/g) and well below the guideline, which would not impact the data assessment considerably.

The same approach was used for adding up total homologue concentrations in tissue samples reported by Metro Vancouver and the *PollutionTracker* program.

3.3 Assessment Results

The following sections discuss the sampling results for the eleven priority PBDE congeners in sediment and tissues in Burrard Inlet.

Heat maps illustrating the distribution of PBDE levels in sediment and tissue across all monitoring programs and years where data are present are provided in Appendix B. These heat maps may be used to illustrate the best understanding of potential areas where PBDE levels may be relatively high in comparison to the rest of Burrard Inlet.

3.3.1 *Sediment Results*

Grant et al. (2011)

The top six congeners detected in surficial sediment samples collected from Burrard Inlet are listed in Table 3. BDE-209 was the dominant congener for the entire study area (including areas outside Burrard Inlet), accounting for 52% of the average total concentration (Grant et al., 2011). The BC WWQG for total PBDEs in sediment was exceeded at both Burrard Inlet sites. This study concluded that there are obvious coast wide “hot spots” for PBDEs, as depicted in Figures 5 and 6, which are likely influenced by local discharges. Higher PBDE concentrations were found at shallower depths compared to deeper water areas, reflecting proximity to current active sources (Grant et al., 2011).

Table 3. Concentrations of PBDE congeners detected in Burrard Inlet sediment (ng/g dry weight) (Grant et al., 2011)

Sample Site	BDE-209	BDE-47	BDE-99	BDE-49	BDE-100	BDE-17	Total PBDEs
Moody Arm (Site 2)	5.3	0.67	0.44	0.41	0.14	0.27	7.93
Burrard Inlet (Site 3)	9.12	0.85	0.82	0.46	0.23	0.25	21.7

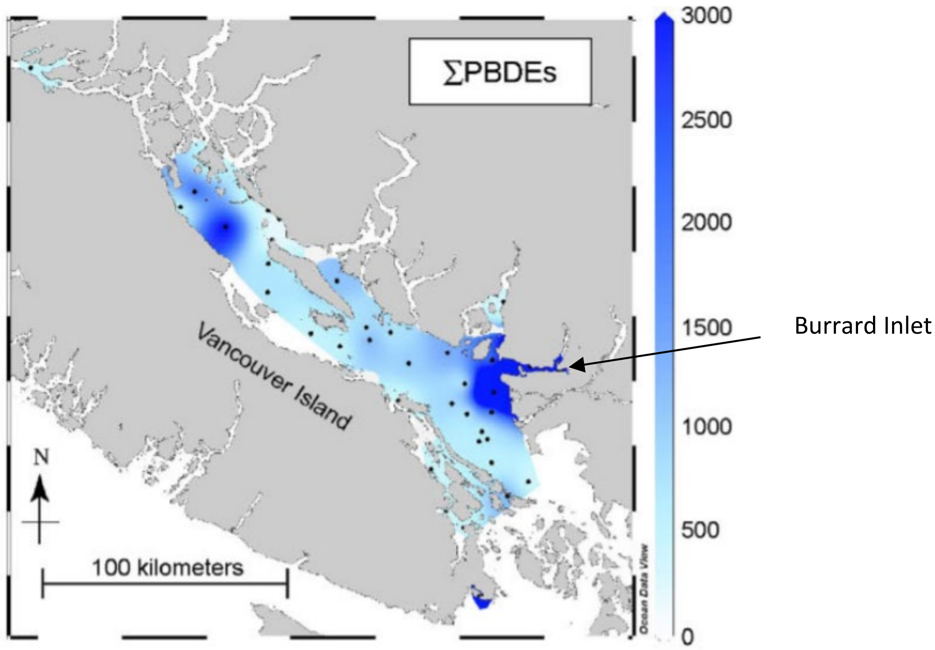

Figure 6. Contour plot of total PBDE levels in surficial sediment samples, showing Burrard Inlet concentrations (in pg/g dry weight) in relation to the rest of the sampled areas in the Strait of Georgia, British Columbia (Grant et al., 2011). Concentrations are presented as ng/g, rather than pg/g, in Table 3 for direct comparison to the BC WWQGs.

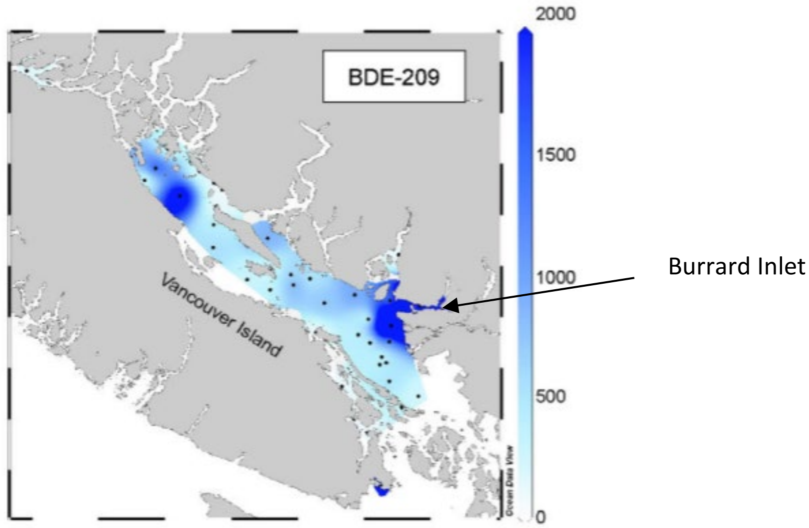

Figure 7. Contour plot of BDE-209, the most prevalent PBDE congener in surficial sediment (in pg/g dry wt) (Grant et al., 2011). Concentrations are presented as ng/g, rather than pg/g, in Table 3 for direct comparison to the BC WWQGs.

Metro Vancouver (Enkon 2015, 2016)

The results for priority PBDEs from Metro Vancouver's sediment monitoring in Burrard Inlet are discussed below, with all units reported as ng/g dry weight.

The total concentration of PBDEs, shown in Figure 8, exceeded the BC WWQGs in all samples, except for two subsamples collected at the north Indian Arm site. In general, the lowest total PBDE concentrations were found in the Indian Arm North site, followed by the Outer Harbour South site. The highest total PBDE concentrations were detected at the Indian Arm South site, followed by the Port Moody Arm site. The total PBDE concentrations in sediments are in the same magnitude as concentrations reported by Grant et al. (2011).

One sediment sample collected from the Indian Arm North site in 2013 showed very high concentrations of several homologues, adding up to a total PBDE concentration of approximately 40 ng/g. The analytical laboratory confirmed the results and noted that they were indicative of contamination by one of the most commonly used flame retardants in foam and carpets. They proposed that the sample contained a speck of foam (Metro Vancouver 2015). This single sample of high concentrations is an outlier; total PBDE concentration in other samples collected from the same site are considerably lower at 1 and 2 ng/g, which would indicate that the site is not a PBDE hot spot compared to other sites in Burrard Inlet.

