

Industrial Contaminants

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Industrial Contaminants

BACKGROUND

Thousands of chemicals and other substances are in use in Canada. Some are intended for controlled use in the environment (such as pesticides and water purification chemicals), some may be released into the environment as byproducts (such as from burning wood and petroleum products) or accidentally through spills and leaks. Apart from pesticides and pharmaceuticals, which are regulated under other acts, 52 substances are currently defined as toxic under the Canadian Environmental Protection Act. CEPA defines a substance as toxic if it enters, or may enter, the environment in amounts or under conditions that may pose a risk to human health or the environment. Substances are also considered to be toxic if any products formed as they break down in the environment meet this definition.

Many possible environmental contaminants could be discussed in a review of toxic substances from industrial sources. The emphasis in this paper is on persistent pollutants, largely from land-based activities, that have been released into the coastal and marine environment; mercury and polycyclic aromatic hydrocarbons (PAHs) are important contaminants also included in this review.

The indicators in this paper were chosen to represent a spectrum of issues and consequences. Indicators show what is, or was, discharged into the marine environment, the movement of persistent pollutants through food webs and ecosystems, and the effect of regulatory efforts to control pollutants.

CONTAMINANT DEFINITIONS

In this paper, the term “contaminant” refers to substances, including those found naturally, that are present at concentrations above natural background levels, or whose distribution in the environment has been altered by human activity.

The term “pollutant” refers to a contaminant whose concentration in the environment is high enough to result in deleterious effects (GESAMP 1983).

Persistent Organic Pollutants (POPs)

Most of the indicators in this paper concern the group of substances known as persistent organic pollutants (POPs). They persist for a very long time in the environment because they resist being broken down by chemical and microbial processes. These chemicals have a variety of toxic effects, including disruption of hormone and immune systems of mammals. POPs include a wide variety of chlorinated and other halogenated chemicals, such as dioxins and furans, polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs), and other less well understood chemicals. They are fat soluble and accumulate in the fat reserves of organisms because they are resistant to being broken down metabolically. Through the process of bioaccumulation, such compounds become more concentrated as living organisms take them up and store them in body tissues faster than they can be broken down or excreted. When contaminated organisms are eaten by other animals, the compounds become more and more concentrated in the bodies of the predators. In a process called biomagnification, concentrations tend to increase at each step up the food web, often reaching high levels in the top predators. The concentration of PCBs in harbour seals, for example, may be five times higher than in the fish they consume (Cullon et al. 2005) and thousands of times higher than the PCB concentration in the surrounding seawater (Macdonald et al. 2003). This means that POPs may accumulate in wildlife to concentrations that affect their health or risk the health of humans who consume them.

POPs were mostly used in industrial applications or are byproducts of incineration or other industrial processes; some were used as insecticides. As a group, POPs vary greatly in their toxicity, persistence in the environment, and how they are transported once they enter the environment. Many are now banned or are subject to stringent regulations controlling their use and release. After restrictions on their production and use came into effect from the 1970s onward, concentrations of several POPs in the Canadian environment decreased substantially. Because they resist chemical decomposition, however, the contaminants already in the environment continue to circulate.

The main groups of POPs discussed in this paper are PCBs, dioxins and furans, PBDEs, PAHs, mercury, and DDT.

PCBs

Polychlorinated biphenyls (PCBs) are a class of stable, waxy to oily compounds that were used as heat-resistant coolants or insulators in electrical equipment. They were also used as plasticizers, solvents, and degreasers, and in several industrial processes. There are 209 different forms or congeners of PCBs. Congeners are variants on a common chemical structure; although related, they differ in toxicity and their fate in the environment. The PCBs sold and used commercially were mixtures of congeners.

Restrictions on industrial uses of PCBs began around 1971. Manufacture, import, and most non-electrical uses of PCBs were banned in North America in 1977 and it was made illegal to release PCBs to the environment in 1985. Canadian legislation allows PCB-

filled equipment to be used until the end of its service life, but handling, storage, and disposal are subject to stringent government requirements. As a result of these measures, the quantity of PCBs in use declined by 54% between 1992 and 2003 (Environment Canada 2005a). Most PCBs now enter the environment through leaks or improper disposal of waste oils and electrical equipment.

PCBs have a wide range of toxic effects. They adversely affect the immune, reproductive, neurological, and endocrine systems of mammals, including causing cancer (Safe 1993). The toxicity and properties of the congeners largely depend on the number and positions of chlorine atoms in the molecule. The most toxic PCB congener, PCB-126, presents about one-tenth the risk to humans of the most toxic dioxins (discussed below). With respect to toxicity, PCBs can be generally considered in two groups: One group of about a dozen congeners are structurally similar to the highly toxic dioxin 2378-TCDD and have similar, dioxin-like effects on immune systems and fetal development. The toxic effects of other PCB congeners stems from their metabolic breakdown into highly toxic intermediate compounds that affect thyroid and vitamin A physiology.

Dioxins and Furans

The dioxins (polychlorinated dibenzo-p-dioxins) consist of a group of 75 congeners; furans (polychlorinated dibenzo-p-furans) are a group of 135 congeners. The most toxic dioxin congener is 2,3,7,8-TCDD. It is the standard for comparing the relative toxicity of other compounds with dioxin- and furan-like activity (see text box: Toxic Equivalents).

These compounds form as byproducts of industrial activities such as chlorine bleaching in pulp mills and incineration of municipal and industrial wastes. Dioxins and furans are structurally similar to PCBs and were also unintended contaminants of commercial PCB formulations and some herbicides. The 2,3,7,8-TCDF congener first appeared in sediments in the early 1960s when pulp mills converted to chlorine liquid bleaching. Although trace amounts continue to be produced by industrial incineration, environmental concentrations of dioxins and furans have decreased with the phase-out of PCBs and changes in pulp and paper mill technology. Because dioxins and furans have a strong affinity for sediments, however, they will remain in the marine environment for decades and will eventually be buried by accumulating sediments.

Long-term exposure in mammals to 2,3,7,8-TCDD can affect reproduction, cause cancer and birth defects, damage the liver, and suppress the immune system (Environment Canada 1990). Rainbow trout exposed to this congener died at concentrations as low as 40 parts per quadrillion (Mehrle et al. 1988). People exposed to dioxins and furans may develop chloracne (a skin condition caused by acute exposure), liver and immune system dysfunction, and show other sensory and behavioural effects, and the US Environmental Protection Agency has classified 2,3,7,8-TCDD as a probable human carcinogen (ATSDR 1998). Because there is the potential for fish and shellfish to bioaccumulate dioxins and furans, people who eat large amounts of seafood may be at risk (reviewed in Ross and Birnbaum 2003).

TEFs & TEQs: TOXIC EQUIVALENTS

A system has been devised to estimate the combined toxic effect of the most common congeners of dioxins, furans, and PCBs. It is based on the similarity between species in the physiological effects of toxicity. Each congener is assigned a toxic equivalency factor (TEF) relative to the most toxic dioxin (2,3,7,8-TCDD), which has been assigned a TEF of 1 (see the table, below). For example, the TEF of PCB#126 for mammals and birds is 0.1, because research has determined that PCB#126 is one-tenth as toxic as 2,3,7,8-TCDD to these organisms. There are some differences in sensitivity between species: for example, fish are less sensitive to PCB#126 than mammals or birds, and are therefore assigned a lower TEF value (0.005). The toxicity of a congener is calculated by multiplying its TEF by the concentration found in the environment to arrive at the toxic equivalent concentration or TEQ. The total toxicity of a PCB, dioxin, or furan mixture is estimated by summing the TEQs of the congeners.

Examples of Toxic Equivalency Factors (TEFs) for mammals, birds, and fish.

| | Congeners | Mammals | Birds | Fish |
|-----------------|------------------|---------|---------|-----------|
| Dioxins | 2,3,7,8-TCDD | 1 | 1 | 1 |
| | 1,2,3,7,8-PCDD | 1 | 1 | 1 |
| | 1,2,3,4,7,8-HCDD | 0.1 | 0.05 | 0.5 |
| | OctaCDD | 0.0001 | 0.0001 | <0.0001 |
| Furans | 2,3,7,8-TCDF | 0.1 | 1 | 0.05 |
| | 1,2,3,7,8-PCDF | 0.05 | 0.1 | 0.05 |
| | 2,3,4,7,8-PCDF | 0.5 | 1 | 0.5 |
| | 1,2,3,4,7,8-HCDF | 0.1 | 0.1 | 0.1 |
| | OctaCDF | 0.0001 | 0.0001 | <0.0001 |
| Non-ortho PCBs | PCB#77 | 0.0001 | 0.05 | 0.0001 |
| | PCB#126 | 0.1 | 0.1 | 0.005 |
| Mono-ortho PCBs | PCB#105 | 0.0001 | 0.0001 | <0.000005 |
| | PCB#123 | 0.0001 | 0.00001 | <0.000005 |

Source: Van den Berg et al. 1998.

PBDEs

Polybrominated diphenyl ethers (PBDEs) are widely used today as flame retardant chemicals in consumer products. One form used in polyurethane foam has been phased out, but others are added to plastics used in furniture upholstery, carpet backings, electrical insulation, computer and TV cases, and other consumer goods. Mammalian toxicity of the 209 congeners of PBDEs is thought to increase as the number of bromine atoms in the molecule decreases (Gill et al. 2004), but further research is needed to understand the mechanisms of PBDE toxicity. The smaller molecules also bioaccumulate more readily and account for most of the environmental residues (Rayne and Ikononou 2002). PBDEs are less toxic than PCBs but are structurally similar and have similar environmental properties. They are generally soluble in fat and biomagnify within the food chain, but they are more susceptible to environmental degradation than PCBs.

PBDEs can be released into the environment during production, but the main environmental source is escape from consumer products during use and after disposal. They are broken down in surface water and sediment through microbial degradation and other metabolic processes (e.g., Gouin and Harner 2003). Laboratory studies have shown that as PBDEs with more bromine atoms break down in the environment they lose bromine atoms, thus may be transformed into more toxic, mobile, and bioaccumulative forms.

Production of PBDEs ramped up during the 1970s. Three commercial mixtures of PBDEs have been manufactured: deca-dibromodiphenyl ether (decaBDE, which accounted for about 80% of the North American market before 2004), octa-dibromodiphenyl ether (octaBDE), and penta-dibromodiphenyl ether (pentaBDE). Production of the octa- and penta- mixtures was banned by the European Union (EU) and voluntarily discontinued by the only North American manufacturer in 2004. DecaBDEs are not currently regulated and continue to be used around the world. According to the revised EU risk assessment, global production of decaBDE was 56,100 tonnes in 2001. The majority (ca. 80%) is used in the plastics and electronics industry in the manufacture of circuit boards, wire coatings, and mobile telephone equipment. The remaining 20% of world use is for textiles, upholstery, cables, and insulation materials.

PBDEs enter the human body by ingestion or inhalation. They are suspected of causing cancer, decreasing thyroid hormone levels, and disrupting endocrine systems (McDonald 2002), causing liver toxicity, immune system effects (Gill et al. 2004), hyperactivity, and reproductive effects (Kuriyama et al. 2005).

Matching the increasing production and use of PBDEs, concentrations in animal tissue and human milk have increased steeply during the last decade (Rahman et al. 2001; Hites 2004). In Canada, PBDEs have been found in human breast milk and blood, food, indoor and outdoor air, and water (Health Canada 2004a,b,c,d). A recent Health Canada study showed that Canadian women have the second highest concentrations of PBDEs in their breast milk in the world after women in the USA (Ryan 2004). PBDEs in the breast milk of Vancouver women increased almost 10-fold between 1992 and 2002, from 3 ng/g to 20 ng/g (Ryan et al. 2002). PBDE concentrations in indoor air and household dust in

Canadian (Wilford et al. 2004) and US (Stapleton et al. 2004) homes were found to be high enough to create a potential health risk to young children, whose hand-to-mouth activity exposes them to greater amounts of household dust than adults. The effect of these residues, if any, is largely unknown, but no studies have conclusively linked present PBDE concentrations in humans to health effects (Health Canada 2004a). The threshold for toxicity in animal studies is 100 to 1000 times greater than exposure estimates for the currently most-exposed humans (LaFlamme 2005).

POPs IN FISH: TO EAT OR NOT TO EAT?

Recent studies have found detectable levels of PCBs, dioxins, and some persistent pesticides in wild and farmed BC salmon (e.g., Hites et al. 2004).

In 2002, Health Canada found low levels of PCBs in fish and seafood purchased in Vancouver, with average values not exceeding 17 ng/g (Health Canada 2004c). Farmed Atlantic salmon contained the highest average level (16.8 ng/g), approximately 2.5 times the level in the wild salmon samples.

In 2003, Health Canada found PBDE concentrations in retail samples of seafood up to 5.5 ng/g (parts per billion, ppb). Farmed salmon contained an average concentration of 2.2 ng/g (Health Canada 2004c) and wild BC chinook averaged 2.9 ng/g PBDEs. Health Canada has considered the risk to human health from reported levels of PBDEs in fish and seafood, available toxicological information, and estimated dietary fish and seafood consumption. They concluded that levels found in seafood to date are not a health concern for average Canadian consumers (Health Canada 2004c), and advised consumers that the known health benefits of eating fish and other seafood outweigh the risks from present contaminant levels.

PAHs

Polycyclic aromatic hydrocarbons (PAHs) are generally thought to be the most widespread of the organic contaminants (e.g., Garrett, 2004). They may bioaccumulate to some extent, depending on the ability of different organisms to metabolize them, but do not tend to biomagnify up the food chain. Spilled oil may be the best-known source of PAH contamination, but the highest concentrations at contaminated sites have come from petroleum refining processes and coal tar production. PAHs are found in industrial liquid effluents, including those from pulp mills and aluminium smelters, and are often found in harbours.

Atmospheric deposition is thought to be the major source of PAHs in most aquatic systems (Garrett 2004). Atmospheric sources include fossil fuel combustion, slash

burning, and forest fires. Other sources include contamination from liquid petroleum products and creosote, as well as from urban activities (roads, wastewater treatment plants, urban runoff; Yunker et al. 2000). There are also natural sources, such as oil seeps, coal, plant debris, and forest fires.

PAHs can be separated into the “parent” group of compounds, such as naphthalene, and the structurally similar alkylated PAHs, such as 3-methylnaphthalene. Most alkyl PAHs appear to be at least as toxic as the parent compounds. They also persist for a longer time than their parent compounds, are less water soluble, and tend to bioaccumulate to a greater degree (Irwin et al. 1997). As petroleum products age, the percentage of alkyl PAHs increases compared to parent PAHs. Because there are natural sources for both, the levels of parent and alkyl PAHs found in the environment as a result of human activity must be evaluated against the natural background (Yunker et al. 2002).

Hazards from PAHs in the aquatic environment are often difficult to assess due to the great number of parent PAHs and alkyl PAHs of potential concern and the number of variables that can either increase or decrease the risk. In general, PAHs bioaccumulate but do not biomagnify. Concern has focused on the metabolic breakdown of benzo[a]pyrene and several other PAHs into metabolites that are far more carcinogenic and hazardous than the parent compounds (e.g., Irwin et al. 1997). Total PAHs as low as 1 mg/kg have been shown to induce tumours in brown bullhead catfish (Eisler 1987), but the significance of this finding is not known.