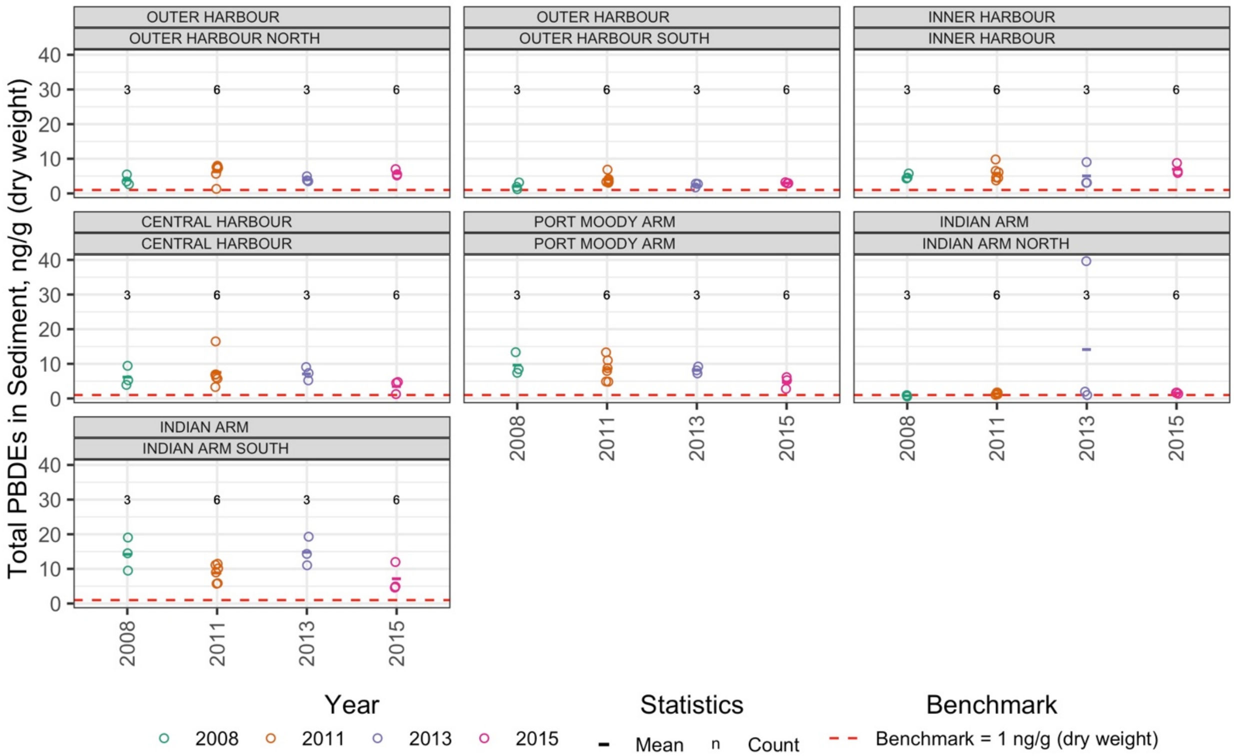

Figure 8. Calculated total PBDEs in Metro Vancouver sediment samples, showing the BC WWQG of 1 ng/g dry weight.

Below follows a discussion of priority BDE congeners that had elevated concentrations in sediment samples:

Tri-BDEs

- BDE-17: The highest concentrations were found in 2008 at the Indian Arm South site (0.21 ng/g dw) and at the Port Moody Arm site (0.13 ng/g dw).

Tetra-BDEs

- BDE-47: The highest concentration occurred at the Indian Arm North site (11 ng/g dw) in 2013.
- BDE-49: Concentrations were highest in 2008 at the Indian Arm South site (0.45 ng/g dw). The Outer Harbour and Port Moody Arm sites also had relatively high BDE-49 concentrations in 2008 (0.34 ng/g and 0.32 ng/g, respectively).
- BDE-66: The highest concentration occurred at the Indian Arm North site in 2013 (0.14 ng/g dw). All other samples had much lower concentrations (less than 0.05 ng/g dw).

Penta-BDE

- BDE-99: The highest concentration occurred in 2013 at the Indian Arm North site at 10 ng/g dw.
- BDE-100: Concentrations were highest at the Indian Arm North site in 2013 at 2 ng/g dw. BDE-100 concentrations at all other sampled sites were below 0.25 ng/g dw.

Hexa-BDEs

- BDE-153: The highest concentration occurred at the Indian Arm North site in 2013 (0.82 ng/g dw).
- BDE-154: The highest concentration was observed at the Indian Arm North site in 2013 (0.82 ng/g dw).

Octa-BDEs

- BDE-203: The highest concentration was measured at the Indian Arm South site in 2008 (0.41 ng/g dw). The same site also had an elevated concentration in 2013 (0.24 ng/g dw).

Nona-BDE

- BDE-207: Concentrations were highest at the Indian Arm South and Central Harbour sites in 2008 and 2013, respectively (0.99 ng/g and 0.75 ng/g dw, respectively).

Deca-BDE

- BDE-209: The highest concentrations occurred at the Indian Arm South site at 15 ng/g dw in 2008 and 2013, and at the Central Harbour site (14 ng/g dw) in 2011. Other sampled areas that had comparatively high concentrations of BDE-209, were the Indian Arm North site (13 ng/g dw) in 2013, and Port Moody Arm site (11 ng/g) in both 2008 and 2011.

To conclude, the highest concentrations were found for BDE-209 (15 ng/g dw), followed by BDE-47 (11 ng/g dw), and BDE-99 (10 ng/g dw). This is consistent with the results reported by Grant et al. (2011) and Morales-Caselles et al. (2017).

Morales-Caselles et al. (2017)

In the Morales-Caselles et al. (2017) study, the top PBDE congeners are listed as a percentage of total PBDEs detected in Burrard Inlet (Table 4). These sediment results are similar to the Grant et al. (2011) and Metro Vancouver's studies, detecting mainly BDE-209 and BDE-47. The total concentration of PBDEs, calculated by Morales-Caselles et al. (2017), exceeded the BC WWQGs at four out of six sites; the highest concentrations were found at the East Terminal, followed by Port Moody Arm. Lower concentrations were found at the Indian Arm and Neptune Bank sites.

Table 4. Contribution of the most prevalent congeners to total PBDE concentrations in Burrard Inlet sediment (Morales-Caselles et al., 2017).

Sample ID	Site	BDE-209 (%)	BDE-47 (%)	BDE-99 (%)	BDE-49 (%)	BDE-207 (%)	BDE-17 (%)	Top 6 as % of total	ΣPBDE concentration (ng/g dry weight)
S1	English Bay	63.9	9.14	5.53	3.57	1.91	1.87	86.0	1.49
S2	Burrard Inlet-Coal Harbour	69.3	5.39	4.41	4.18	2.54	2.11	88.0	4.97
S3	Burrard Inlet-Neptune Bank	66.4	3.44	1.65	1.27	4.47	0.34	77.5	0.536
S4	Burrard Inlet – Indian Arm	10.8	32.2	14.5	5.78	2.86	4.12	70.2	0.457
S5	Burrard Inlet – Moody Arm	73.8	5.21	3.83	3.35	1.86	1.83	89.9	5.42
S6	Burrard Inlet – East Terminal	53.7	7.83	10.7	4.23	1.50	2.43	80.4	9.87

PollutionTracker

The results for priority PBDEs from *PollutionTracker*'s sediment monitoring in Burrard Inlet are discussed below, with all units reported as ng/g dry weight.