Mercury

Unlike other metals, mercury bioaccumulates and biomagnifies in aquatic food webs and can present a health risk to high-level consumers. Mercury occurs as elemental mercury, inorganic salts, and in organic forms. It occurs naturally in soil and rocks and is a byproduct of industrial processes, incineration, and metal smelting. Mercurial compounds are also used in scientific research and pharmaceuticals. Elemental mercury is relatively inert, but other forms are very toxic. Organic mercury compounds, such as methyl mercury, are used medically as fungicides and antibacterials, but the main source of human poisoning is through eating mercury-contaminated fish. Mercury is released from flooded soils and vegetation in reservoirs, where bacteria use the organic carbon in soils to produce methyl mercury from natural concentrations of inorganic mercury (e.g., Brigham et al. 2002). Methyl mercury biomagnifies in aquatic food webs, in invertebrates, fish, birds, and mammals (e.g., Bodaly and Fudge 1999). The highest concentrations occur in large, longer-lived animals and in species at the top of the food chain.

Mercury's primary health effects are neurological, but high exposure can also permanently damage the brain and kidneys and cause birth defects in humans (e.g., reviewed in Clarkson 1997) and wildlife (e.g., Wolfe et al. 1998). Children are particularly susceptible to its toxic effects. Methyl mercury easily passes the placental barrier, where it can damage the brain and neurological system of the fetus.

Like other heavy metals, mercury enters the marine environment from many sources, including coal-fired power stations, industrial processes, wastewater outfalls, mine effluents and discharges, atmospheric emissions, as well as natural ore deposits. Mercury contamination is of international concern, with many countries, including Canada, taking stringent measures to regulate and reduce emissions. Mercury emissions from metal smelting in Canada have been significantly reduced since 1988, as have releases from chloralkali manufacturing, pesticides, paints, and batteries. Heavily contaminated tailings are still present, however, in historic gold mining areas, where it was used in the 19th and early 20th centuries (Pacyna and Pacyna 2002). Mercury from metal smelting, coal-fired utilities and municipal waste incinerators is transported globally in the atmosphere. Best estimates suggest that human activities have about doubled or tripled the amount of mercury in the atmosphere, with the atmospheric burden increasing by about 1.5% per year (Johannessen et al. 2005a).

DDT

The once commonly used pesticide DDT (dichlorodiphenyl trichloroethane) breaks down in the environment into DDE (dichlorodiphenyl dichloroethylene)—a highly persistent and endocrine disrupting chemical. DDE is produced in most animals when the body attempts to break down and excrete DDT; it was also a contaminant in DDT formulations. DDT was used in Canada to control biting insects and agricultural and forest pests from 1947 to 1969. DDT was banned after it was discovered that metabolites of DDT (mainly DDE), bioaccumulating in tissue of predatory birds, disrupted the layering of calcium in their eggs, causing egg-shell thinning and breakage. In Canada, DDT was banned in 1972 and its use is also currently prohibited in the US and Europe. It continues to be used elsewhere in the world to control mosquitoes that carry malaria and on agricultural crops.

DDE residues from past DDT use are still measurable in soil, sediment, and wildlife. These residues continue to be augmented by atmospheric transport of DDE from other continents where DDT is still in use.

ENDOCRINE DISRUPTING SUBSTANCES

Most of the POPs discussed in this report have endocrine disrupting properties in addition to other toxic effects. Endocrine disrupting substances (EDS) mimic or block vertebrate steroid hormones by interacting with hormone receptors in the body's hormone regulatory system. Typical effects include feminization of male animals or masculinization of females (e.g., Matthiessen et al. 2000 study of fish in estuaries), eggshell thinning in birds, and disruption of vitamin A and thyroid hormone physiology in mammals. Although such effects were not suspected until research in the late 1980s began to raise questions, it is now a key area of research because of the health implications.

A major British research project on endocrine disruption in the marine environment (EDMAR) found feminization of flounder and of blenny (a small forage fish) in some estuaries (DEFRA 2003). Although the impacts of EDS have not been as well studied in BC, elevated levels of egg proteins have been found in male rockfish (West et al. 2001) and English sole (Lomax et al. 2001) from sites in Puget Sound.

The range of EDS effects on animal behaviour may be more widespread than originally thought. Recent studies on a wide range of organisms, including fish, mice, birds, and primates, found changes in social and mating behaviour, increases in hyperactivity and aggression, impaired motor skills, and reduced ability to learn (Clotfelter et al. 2004; Zala and Penn 2004).

How Contaminants Move in the Environment

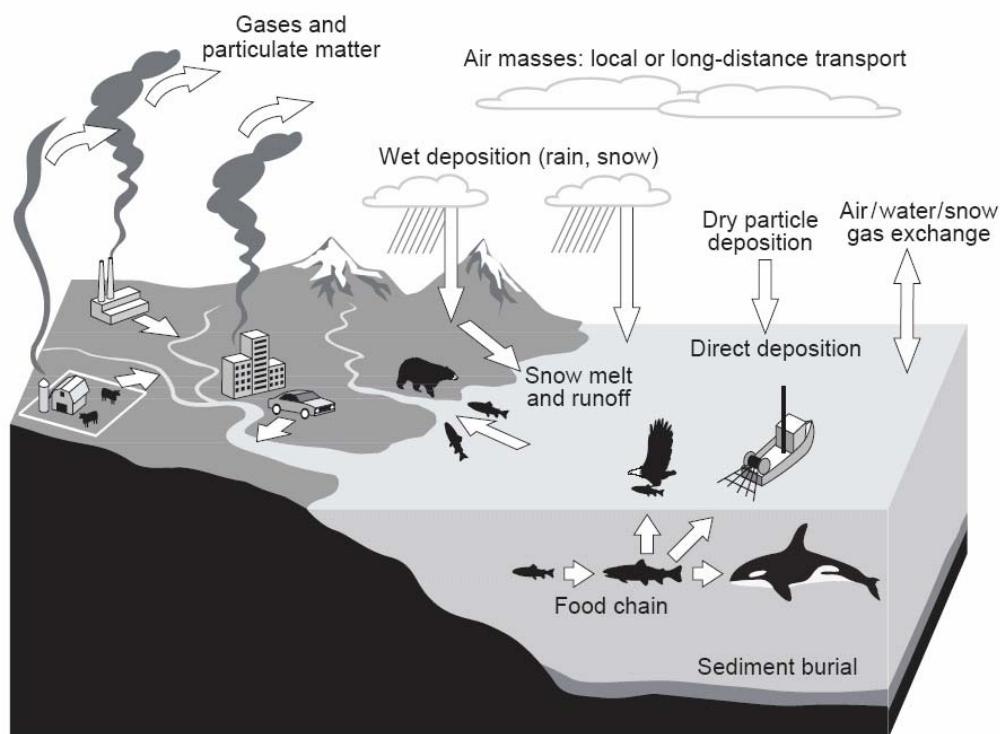
Contaminants enter the coastal environment directly and indirectly through several routes (Figure 1). They may originate from a specific outlet ("point source"), such as a discharge pipe from an industrial factory; they also may come from "non-point sources," which are the many and diffuse sources of pollutants, such as those carried off the land in urban or agriculture runoff. From such sources, they may be discharged directly into water near the shoreline or into rivers that carry the contaminants to the marine environment. POPs do not dissolve easily in water, but they can become weakly attached (adsorbed) to suspended particles in the water, which eventually carry them to the bottom. From there, contaminated sediments may be gradually covered over by new material brought down by rivers or carried along in coastal currents.

Contaminants are also emitted into the air from combustion, incineration, and industrial processes. They may also enter the atmosphere as vapours re-emitted from chemical residues already in the environment. Once in the atmosphere, POPs can travel great distances before being deposited. They may be directly deposited on land or water by precipitation or on dust particles. The vapour form of contaminants may also enter water

or snow through gas exchange processes. Once in the atmosphere, POPs travel on prevailing winds until the air masses reach cool regions, such as mountaintops and high latitudes, where the vapours condense. The chemicals may accumulate in ice or snow until they are released into ecosystems through snow melt and spring runoff (e.g., Li and Macdonald 2005). Many small lakes and reservoirs in the Rocky Mountains fed by glacial runoff contain concentrations high enough to affect wildlife at the top of the food chain (e.g., Hempel 2000).