The total concentration of PBDEs, shown in Figure 9, exceeded the BC WWQGs in 12 out of 15 sediment samples collected in 2015 and 2016. The highest total PBDE concentration at over 17.4 ng/g dw was found in sediments from the Inner Harbour site BI 12, followed by 15.2 ng/g dw at the Central Harbour site BI 2, and 6.71 ng/g dw at Outer Harbour site BI 7. The lowest total PBDE concentrations were found at the Indian Arm site 2, and Burrard Inlet sites 3 and 4, which are located in the Central Harbour and the Inner Harbour, respectively. The total PBDE concentrations reported by *PollutionTracker* in sediments are generally in the same magnitude as concentrations reported in other Burrard Inlet sediment studies.

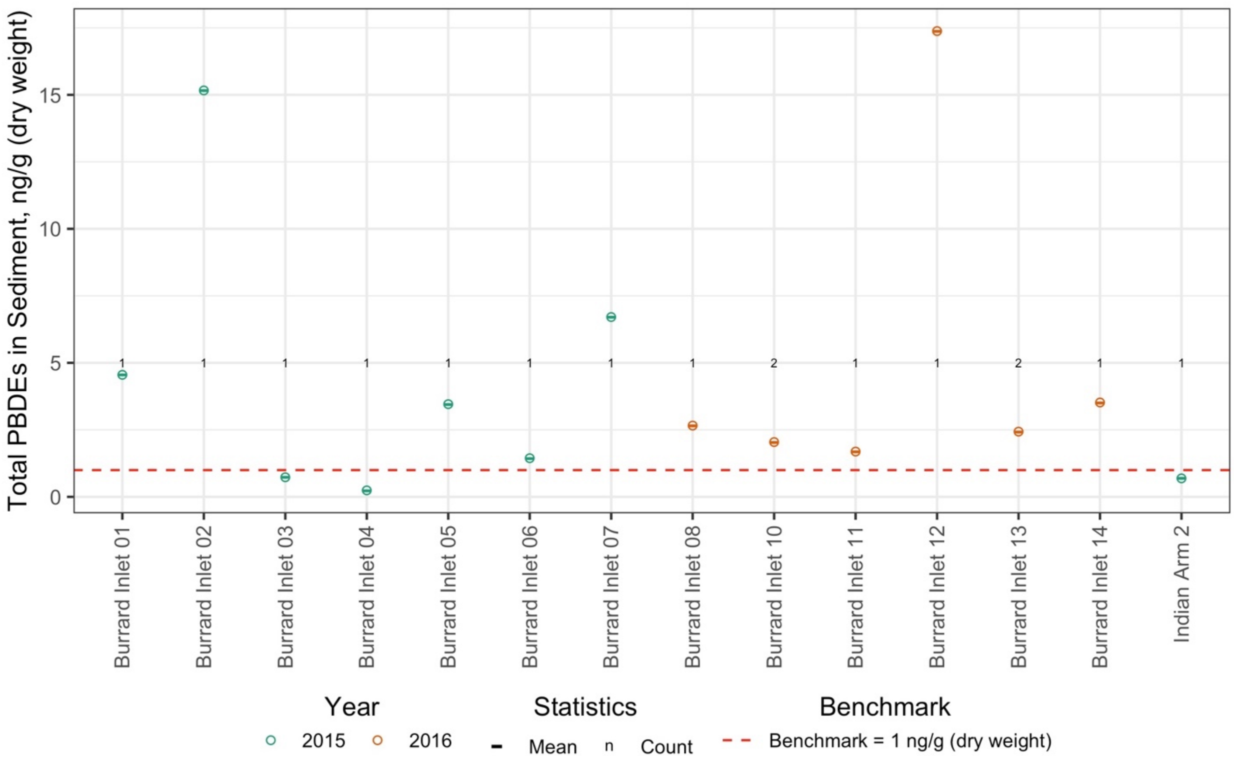

Figure 9. Calculated Total PBDEs in PollutionTracker sediment samples, showing the BC WWQG of 1 ng/g dry weight.

Below follows a discussion of priority BDE congeners that had elevated concentrations in sediment samples:

Tri-BDE

- BDE-17: The highest BDE-17 concentration measured in sediment was at the Port Moody site BI 1 in 2015 at 0.12 ng/g dw.

Tetra-BDE

- BDE-47: The highest concentration of BDE-47 was measured at Inner Harbour site BI 12 in 2015 (0.23 ng/g dw). Other sites with elevated concentrations of BDE-47 include the Inner Harbour site BI 5 (0.20 ng/g dw in 2015), Inner Harbour site BI 8 (0.175 ng/g dw), and Inner Harbour site BI 14 (0.18 ng/g dw).
- BDE-49: The highest BDE-49 concentration was measured at the Inner Harbour site BI 5 in 2015 at 0.11 ng/g dw.
- BDE-66: The highest BDE-66 concentration occurred at the Inner Harbour site BI 12 in 2016 at 0.0100 ng/g dw. Another Inner Harbour site (BI 14) also had an elevated level of BDE-66 (0.0026 ng/g dw) in 2016.

Penta-BDE

- BDE-99: The highest concentration of BDE-99 was measured at the Inner Harbour site BI 12 in 2016 (0.25 ng/g dw). The Inner Harbour site BI 14 also had an elevated concentration in 2016 (0.21 ng/g dw).

- BDE-100: Most of the sites had similar concentrations with a maximum concentration occurring at the Inner Harbour site BI 12 (0.074 ng/g dw).

Hexa-BDE

- BDE-153: The highest concentration was observed at the Inner Harbour site BI 12 (0.037 ng/g dw) in 2016.
- BDE-154: The highest concentration of BDE-154 was observed at the Inner Harbour site BI 12 in 2016 (0.029 ng/g dw).

Octa-BDE

- BDE-203: The highest concentration of BDE-203 was measured at the Port Moody site BI 2 in 2015 at 0.068 ng/g dw.

Nona-BDE

- BDE-207: The highest mean concentration was measured at the Port Moody site BI 2 in 2015 at 1.4 ng/g dw.

Deca-BDE

- BDE-209: The highest level of BDE-209 was measured at the Inner Harbour site BI 12 (16 ng/g dw) in 2016 (Figure 12). The Port Moody site BI 2 also had an elevated concentration of BDE-209 (11 ng/g dw) in 2015.

To conclude, the highest concentrations were found for BDE-209 (16 ng/g dw), followed by BDE-207 (1.4 ng/g dw), BDE-99 (0.25 ng/g dw), and BDE-47 (0.23 ng/g dw). This is consistent with the results reported in other Burrard Inlet sediment studies.

3.3.2 *Tissue Results*

PBDE tissue results were available from Metro Vancouver, Ross et al. (2013), and *PollutionTracker*. Metro Vancouver and *PollutionTracker* results were screened against the most conservative tissue benchmark in Table 1, i.e., either the fish tissue, wildlife diet, or the calculated screening value benchmark, for each homologue. The total homologue group benchmark was used for all except tetra-BDE, for which the BDE-47 benchmark was used at it is more conservative than the total tetra-BDE benchmark.

Ross et al. (2013) and Nöel and Ross (2017)

Ross et al. (2013) indicated that PBDE patterns in seals did not vary much between the four study sites (with the exception of BDE-99, which was highest for Gertrude Island seals). BDE-47 was the dominant PBDE congener in seals, comprising between 77.2 ± 1.1 and 87.3 ± 2.3% of total PBDE concentrations.