Contaminants enter the marine food chain when they are taken up by organisms living in the water column or sediments. The contaminants attach to the lipids in the cells of organisms and bioaccumulate within organisms. Through the process of biomagnification as larger organisms eat smaller ones, the contaminants move through the food chain, becoming concentrated in top predators.

Figure 1. Direct and indirect routes for movement of POPs in the environment.



Source: Based on Ross and Birnbaum (2003), with additional sources.

TBT: HISTORY OF A MARINE CHEMICAL

Tributyltin (TBT) was widely used as the active ingredient in antifouling paints to prevent the growth of marine organisms on ships' hulls and underwater structures. It is toxic to many aquatic organisms, including fish, molluscs (especially whelks, oysters, and clams), and other benthic animals. It slows growth, causes abnormal shell development, kills larvae, and is also an endocrine disrupting substance. At extremely low concentrations (4–10 ng/L), it causes imposex (the imposition of male characteristics on females) in marine snails (EPA 2002). In some areas of BC (e.g., Burrard Inlet), TBT contamination had eradicated whelk populations by 1989, and juvenile whelks failed to mature in other areas (Bright and Ellis 1990).

Canada restricted most uses of TBT-based antifouling paints in 1989 and later banned all uses of TBT in November 2002 (PMRA 2002). In December 2004, the last US producer of TBT-based paint requested voluntary cancellation of its product registration. An international resolution has been put forward to phase out TBT-based paints with a full prohibition to be in place by 2008 (IMO 2001).

Despite Canadian prohibitions, significant concentrations of TBT are likely to remain in sediment for 20 to 30 years (Stewart and Thompson 1997; Maguire 2000). At the Ogden Point breakwater in Victoria, all female whelks sampled in 2000 showed reproductive effects caused by TBT, which was 11 years after most uses were banned (Reitsema et al. 2002). Because the ban is not yet worldwide, there will also continue to be some new TBT inputs as ships with TBT-based hull paint enter BC waters.

INDICATORS

1. Key Indicator: Trends in dioxin and furan levels in pulp and paper mill effluent, sediments, and Dungeness crab tissues

This is both a pressure and a response indicator. It shows the past pressure on the environment from dioxins and furans in pulp mill effluents and the results of the societal response by governments and industry to eliminate this pressure on the environment. The indicator addresses the questions: What is the impact of industrial pollutants released to the environment? Are efforts to protect the environment from industrial pollutants effective?

In BC, elevated dioxin and furan concentrations were first found in edible fish and shellfish collected near coastal pulp and paper mills between 1987 and 1989. After investigation showed that the chemicals were being generated as a byproduct of the pulp

bleaching process, both the federal and provincial government introduced regulations to control dioxin and furan discharges. The initial impetus for industrial change came from BC provincial regulations in 1990, which established a legally binding requirement to eliminate adsorbable organic halide (AOX)—a surrogate measure of the amount of chlorinated organic compounds in pulp and paper effluent. Federal regulations in 1992 prohibited release of dioxins and furans above certain levels, required pulp mills to conduct annual monitoring, and closed crab and shellfish fisheries in affected areas. This indicator shows the effect of regulatory measures and changes in mill technology as industry complied.

Methodology and Data

Data for this indicator were collected under two federal programs:

- The Pulp and Paper Mill Effluent Chlorinated Dioxins and Furans Regulations (1992), of the Canadian Environmental Protection Act. The regulations prohibited the release of 2,3,7,8-TCDD in pulp and paper mill effluents above 15 parts per quadrillion (ppq) and of 2,3,7,8-TCDF above 50 ppq by 1 January 1994.
- The Coastal Mills Dioxin and Furan Trend Monitoring Program. This federal program requires mills on the BC coast to monitor dioxins and furans in the environment around their effluent outfalls. Environment Canada specifies the sampling locations, species, and numbers of samples to be collected annually. Data are reviewed by Health Canada, which conducts a human health risk assessment for consumption of Dungeness crab. Fisheries and Oceans Canada reviews the data and health risk assessment to evaluate the need for fisheries closures and advisories.

Sampling Methods

Mill effluent: Under the Pulp and Paper Mills Chlorinated Dioxins and Furans Regulations, mill operators are required to collect 24-hour composite samples of their final effluent and determine the concentrations of all toxic congeners of dioxins and furans. Data for this indicator came from samples collected at least once a year, more frequently if the mill found a measurable concentration of dioxins or furans. Samples were analyzed with high-resolution gas chromatography/high-resolution mass spectrometry for 27 different dioxin and furan congeners (Environment Canada 1992).

In 1991, nine pulp and paper mills discharged secondary-treated effluent to BC's coastal waters and were included in the monitoring program. Since then, some mills have closed and others have switched to bleaching technology that does not use elemental chlorine. By 2002, six mills and by 2004 only three mills required annual monitoring.

Sediment: Three sediment samples were taken annually at the same location close to each mill outfall. A composite of the top 2 cm of the three samples was analyzed similarly to the mill effluent. TEQs were derived using internationally accepted procedures (see text box: Toxic Equivalents).

Crab tissue: Adult Dungeness crabs (*Cancer magister*) concentrate POPs in their fat-rich digestive gland (hepatopancreas). This was chosen as an indicator species because the crabs do not move large distances and can therefore be used to indicate local levels of contamination. They have a relatively long life span and are widely distributed, preferring sandy-bottomed waters where contaminated sediments often accumulate. The crabs are also economically important because they are fished commercially and recreationally.

Dungeness crabs collected for analysis were legal-sized males (minimum carapace width, 165 mm). They were collected from sites near mill outfalls, usually in March. The hepatopancreas samples of up to seven crabs collected near each mill outfall were combined into a single sample. Dioxin and furan analyses were conducted using ultra-trace high-resolution gas chromatography/high-resolution mass spectrometry. TEQs were derived using the internationally accepted procedure (Van den Berg et al. 1998).

Results from all three types of sampling are reported in Tables 1 to 3 and displayed graphically in Figure 2.

Table 1. Effluent loadings of the dioxin 2,3,7,8-TCDD from coastal BC pulp and paper mills, 1989 to 2003 (mg per day).

| Mill | 1989 | 1990 | 1991 | 1992 | 1993 | 1994 | 1995 | 1996 | 1997 | 1998 | 1999 | 2000 | 2001 | 2002 | 2003 |
|------------------------|-------|-------|-------|-------|------|--------------------|------|------|------|--------|-------|-------|---------------|-------------------|-------|
| Crofton | 6.32 | 0.99 | 2.60 | 0.28 | 0.31 | 0.40 | 0.31 | 0.28 | 0.20 | 0.15 | 0.16 | ND | ND | ND | – |
| Elk Falls | 5.82 | 6.22 | 1.16 | 0.65 | 1.13 | 0.57 | 0.33 | 0.39 | 0.29 | 0.21 | 0.20 | ND | ND | ND | ND |
| Gold River | – | – | 1.50 | 0.12 | 0.55 | 0.50 | 0.77 | 0.50 | 0.27 | Closed | | | | | |
| Harmac | 6.22 | – | 0.84 | 2.61 | 0.40 | 0.32 | 0.32 | 0.33 | 0.60 | 0.56 | 0.14 | ND | ND | ND | – |
| Port Alberni | – | – | 0.65 | 0.37 | 0.58 | No chlorine in use | | | | | | | | | |
| Port Mellon | 19.29 | 1.91 | 0.31 | 0.16 | 0.13 | 0.12 | 0.16 | 0.16 | 0.11 | 0.14 | 0.08 | ND | ND | ND | ND |
| Powell River | – | 4.51 | 3.57 | 0.33 | 0.55 | 0.80 | 0.21 | 0.30 | 0.38 | 0.37 | 0.26 | ND | ND | Kraft mill closed | |
| Skeena | 9.77 | 2.79 | 3.31 | 7.44 | 0.76 | 0.53 | 0.41 | 0.37 | 0.18 | 0.30 | 0.07 | ND | Not operating | | |
| Squamish | 3.46 | 1.60 | 0.70 | 0.46 | 0.50 | 0.17 | 0.20 | 0.15 | 0.14 | 0.10 | 0.10 | ND | ND | ND | ND |
| No. of mills reporting | 6 | 6 | 9 | 9 | 9 | 8 | 8 | 8 | 8 | 7 | 7 | 7 | 6 | 5 | 5 |
| Total | 50.88 | 18.02 | 14.64 | 12.42 | 4.91 | 3.41 | 2.71 | 2.48 | 2.17 | 1.83 | 1.01* | 1.00* | 1.00* | 1.00* | 1.00* |

Source: Pollution Prevention and Assessment, Environmental Protection, Environment Canada 2005.
Coastal Mills Dioxin and Furan Trend Monitoring Program
www.ecoinfo.ec.gc.ca/env_ind/region/dioxinfuran/dioxin_data_e.cfm#Graph1.

* Individual loadings not calculated for each mill, but a total dioxin loading of 1.00 mg/day was estimated.

Note: Six mills reported voluntarily before 1991; regulations were not in effect.

ND = Non-detectable 2,3,7,8-TCDD in the effluent.

Table 2. Dioxin and furan concentrations in sediment near coastal BC pulp and paper mill outfalls, 1989–2003 (TEQ*, pg/g).

| Mill area | 1989 | 1990 | 1991 | 1992 | 1993 | 1994 | 1995 | 1996 | 1997 | 1998 | 1999 | 2000 | 2001 | 2002 | 2003 |
|-----------|------|-------|------|-------|------|------|-------|------|------|------|------|------|------|------|------|
| Crofton | – | 237.2 | 16.1 | 136.3 | 27.6 | 154 | 157.2 | 86.6 | 95.1 | 71.1 | 75.8 | 60.6 | 68.3 | 61.7 | 12.6 |

| | | | | | | | | | | | | | | | |
|------------------------|-------|-------|-------|-------|-------|-------|-------|-------|-------|-------|-------|------|-------|-------|------|
| Elk Falls | – | 10.8 | 10.9 | 18.9 | 1.7 | 9.7 | 0.3 | – | – | – | 4.6 | 5.3 | 15.7 | 16.3 | 15.3 |
| Gold River | – | 972 | 503.3 | 584.8 | 117.5 | 135.6 | 112.8 | 305.1 | 373.1 | 337.4 | – | – | – | – | – |
| Harmac | – | 380.7 | 209.7 | 260.8 | 310.8 | 258.6 | 392.7 | 216.3 | 216.9 | 246.6 | 108.4 | – | – | – | – |
| Port Alberni | – | 76.6 | 106.4 | 36.4 | 35.7 | 26.8 | 23.9 | 28 | 128.6 | – | – | – | – | – | – |
| Port Mellon | 246.2 | 197.4 | 198.5 | 57.4 | 47.4 | 109.4 | 117.6 | 84.2 | 95.4 | 34.3 | 142.7 | 20.9 | 158.9 | 115.8 | 82.4 |
| Powell River | – | 55.1 | 69.3 | 41.5 | 110.9 | 33.4 | 66.4 | 31.5 | 19.5 | 22.8 | – | – | – | – | – |
| Skeena | 45.3 | – | 92.6 | 45.1 | 14.1 | 16.2 | 21.4 | 17.3 | 23.4 | 14.3 | 16.2 | 20.8 | 11.1 | 9.3 | – |
| Squamish | 84.6 | 89 | 19.3 | 1.9 | 1 | 4.6 | 2.8 | 2.7 | – | – | – | – | – | – | – |
| No. of mills reporting | 3 | 8 | 9 | 9 | 9 | 9 | 9 | 8 | 7 | 6 | 4 | 4 | 4 | 4 | 3 |
| Average | 125 | 252 | 136 | 131 | 74.1 | 83.1 | 99.5 | 96.5 | 136 | 121 | 69.5 | 26.9 | 63.5 | 50.8 | 36.8 |

Source: Pollution Prevention and Assessment, Environmental Protection, Environment Canada 2005.

* TEQ = Toxic equivalent concentration; used to standardize the total concentrations of the different dioxins and furan congeners to 2,3,7,8-TCDD equivalents.

Table 3. Dioxin and furan concentrations in the hepatopancreas of Dungeness crabs sampled near BC pulp and paper mills, 1989–2003 (TEQ pg/g).

| Mill area | 1989 | 1990 | 1991 | 1992 | 1993 | 1994 | 1995 | 1996 | 1997 | 1998 | 1999 | 2000 | 2001 | 2002 | 2003 |
|--------------------------|------|-------|-------|-------|------|------|------|------|------|------|---------------------------|------|------|------|------|
| Crofton | – | 179.1 | 82.4 | 130.2 | 97.1 | 49.3 | 57.1 | 57.4 | 29.9 | 48.5 | 49.7 | 34.3 | 59.3 | 34.9 | 28.2 |
| Elk Falls | – | 104 | 116.3 | 103.8 | 68.5 | 63.3 | 28.1 | 21.3 | 14.5 | 26.1 | 22.1 | 17.6 | 21.8 | 24.2 | 30.1 |
| Gold River | – | 1075 | 510 | 178 | 59.8 | 46 | 17.6 | 30.7 | 19.4 | 45.6 | – | – | – | – | – |
| Harmac | – | 228.7 | 164.1 | 92.2 | 22.7 | 8.2 | 4.6 | 15.9 | 8.9 | 5.8 | – | – | – | – | – |
| Port Alberni | – | 16.2 | 41.8 | 52.7 | 20.2 | 26.5 | 57.2 | 26.9 | 17.3 | – | – | – | – | – | – |
| Port Mellon ^a | 1490 | 482 | 196 | 79.3 | 143 | 35.8 | 67.9 | 20.9 | 16.5 | 12.8 | 52.2 | 27.2 | 35.4 | 20.1 | 12.3 |
| Powell River | 490 | 116 | 78.2 | 140.2 | 87.5 | 2.2 | 8.5 | 25.2 | 14.2 | 11.4 | – | – | – | – | – |
| Skeena | 617 | 306 | 344.1 | 142.4 | 30.8 | 30.4 | 34.8 | 37 | 25.9 | 31.9 | 57.2 | 40.0 | 38.9 | 33.5 | 27.3 |
| Squamish ^a | 343 | 67.2 | 113.2 | 48.6 | 24.6 | 15.6 | 19.3 | 12.5 | 44.6 | 12.1 | Reported with Port Mellon | | | | |
| No. of mills reporting | 4 | 9 | 9 | 9 | 9 | 9 | 9 | 9 | 9 | 8 | 4 | 4 | 4 | 4 | 4 |
| Average | 735 | 286 | 183 | 107.5 | 61.6 | 30.8 | 32.8 | 27.5 | 21.2 | 24.3 | 45.3 | 29.8 | 38.8 | 28.2 | 24.5 |

Source: Pollution Prevention and Assessment, Environmental Protection, Environment Canada 2005.