Of the total contaminant concentrations measured in seal pups from all four sampling sites in the Salish Sea, PCBs dominated, comprising an average of 64.1%, while PBDEs comprised 35.5%, indicating that PBDEs also represent an important contaminant of concern (Ross et al., 2013).

Preliminary PBDE studies were also done in harbour seal pups in Burrard Inlet (Noël and Ross, 2017). The average PBDE level in harbour seal pup blubber biopsies was 292.7 ng/g lipid weight (lw)[4], with a maximum PBDE concentration of 588.95 ng/g lw.

[4] This study reports concentrations in lipid weight, while the other tissue-related studies report in wet weight.

These two harbour seal studies showed that PBDE concentrations were higher at sites close to urban areas (Gertrude Island, Burrard Inlet) relative to seals from more remote areas. These studies also documented declines in PBDEs in harbour seal pups following regulatory changes to the use and sale of PBDEs in Canada (Ross et. al., 2013 and Nöel and Ross, 2017).

Metro Vancouver

Results for PBDEs in English sole liver, muscle, and whole body samples collected from Burrard Inlet are discussed below. In 2007, whole bodies, livers and muscle were sampled, while in 2012, only whole bodies and muscle were sampled. All results are reported in ng/g wet weight (ww). Figures are shown for those homologue groups where total concentrations are exceeding the homologue-specific benchmark. Also, priority congeners are discussed in more detail. Although additional congeners within homologue groups may have been detected, only those identified as priority congeners were considered.

Tri-BDE

- Total tri-BDE concentrations were several orders of magnitude below the homologue benchmark at 120 ng/g ww in all tissue samples. The highest total tri-BDE concentration at 0.472 ng/g ww was found in a whole-body tissue sample from the Central Harbour site. However, no specific site appeared to have considerably higher total tri-BDE concentrations in tissue samples than other sites.
- BDE-17: The congener was analyzed only in samples collected in 2012. The highest tissue concentration of BDE-17 was measured in whole-body tissue at Port Moody Arm and Central Harbour (both 0.35 ng/g ww).

Tetra-BDE

- Total tetra-BDE and BDE-47 concentrations exceeded the benchmark for BDE-47 (4 ng/g ww) in numerous tissue samples collected from all Burrard Inlet sites (Figure 10 illustrates BDE 47).
- The most conservative benchmark among the tetra-BDEs is the BDE-47 screening values for human consumption of finfish and shellfish (Table 1) and was used for this assessment because BDE-47 was determined to comprise a large majority of total tetra-BDEs. The highest total tetra-BDE concentration at 14.4 ng/g ww (14 ng/g ww BDE-47) was found in a sample from the Outer Harbour Site, followed by samples from the Inner Harbour (13.5 ng/g ww tetra-BDE and 11.9 ng/g ww BDE-47) and Central Harbour (10.8 ng/g ww tetra-BDE and 9.5 ng/g ww BDE-47) sites.

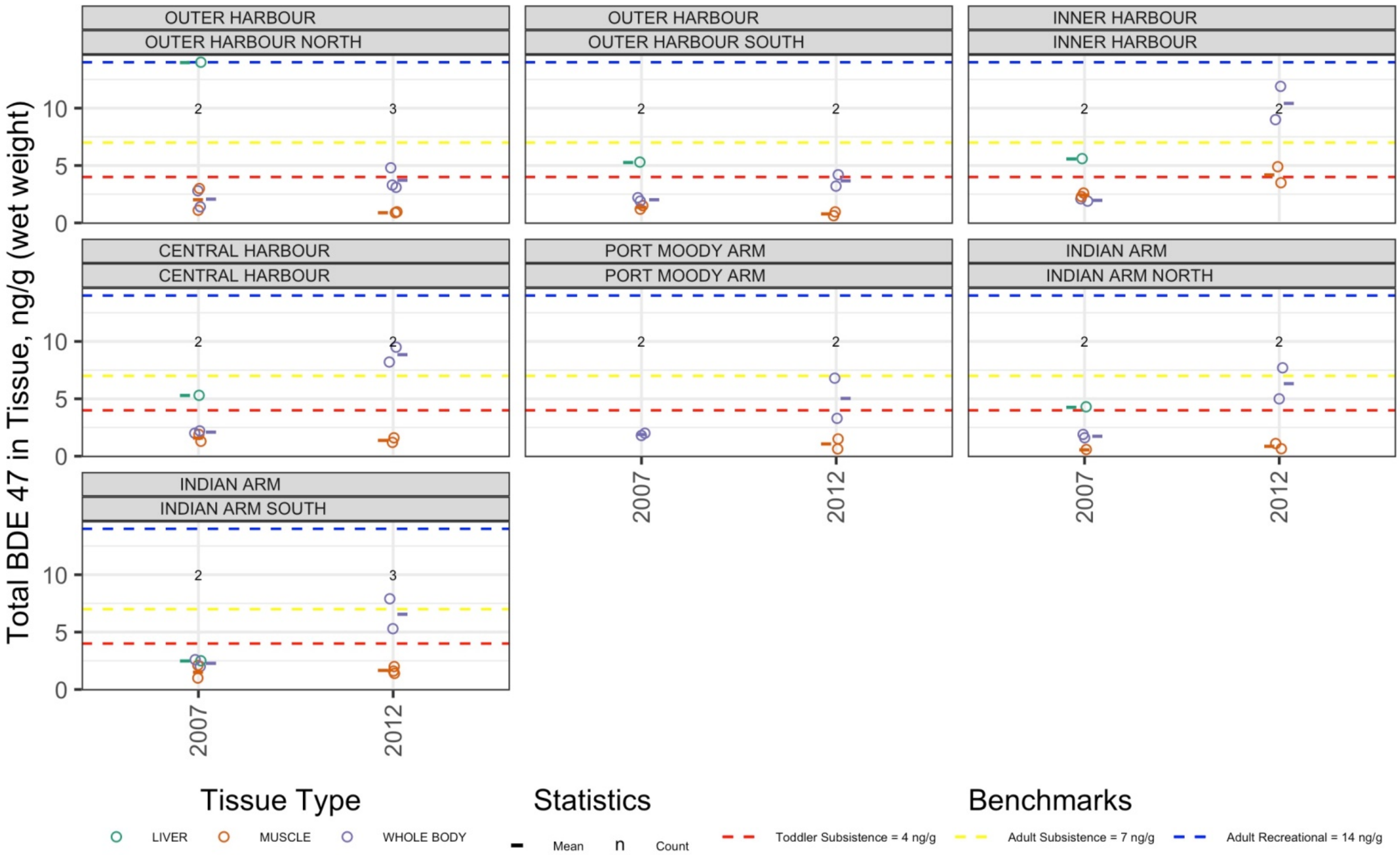

Figure 10. BDE 47 levels in Metro Vancouver English sole fish tissue samples, showing screening benchmarks of 4 ng/g wet weight for toddler subsistence fishers, 7 ng/g wet weight for adult subsistence fishers and 14 ng/g wet weight for adult recreational fishers.