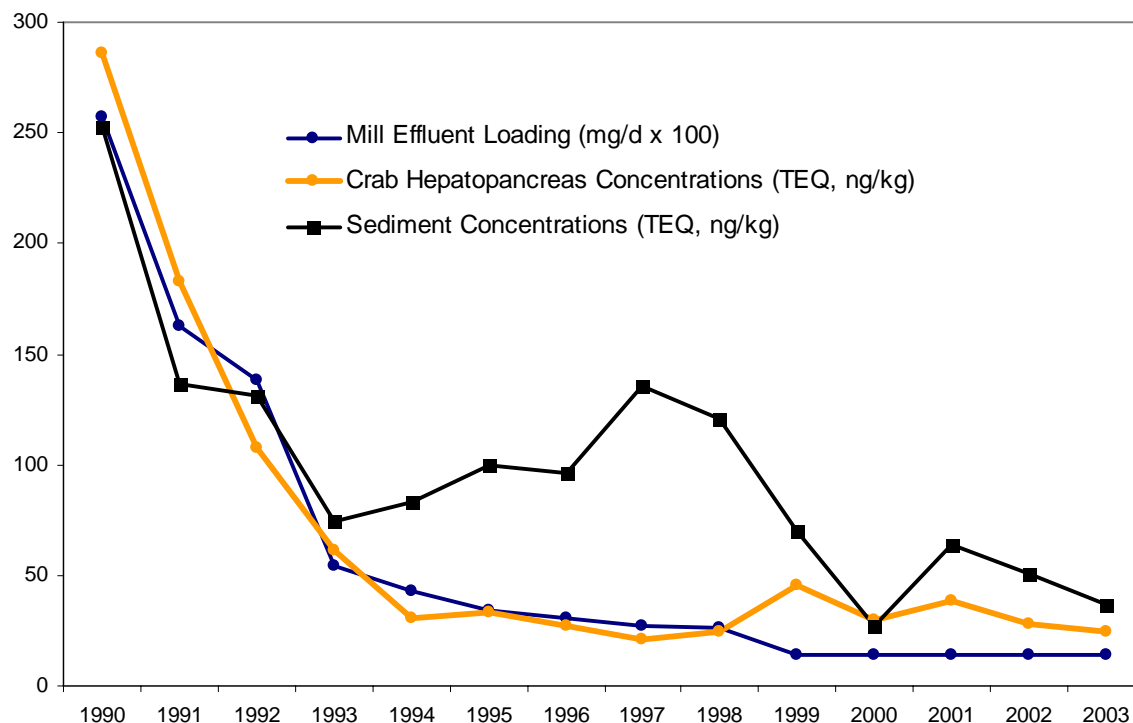
Note: All results are reported in TEQ (toxic equivalent concentration) to standardize total concentrations of dioxins and furan congeners to 2,3,7,8-TCDD equivalents.

^a Monitoring at the Port Mellon and Squamish mills was combined in 1999 because 2,3,7,8 TCDD was not detected in their effluents and the mills are close together in Howe Sound; after 1996, results for both are reported under Port Mellon.

From 1989 to 2003, dioxin and furan concentrations (TEQ, pg/g) in the sediment near the nine different outfall sites dropped an average of 76% (Table 2). Contamination in the sediment dropped from an average TEQ of 252 pg/g (parts per trillion) in 1990 to 36.8 pg/g in 2003, an overall decline of 85%.

Dioxin and furan concentrations in the hepatopancreas of Dungeness crab declined by 71–97% near the mills that were still reporting data in 2003 (Table 3). Levels went from an average TEQ of 735 pg/g in 1989 to 24.5 pg/g in 2003—an average drop of 96%. Since 1996, the average TEQ in Dungeness crabs near pulp mills has remained near or below the established 30 pg/g guideline for tolerable daily intake.

Figure 2. Total 2,3,7,8-TCDD loadings in pulp mill effluents from all mills reporting at each date, and dioxin and furan concentrations in sediments near the outfall and in crab hepatopancreas, 1990–2003.



Source: Data from Tables 1, 2, and 3).

Note: Number of mills reporting: 9 mills for 1990-1997; 8 mills in 1998; 4 mills for 1999-2003.

Figure 2 displays data from Tables 1 to 3 and illustrates a relationship between the steep decline in dioxin concentrations in mill effluents and the improvement in concentrations in local sediment and crab tissue.

Interpretation

In 1989, the pulp and paper industry began to invest in changes to the bleaching process in response to the concern over dioxin and furan loading in the environment and to address upcoming federal and provincial regulations. Changes in technology included using uncontaminated wood chips and de-foaming agents, new pulp bleaching and washing practices, oxygen-based pulp cooking, and installation of new treatment works. As a result, between 1989 and 1999, there was a 95% drop in total daily loading from all

coastal BC pulp and paper mills for 2,3,7,8-TCDD in effluents. After 1999, 2,3,7,8-TCDD was not detectable in the effluent of any mills. At the same time, total daily loadings for the furan 2,3,7,8-TCDF also declined by more than 99% by 2004. Both substances are now present only in minute quantities that fall within federal discharge limits.

The improvement in effluent quality was reflected in a rapid decline in levels of dioxins and furans in the environment (sediments) and local organisms (crabs). Contamination in crabs near outfalls closely tracked the drop in concentrations in effluent, with a decline of 96% from 1989 to 2003. Contamination in the sediment dropped by more than 85% from 1990 to 2003. The decline was not as rapid as in crab tissue, in part because dioxins and furans have a strong affinity for sediments and break down very slowly. Although burial under clean sediment helps to reduce the availability of contaminants, currents, tidal motion, and marine organisms continue to disturb the upper sediment, bringing contaminants to the surface where they can recirculate in the ecosystem.

This indicator shows that coastal BC pulp mills no longer discharge detectable levels of toxic dioxins to marine waters and shows that measures taken to eliminate this source of persistent contaminants in the marine environment have been effective. The continuing low, relatively stable, levels of dioxins and furans in sediments and crab tissue also shows how persistent these chemicals are. It may also indicate that there is a continuing low level of input of dioxins and furans from other sources (e.g., regional incinerators and other local combustion, global atmospheric transport). Given their persistence, low levels of dioxins and furans will remain in the environment for many years.

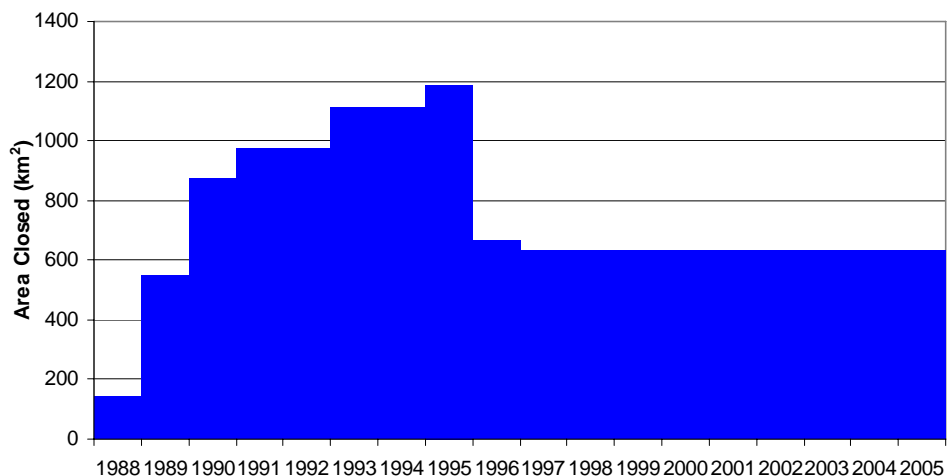
Supplementary Information: Impact of Dioxin and Furan Contamination on Shellfish Harvesting

Fisheries and Oceans Canada closed several areas around pulp mills to shellfish harvesting in 1988 when the health risk from dioxin and furan contamination became known. As monitoring expanded to include more mills, more affected areas were found. By February 1995, nearly 1200 km² of BC coastal waters had been closed (Figure 3). The total area closed to shellfish harvesting near eight BC coastal pulp and paper mill locations peaked in 1995.

The sharp decrease in effluent loadings from pulp mills in the early 1990s was followed by a rapid decline in dioxin and furan concentrations in sediments and crab tissue. This made it possible to lift restrictions for all shrimp, prawn, and oyster harvesting in 1995 and later reopen some areas to crab harvesting. More than 46% (550 km²) of the maximum area closed in 1995 had been reopened by the end of 1997. No restrictions have been lifted since, in part because Health Canada is currently reassessing the safe consumption limits for dioxins and furans.

The remaining shellfish closures prohibit crab harvesting only. The full economic impact of these closures is not known, but the Dungeness crab fishery ranks as one of the most valuable invertebrate fisheries in BC. In 2004, the wholesale value was \$67.9 million.

Figure 3. Areas closed to harvesting for crabs, shrimp, oysters, and prawns near eight coastal BC pulp and paper mills, 1988–2005. After 1995, all closures were for crab harvesting only.