- BDE 47: The highest concentration in 2007 occurred in liver samples from the Outer Harbour North site (14 ng/g ww). In 2012, the highest tissue concentration occurred in whole body tissue at the Inner Harbour site (12 ng/g ww). See Figure 10.
- BDE-49: Concentrations were only measured in 2012, with the highest tissue concentration occurring in whole body tissue at the Inner Harbour site and Indian Arm South site (1.2 ng/g ww).
- BDE-66: In 2007, the highest concentration of BDE-66 was found in whole body tissue at the Outer Harbour North site (0.44 ng/g ww). In 2012, the highest concentration occurred in whole body tissue at the Inner Harbour site (0.3 ng/g ww).

Penta-BDE

- Total penta-BDE concentrations exceeded the benchmark at 1 ng/g ww in most tissue samples collected in Burrard Inlet (Figure 11). The highest total penta-BDE concentration at 10.3 ng/g ww was found in a liver tissue sample collected from the Outer Harbour North site. This was followed by whole body tissue samples collected from the Inner Harbour (7.10 and 6.84 ng/g ww) and Central Harbour (6.10 and 5.57 ng/g ww) sites, all collected in 2012.

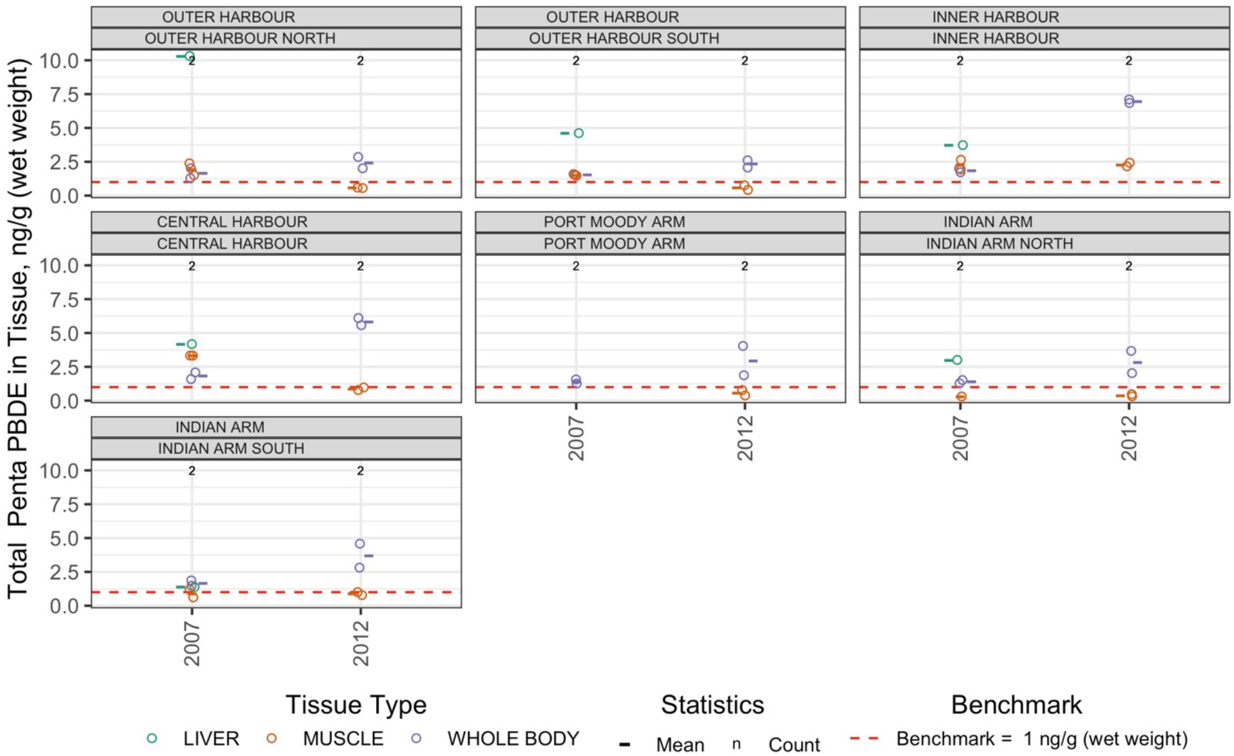

Figure 11. Total penta-BDE levels in Metro Vancouver English sole fish tissue samples, with screening benchmark of 1 ng/g wet weight.

- BDE-99: Concentrations measured in liver samples were above the congener-specific BC WWQG of 1 ng/g ww, except for the Port Moody Arm and Indian Arm South sites. The highest liver concentration measured in 2007 occurred at the Outer Harbour North site (6.8 ng/g ww). In 2012, all whole-body tissue concentrations were above the BC WWQG, with the highest concentration occurring at the inner harbour site (4.9 ng/g ww).
- BDE-100: In 2007, most liver samples had BDE-100 concentrations above the congener-specific BC WWQG of 1 ng/g, with the exception of the Port Moody Arm and Indian Arm South sites. The highest concentration occurred in liver at the Outer Harbour North site in 2007 (3.4 ng/g ww). In 2012, whole body tissue concentrations were above the BC WWQG for all sites, with the highest concentration occurring at the Inner Harbour site (2.5 ng/g ww).

Hexa-BDE

- Total hexa-BDE concentrations exceeded the benchmark (4 ng/g ww) in one liver sample collected in 2007 from the Outer Harbour North site (Figure 12). Other tissue samples from this site had total hexa-BDE concentration below 1 ng/g ww. Among the other sites, higher concentrations were generally found at the Inner Harbour (2.43 and 2.75 ng/g ww) and Central Harbour (1.45 and 1.80 ng/g ww) sites, although the concentrations were below the benchmark value.

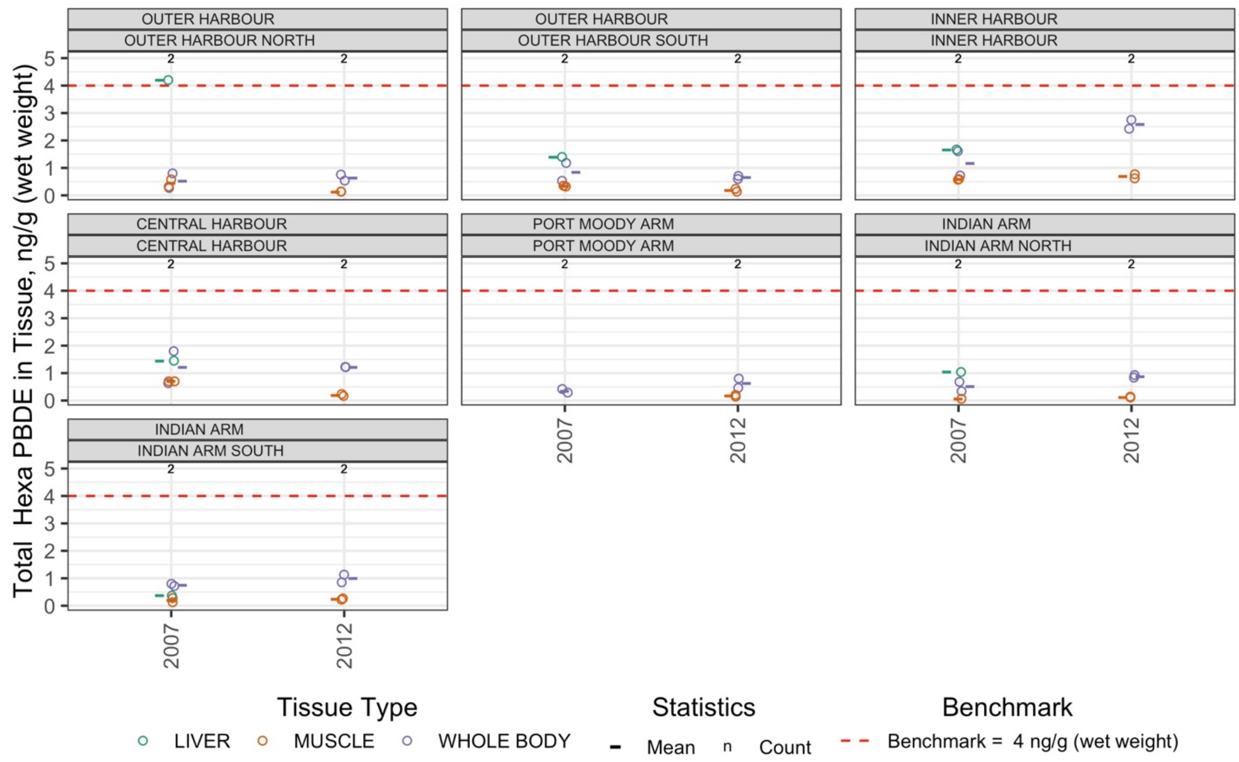

Figure 12. Total hexa-BDE levels in Metro Vancouver English Sole fish tissue samples, with screening benchmark of 4 ng/g wet weight.

- BDE-153: In 2007, liver samples had the highest concentration of BDE-153 at the Outer Harbour North site (2.5 ng/g ww). In 2012, the highest tissue concentration occurred in whole-body samples from the Inner Harbour site (1.7 ng/g ww).
- BDE-154: The highest tissue concentration occurred in liver from the Outer Harbour North site (1.8 ng/g ww), in 2007. In 2012, the highest concentration occurred in whole body tissues from the Inner Harbour site (1.1 ng/g ww).

Hepta-BDE

- Concentrations of total hepta-BDEs were far below the benchmark (64 ng/g ww) in all samples collected from Burrard Inlet. The highest total hepta-BDE concentrations at 0.091 ng/g ww was detected in a whole-body tissue sample collected from the Inner Harbour site, followed by 0.085 ng/g ww in a liver sample collected from the Outer Harbour North site.

Octa-BDE

- Samples are only available from 2012. Concentrations of total octa-BDEs were far below the benchmark (63 ng/g ww) in all four samples collected from Burrard Inlet. The highest total octa-BDE concentration at 0.057 ng/g ww was detected in a whole-body sample collected from the Inner Harbour site.

Nona-BDE

- Samples are only available from 2012. Total nona-BDE concentrations in the tissue samples from Burrard Inlet were several magnitudes lower than the benchmark value of 78 ng/g ww. The highest total nona-BDE value at 0.048 ng/g ww was detected in a whole-body tissue sample collected at the Outer Harbour North site.

- BDE-207: Concentrations were below the laboratory detection limit, with the exception of samples from the Outer Harbour North and Central Harbour sites. These had whole body concentrations of 0.028 ng/g and 0.016 ng/g, respectively.

Deca-BDE

- Concentrations of total deca-BDEs were below the benchmark of 9 ng/g ww in all tissue samples. The highest total deca-BDE concentration at 1.2 ng/g ww was detected in a liver sample collected from the south outer harbour site; total deca-BDE concentrations in other tissue samples were at least one order of magnitude lower than this sample.
- BDE-209: The Central Harbour site had the highest concentration occurring in liver samples, at 0.1 ng/g ww in 2007. All other sites in that year were below detection limits. In 2012, the highest tissue concentration occurred in whole body samples at 0.3 ng/g ww at the Outer Harbour North site.

PollutionTracker

PollutionTracker mussel tissue data results are discussed below. All concentrations are in ng/g wet weight (ww). Priority congeners are discussed more in detail. Although additional congeners within homologue groups may have been detected, only those identified as priority congeners were considered.

Tri-BDE

- Total tri-BDE concentrations were several orders of magnitude below the homologue benchmark at 120 ng/g ww in all mussel tissue samples. The highest total tri-BDE concentration at 0.051 ng/g ww was found in a sample collected from the Port Moody Arm (BI 1). The lowest tri-BDE concentration was found in a mussel tissue sample from the south Indian Arm site (IA 2).

Tetra-BDE

- Total tetra-BDE concentrations were below the benchmark (4 ng/g ww) in all tissue samples. The most conservative benchmark among the tetra-BDEs is the BDE-47 screening values for human consumption of finfish and shellfish (Table 1), and was used for this assessment. The highest total tetra-BDE concentration at 0.705 ng/g ww was found in a sample from the Outer Harbour site BI 6. The lowest tetra-BDE concentration was found in a tissue sample from the south Indian Arm site IA 2.
- BDE-49: The highest BDE-49 concentration occurred at the Outer Harbour BI 6 in 2015 (0.034 ng/g ww). The Port Moody Arm (BI 1) and Central Harbour (BI 3) sites also had elevated concentrations of BDE-49 in 2015 (0.022 and 0.023 ng/g ww, respectively).
- BDE-47: The highest concentration of BDE-47 measured in mussel tissue was at the Outer Harbour site BI 6 (0.62 ng/g ww) in 2015.

Penta-BDE

- Total penta-BDE concentrations were below the benchmark (1 ng/g ww) in all collected mussel tissue samples. The highest total penta-BDE concentration at 0.547 ng/g ww was found in a mussel tissue sample collected from the Outer Harbour site BI 6. The lowest penta-BDE concentration was found in a tissue sample from the south Indian Arm site IA 2.

- BDE-99: Concentrations were slightly elevated in the Outer Harbour site BI 6 (0.37 ng/g ww), relative to other sites.
- BDE-100: Although most sites had similar concentrations of BDE-100, mussels at the Outer Harbour site BI 6 had a slightly higher concentration in 2016 (0.126 ng/g ww).

Hexa-BDE

- Total hexa-BDE concentrations were several orders of magnitude below the homologue benchmark (4 ng/g ww) in all collected mussel tissue samples. The highest total hexa-BDE concentration at 0.061 ng/g ww was found in a mussel tissue sample collected from the Outer Harbour site BI 6. The lowest hexa-BDE concentration was found in a tissue sample from the south Indian Arm site IA 2.

Hepta-BDE

- Total hepta-BDE concentrations were several orders of magnitude below the homologue benchmark (64 ng/g ww) in all collected mussel tissue samples. The highest total hepta-BDE concentration at 0.009 ng/g ww was found in a mussel tissue sample collected from the Outer Harbour site BI 6.

Octa-BDE

- Total octa-BDE concentrations were several orders of magnitude below the homologue benchmark (63 ng/g ww) in all collected mussel tissue samples. The highest total octa-BDE concentration at 0.004 ng/g ww was found in a mussel tissue sample collected from the Outer Harbour site BI 6.

Nona-BDE

- Total nona-BDE concentrations were several orders of magnitude below the homologue benchmark (78 ng/g ww) in all collected mussel tissue samples. The highest total nona-BDE concentration at 0.039 ng/g ww was found in a mussel tissue sample collected from the Outer Harbour site BI 6.
- BDE-207: The highest concentration measured in mussel tissue was from the Outer Harbour site BI 6 (0.015 ng/g ww).