Source: W. Knapp, Department of Fisheries and Oceans, Vancouver, 2005.

2. Secondary Indicator: Long-term trends in deposition of pollutants (PCBs, dioxins and furans, PAHs, and mercury) in sediments on the BC coast

This is a pressure and a response indicator. It shows the pressure on the environment from contaminants deposited in sediments, both historically and currently. It also shows the combined effects of industrial, technological, and regulatory changes that have been made to curb contamination from these pollutants.

The indicator addresses the questions: What industrial pollutants are humans releasing into the environment? Are measures to protect the environment from industrial pollutants effective?

Contaminants discussed in this indicator attach strongly to particles and are deposited with the particles as they settle to the bottom. Contaminants can be deposited in sediments close to the discharge point or they may settle after being moved by marine currents (typically over short distances). They may also have been transported in the atmosphere over longer, even global, distances before being deposited in the marine environment.

Methodology and Data

Data from the following sources are presented on the map in Figure 4.

- The surface sediment data came from studies conducted in BC by several investigators between 1993 and 2003 and summarized in a separate review (Burd 2005).
- Data for 17 sediment cores in the Strait of Georgia came from Johannessen et al. (2005a).
- Supplementary unpublished data for PAHs and mercury (e.g., Esquimalt Harbour) came from Environment Canada's Disposal at Sea Program.

The analytical methods used in the above studies were comparable and are expected to produce comparable results for PCBs, dioxins and furans, PAHs, and mercury.

Sediment Sampling

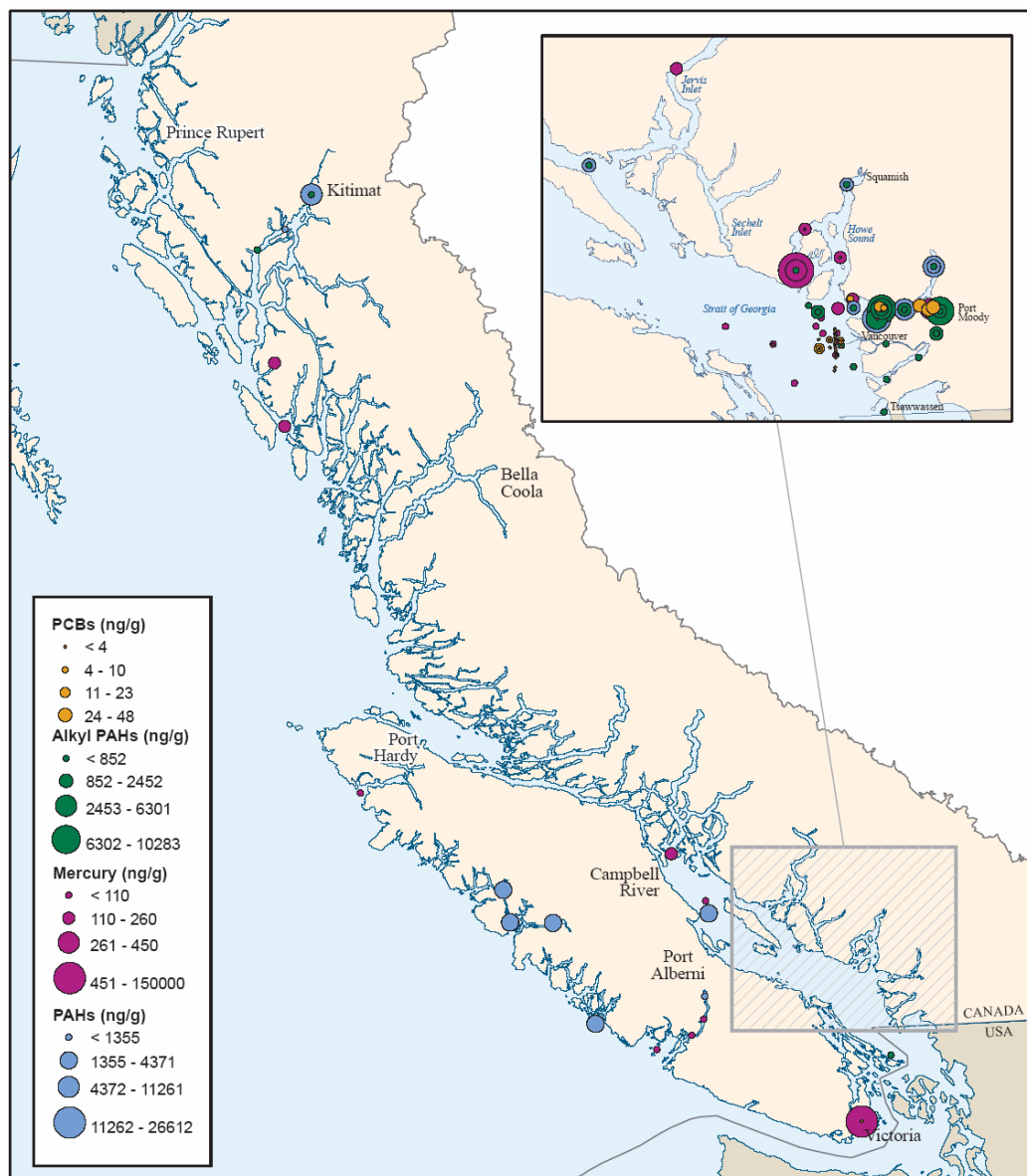
- Grab samplers are used to collect surface sediments. Common models have a set of jaws that shut when lowered into the surface sediments. They usually cover a sampling area of about 0.1 m². Chemical analysis is usually done on the top 2 cm of sediment collected. Grabs are most useful for determining the horizontal distribution of pollutants.

Analysis of surface sediment can show spatial variations in sediment contamination but not an accurate time line of historical deposits, even when samples from various dates are compared. This is because organisms living in the sediment mix the top layers to a depth of about 10 cm. Surface grab samples generally represent the upper mixed layer, which, in areas with low deposition rates, can include material from 20 or more years mixed together.

- Core samplers are used to collect a deeper profile of sediments. These samplers penetrate the sediment to a depth of several metres, depending on the sediment characteristics, and provide an undisturbed cross-section. Cores provide more reliable data on the history of contaminant deposition than grab samplers because mixing in the upper layer can be resolved through mathematical modelling of the total profile. Such models typically fit concentration data from the vertical profiles to the measured sedimentation rate for each core. This produces an estimate of the deposition rate or the amount of contaminant deposited per unit of bottom area. The model can then be used to calculate a more accurate estimate of the recent deposition in terms of concentration.

For analysis, sediment cores were sectioned into 1-cm intervals for the top 10 cm, 2-cm intervals for the next 10 cm, and 5-cm intervals for the rest of the core (Johannessen et al. 2005a). The deposition history reflected by the cores ranges from less than 50 years (a site with a high rate of sediment deposition) to more than 140 years (to the 1860s) for sites with lower sedimentation rates. The compiled analyses are portrayed in Figure 4.

Figure 4. Levels of PCBs, PAHs, alkyl-PAHs, and mercury in sediments on the eastern side of the Strait of Georgia [inset] and in other locations on the BC coast.



Source: Burd (2005); Johannessen et al. (2005a,b); Environment Canada's Disposal at Sea program (unpubl. data).

Note: Sites in the classes with the highest level of mercury (>451 ng/g) and highest total PAHs (>11,262 ng/g) exceed levels considered contaminated for pristine (sensitive) sites; some sites in this class also have levels that would not be acceptable in a working harbour, which has higher criteria for contaminated sediments (c.f., BC Environmental Management Act, Contaminated Sites Regulation, Sched. 9, BC Reg. 375/96).