Deca-BDE

- Total deca-BDE concentrations were several orders of magnitude below the homologue benchmark (9 ng/g ww) in all collected mussel tissue samples. The highest total nona-BDE concentration at 0.080 ng/g ww was found in a mussel tissue sample collected from the Outer Harbour site BI 9.
- BDE-209: The highest concentration measured in mussel tissue was 0.08 ng/g ww at the Inner Harbour site BI 9 in 2016.

3.3.3 *Summary of Data Assessments*

Analysis of the Burrard Inlet data showed that concentrations of BDE-47, -99, -100, and -209 were dominant in sediment samples. Total PBDE concentrations reported by Morales-Caselles et al. (2017) and calculated from data reported by Metro Vancouver (2015, 2016) and the *PollutionTracker* program were above the BC Working Sediment Quality Guideline of 1 ng/g dw in most samples. Sites with the highest PBDE sediment concentrations were generally in the Inner Harbour, Outer Harbour, and Port Moody Arm.

Analysis of fish tissue data showed that BDE-47, -99 and -100 were found at the highest concentrations. Concentrations of total tetra- and penta-BDEs were above the homologue-specific benchmark values in most samples. In both fish and mussel tissue samples, tetra-BDEs were found at the highest concentrations. PBDE levels in fish tissues (English sole, *Parophrys vetulus*) were highest in the Outer and Inner Harbour sites, with some elevated concentrations in the Port Moody area. For mussel tissue, PBDE levels were highest in the Outer, Central, and Inner Harbours. PBDE concentrations in mussel tissue were at least one order of magnitude lower than in fish tissue samples and below the benchmarks for all homologue groups in all mussel samples.

3.4 Knowledge Gaps and Research Needs

The following key knowledge gaps and research needs have been identified:

1. Research and literature review is needed to update objectives and guidelines that are not currently protective of apex predators such as killer whales and human populations with higher-than-average seafood consumption patterns.
2. PBDEs are currently not included under the Minimum Sample Analytical Requirements during ocean disposal assessments, as part of the *Canadian Environmental Protection Act* (CEPA). PBDEs are only listed as "other contaminants of concern that may need to be characterized based on site history" (ECCC, 2013a). Mandatory inclusion of PBDEs should be considered.
3. While toxic effects mechanisms based on short-term, single life stage exposures on model organisms such as fish or rodents are relatively well understood, long-term toxic effects of PBDEs at environmentally relevant concentrations for both fish and marine mammals is lacking. Further research is needed.
4. Interactions of co-contaminants and effects on marine biota (e.g., synergistic effects of PCBs and PBDEs), as well as the effects of PBDE mixtures are not well understood and require further investigation.
5. While the use of PBDE-containing products has declined in Canada, less industrialized countries have increased their use of PBDEs (Abbasi et al., 2019). Waste products that contain PBDEs are also exported to less industrialized countries or countries in transition, contributing to continued PBDE emissions to the marine environment. Life cycle tracking of products containing PBDEs, including movement between industrialized and less industrialized regions, should be considered when looking at reducing inputs of PBDEs (Abbasi et al. 2019).

4. PROPOSED OBJECTIVES FOR PBDEs IN BURRARD INLET

4.1 Proposed Objectives

The proposed objectives for PBDEs in Burrard Inlet are presented in Table 5 in bold print. Other benchmarks are also included in Table 5 for the purposes of monitoring for protection of other water values, in addition to the most sensitive values for each homologue. With the exception of the sediment objective, these objectives may not be protective of apex predators such as southern resident killer whales. In addition to these numeric objectives, an overall objective is for a decreasing trend in PBDE concentrations in all media.

Table 5. Proposed Benchmarks and Water Quality Objectives for PBDEs in Burrard Inlet (all sub-basins); proposed WQOs are bolded and highlighted

Homologue	Congener(s)	Water[1] (ng/L)	Sediment[2] (ng/g dry weight)	Tissue		
				Fish Tissue (ng/g wet weight)[3]	Wildlife Diet (ng/g wet weight food source)[4]	Screening value for human consumption of finfish and shellfish (ng/g wet weight)[5]
Tri-BDE	Total	**46**		**120**	-	
Tetra-BDE	Total	**24**		88	**44**	
	BDE-47					**4**
Penta-BDE	Total	**0.2**		**1**	3 (mammal) 13 (birds)	70
Hexa-BDE	Total	**120**		420	**4**	
	BDE-153					7
Hepta-BDE	Total	**17**		-	**64**	
Octa-BDE	Total	**17**		-	**63**	105
Nona-BDE	Total	-		-	**78**	
Deca-BDE	Total	-		-	**9**	
	BDE-209					246
Total PBDEs[5]			**1**			

[1] Based on a minimum of 5 surface samples in 30 days collected during the wet season.
[2] Based on a minimum of 1 composite sample composed of at least 3 replicates.
[3] The concentration not expected to cause adverse effects to the fish, based on a whole-body composite sample consisting of at least 5 individual fish. See Rao et al. (in prep.) for additional guidance.
[4] The concentration in whole food not expected to cause adverse effects to wildlife consumers, based on a whole-body composite sample consisting of at least 5 individual fish or 25 bivalves. See Rao et al. (in prep.) for additional guidance.
[5] The concentration not expected to cause adverse effects to human consumers, based on a composite sample consisting of at least 5 individual fish or 25 bivalves. See Rao et al. (in prep.) for additional guidance.

4.2 Rationale

The most conservative and protective of the available benchmarks are proposed as water quality objectives for Burrard Inlet. In July 2020, ENV adopted the FEQGs (ECCC, 2013b) for PBDEs as working water quality guidelines for water and tissue. The FEQGs were developed recently, include guidelines for the marine environment, and are the only relevant environmental quality guidelines currently available for PBDEs in most media.

Although no water sampling data were available or assessed in this report for PBDEs, objectives are proposed for PBDEs in water for consistency with the federal Environmental Quality Guidelines being recommended for the protection of Southern Resident Killer Whales (ECCC, 2020). The recommended EQGs for PBDEs in water have been adopted by BC ENV as Working Water Quality Guidelines (ENV, 2020).

They are recommended as Environmental Quality Guidelines in the context of the federal work on SRKW, and have been adopted as BC WWQGs.

There are many limitations to the FEQGs, however. The FEQGs for individual PBDE homologue groups in most cases are based on calculations derived from a single data point due to lack of data availability (ECCC, 2020b). This results in considerable uncertainty. The FEQGs for sediment were not adopted as BC Working Water Quality Guidelines, as they are protective of sediment dwelling invertebrates, but not higher trophic organisms. This is because there is considerable uncertainty in these values, which are based on endpoints such as mortality, growth, and reproduction, and they do not consider more subtle or sublethal effects (e.g., hormone disruption) in long-lived species with limited reproductive output (e.g., marine mammals) (Moss et al., 2010). The FEQGs for wildlife diet were calculated from studies done on laboratory animals. If the mammalian wildlife diet guideline were used to back calculate the sediment quality guideline, the value would be reduced by two orders of magnitude for hexa-BDE and halved for tetra-BDE (A. Tillmanns, BC ENV, *pers. comm.*, June 2020). Ideally, the FEQGs need to be developed further so that they are protective of higher trophic level organisms. At the time of writing, multiple parties are working to address this gap through the Southern Resident Killer Whale Contaminants Technical Working Group (ECCC, 2020).

To address concerns about the bioaccumulation potential of PBDEs and potential effects in killer whales, Alava et al. (2016) used a bioaccumulation model to determine the acceptable concentration of PBDEs in sediment given the PBDE endocrine disruption threshold of 1500 ug/kg lipid total PBDE. They used this model to generate a recommended sediment guideline for total PBDEs (1 µg/kg dw), based on the sum of congeners that were measured in the blubber samples (BDEs 17, 28, 35, 37, 47, 49, 71, 75, 85, 99, 100, 120, 153, 154). This value is protective of 95% of northern and southern resident killer whale populations[5]. The model allows sediment quality guidelines to be calculated[6] using bioaccumulation factors (Morales-Caselles et al., 2017). The ECCC-led Southern Resident Killer Whale Contaminants Technical Working Group recently recommended guidelines that are protective of Southern Resident Killer Whales, including the sediment guideline for PBDEs derived for this purpose by Alava et al. (2016). These recommended guidelines are intended to be made publicly available by ECCC. In the interim, the sediment guideline for PBDEs from Alava et al (2016) was adopted by ENV as a working sediment quality guideline (ENV, 2020), and is being proposed as the sediment quality objective for Burrard Inlet.

Tsleil-Waututh Nation has a goal to obtain 10% of their diet from Burrard Inlet. The federal fish tissue and wildlife diet guidelines, adopted by BC as Working Water Quality Guidelines, were compared to calculated screening values protective of the most sensitive human receptor, i.e., a toddler from an Indigenous or subsistence fishing population. Toxicological information and human health risk assessment guidance from the US EPA (2017) and Health Canada (2010 a,b,c; 2012) was used to calculate these screening values (see Section 3.1). The most protective of these benchmarks is proposed as an objective for each homologue group. Toxicological information is available for individual congeners rather than homologue groups, but in most cases the Working Water Quality Guideline for fish tissue or wildlife diet for an entire homologue group was more protective than the human health screening value for an individual congener within that group. The exception is BDE-47, so an individual objective is proposed for that congener to be protective of human consumers of fish and shellfish.

5. MONITORING RECOMMENDATIONS

Although PBDE congeners are not all equally biologically active or toxic, there is currently no conclusive list of the most toxic congeners, to be able to focus monitoring efforts. The following recommendations are provided to help guide future monitoring programs and to inform further development of proposed water quality objectives for Burrard Inlet.

1. Further sediment and biota sampling for PBDEs:

[5] The backward application of the Biota Sediment Accumulation Factor (BSAF) model was performed to come up with proposed sediment concentrations that are protective of 95% of southern resident killer whale populations (Alava et al., 2016).

[6] The model was designed to estimate PBDE concentrations in killer whales based on PBDE concentrations in sediments and the water column throughout a lifetime of exposure (Morales-Caselles et al., 2017).

- Currently, the Lions Gate WWTP is being upgraded from primary to tertiary treatment (completion by 2024). Monitoring of the WWTP effluent following this upgrade is recommended to determine potential changes in PBDE releases.
- Further sampling for PBDEs in fish and marine mammals in Burrard Inlet is recommended, as well as monitoring of clams, crabs, and prawns from areas where harvesting occurs or is planned to occur.
- Due to restrictions on PBDEs in Canada, there is likely to be a change in the use of alternative flame retardants. Monitoring the occurrence and temporal trends of these alternative flame retardants in Burrard Inlet sediments and biota samples is recommended (Eljarrat and Barcelo, 2018).
- Assessment of the fate and behaviour of PBDEs in Burrard Inlet is recommended, as natural events or human activities such as dredging can remobilize pollutants bound to contaminated sediments and make them available to biota (Morales-Caselles et al., 2017)

2. Clarify the status of potential releases of PBDEs from sources around Burrard Inlet. Surveys of PBDE levels in air, soil, and sediment at or near emission sources is recommended. A chemical by chemical approach will help in prioritizing PBDE congeners and identifying sources.

6. MANAGEMENT OPTIONS

The following are recommendations to reduce the flow of PBDEs into Burrard Inlet:

1. Work collaboratively to develop marine sediment quality guidelines protective of higher trophic level species, such as killer whales. Measures are required to further document and understand the impacts of PBDEs on marine mammals.
2. Divert PBDE-containing products (produced prior to bans) from the waste stream to appropriate hazardous waste disposal sites.
3. Monitor and research the environmental effects of alternative flame retardant chemicals in the marine environment in order to develop appropriate and timely guidelines for the use of these chemicals, and prior to their application in consumer products.
4. Implement mandatory inclusion of PBDE sampling under the Minimum Analytical Requirements during ocean disposal assessments (CEPA).

Initiatives in progress:

- Complete the separation of CSOs within the City of Vancouver and the City of Burnaby. This will eliminate combined sanitary and storm water discharges to Burrard Inlet, and will decrease the amount of PBDEs reaching marine species and habitats. Combined sewers are still present in Vancouver and Burnaby, but separation is in progress. Metro Vancouver’s strategy is to work with Burnaby and Vancouver to eliminate CSOs by 2050. CSO separation is a provincial goal, with each municipality working under the same target of 2050 in the Vancouver Sewage Area (Metro Vancouver, 2017).

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APPENDIX A: CALCULATIONS FOR SCREENING VALUES FOR HUMAN FISH CONSUMPTION

Human health screening values were calculated from the following equation (see Thompson and Stein [2021] for details) and listed in the table below. TDI was obtained from the US EPA (2021).

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

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.

Receptor population	Receptor life stage	IR (g/day)	TDI (µg/kg BW/day)	BW (kg)	RAF (fraction)	SV (ng/g, w/w)
Tetra-BDE (BDE-47)						
Recreational fisher	Adult	111	0.1	76.5	100%	14
Subsistence fisher	Adult	220	0.1	76.5	100%	7
	Child	165	0.1	35.2	100%	4
	Toddler	94	0.1	16.5	100%	4
Penta-BDE						
Recreational fisher	Adult	111	2	76.5	100%	276
Subsistence fisher	Adult	220	2	76.5	100%	139
	Child	165	2	35.2	100%	85
	Toddler	94	2	16.5	100%	70
Hexa-BDE						
Recreational fisher	Adult	111	0.2	76.5	100%	28
Subsistence fisher	Adult	220	0.2	76.5	100%	14
	Child	165	0.2	35.2	100%	9
	Toddler	94	0.2	16.5	100%	7
Octa-BDE						
Recreational fisher	Adult	111	3	76.5	100%	414
Subsistence fisher	Adult	220	3	76.5	100%	209
	Child	165	3	35.2	100%	128
	Toddler	94	3	16.5	100%	105
Deca-BDE						
Recreational fisher	Adult	111	7	76.5	100%	965
Subsistence fisher	Adult	220	7	76.5	100%	487
	Child	165	7	35.2	100%	299
	Toddler	94	7	16.5	100%	246

APPENDIX B: HEAT MAP ILLUSTRATIONS OF PBDE LEVELS IN BURRARD INLET SEDIMENT AND TISSUE

(provided in separate document)