Interpretation

The heaviest industrialization in BC is on the eastern side of the Strait of Georgia, as shown in Figure 4. Sources of contaminants to the Strait currently and historically include pulp and paper mills, municipal and industrial effluents, petroleum refineries, acid mine

drainage and mine tailings disposal, aquaculture, agricultural and urban runoff, wood treatment facilities, ships, spills, local combustion, and atmospheric transport from other parts of the globe. Industrial effluents and runoff from agriculture, forestry, and urban areas are also carried into the Strait by rivers. Once in the Strait, depending on their physical properties, contaminants may remain in the water, accumulate in sediments, and/or be taken up by living organisms. Those that remain dissolved are probably flushed out of the Strait within a year (Waldichuk 1983), which helps to prevent their accumulation. Contaminants discussed in this paper have physical properties that favour attachment to sediment particles in water. They end up primarily in sediments and organisms, and thus remain within the local water basin (Johannessen et al. 2005b).

Sediment from the Fraser River is a major influence in the southern Strait. Particles settle and then move northward along the bottom, perhaps driven by the deep, northward return flow of waters that are part of estuary circulation patterns (Johannessen et al. 2005b). Contaminant deposition in sediments within the southern Strait of Georgia is generally higher than it is farther north. This is because sediments accumulate at a higher rate in deep waters of the southern Strait (2.8 cm per year) than at the northern end (0.28 cm per year). Sedimentation rates in the inlets and channels of the Strait and its shallower coastal waters lie between these two extremes (Johannessen et al. 2005a).

PCBs

Sediments are a permanent sink for PCBs because they largely remain in place, with minor degradation, unless the sediment is disturbed (Macdonald and Crecelius 1994). The beginning of commercial production of PCBs in North America in 1929 shows up in sediment cores from the Strait of Georgia. The sediments show a rapid increase in PCB levels starting in the horizon for 1930-40. Concentrations increased to a peak of around 30 ng/g in the 1960s, declining sharply to an average of about 5 ng/g after 1970. This is consistent with the decreasing PCB release to the environment as a result of regulatory controls after 1970.

PAHs and alkyl PAHs

The background level of PAH contamination in sediments in the early 1900s was less than 1 ng/g (ppb). This low level continued through the 1980s in the deep basin of the Strait of Georgia (Macdonald and Crecelius 1994), but had reached 1397 ng/g in the surface mixed layer at the Point Grey disposal site by 2000 (Wilson and McKinnon 2003). Parent and alkyl PAH concentrations in sediments now exceed provincial or interim federal guidelines at most sampling locations in the Fraser River estuary and Vancouver Harbour, as well as in parts of the southeastern Strait of Georgia. Vancouver Harbour has the highest concentrations (Yunker and Macdonald 2003a,b). Analyses show that PAHs in the southeastern Strait of Georgia came primarily from the Main and North arms of the Fraser River, and from sediment dredged from Vancouver Harbour (Yunker 2000). As shown on Figure 4, some of the total PAH levels are above 10,000 ng/g, which is the criterion for sensitive estuarine sediments as defined in the BC Environmental Management Act (Contaminated Sites Regulation, Schedule 9, BC Reg. 375/96). Some

are also above 20,000 ng/g, which is the contaminated sites criterion for typical marine sediments, such as in a working harbour.

Mercury

The Fraser River is the main source of mercury (about 75%) deposited in the Strait of Georgia. Local, “point” sources, such as a former chlor-alkali plant, pulp mills, mine tailings, shipyards, and municipal outfalls are also important in smaller geographic areas.

The pre-industrial, background concentration of mercury in the Strait appears to have been 60 ng/g (ppb) or less (Johannessen et al. 2005a). By 1900, Vancouver Harbour had a mercury concentration of 300 ng/g. The sediment record shows peaks in mercury contamination from placer gold mining along the Fraser River in the 1860s, from mercury mining in the Fraser basin during World War II, from a chlor-alkali plant at the head of Howe Sound (1965-1970), and from bleaching processes in pulp mills around the Strait before chlorine was introduced in the 1960s. In addition, a mercury contamination event from an unknown source deposited sediment containing 230,000 ng/g in Port Moody Arm around 1970. This was enough mercury to poison the organisms on the bottom and halt biological mixing of sediments for about 20 years (Johannessen et al. 2005a).

Recent measurements from a variety of studies suggest that surface mercury concentrations are lowest in the central Strait and near the mouth of the Fraser River (10–154 ng/g), and highest in Port Moody Arm (77–420 ng/g), Burrard Inlet (230–250 ng/g), False Creek (399 ng/g), and Howe Sound (150–500 ng/g). Concentrations of 180–31,800 ng/g have been found in Esquimalt Harbour, with Victoria Harbour showing up to 2760 ng/g, reflecting a variety of industrial sources (Garrett 2004). These concentrations range from well below, to well above, the effects threshold used by CCME, which is 130 ng/g for mercury (0.13 mg/kg) (CCME 1999). To put these concentrations in context, the criteria for contaminated sites under BC Environmental Management Act is 430 ng/g of mercury for sensitive estuarine sediments and 840 ng/g for typical harbour sediments. With respect to Georgia Strait sediments, Johannessen et al. (2005a) stated that it is likely that mercury concentrations will remain above the threshold for effects on benthic organisms for many years.

Supplementary Information: Mercury Deposition Rates in the Strait of Georgia

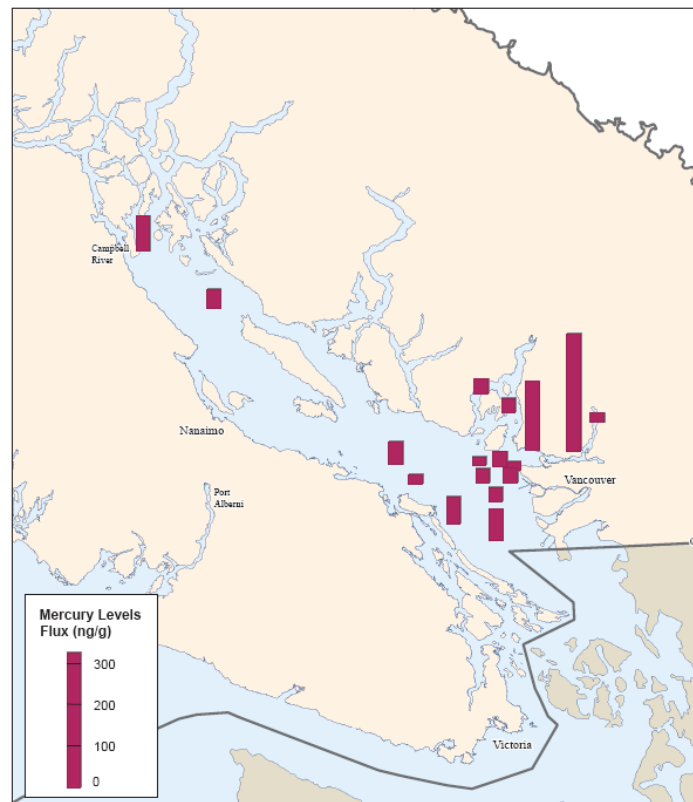
The interpretation of sediment data provides no indication that mercury contamination is declining in marine sediments, although Johannessen et al. (2005a) recently found a trend toward lower deposition of mercury throughout the Strait of Georgia.

Geographic differences in local sedimentation and mixing rates mean that the spatial variation in mercury concentrations (Figure 4) does not directly reflect the rate at which mercury is deposited (Figure 5). In areas where the sedimentation rate is high, dilution by clean sediment acts to lower the measured concentration of mercury. In areas where the sedimentation rate is low, continued mixing of the upper sediment by marine organisms keeps the mercury concentration higher at the surface, even though there may no longer

be local sources. In particular, the concentration of mercury near the mouth of the Fraser River is low because the mercury carried by the river is diluted by a large volume of other, uncontaminated particles (Johannessen et al. 2005a).

Although the Fraser River is the main present-day source of mercury to the southern Strait of Georgia, contributing about 75% of the total mass deposited annually, the situation is reversed in Vancouver Harbour. There, industrialization and other human activities account for at least 75% of the annual deposition (Johannessen et al. 2005a). Direct atmospheric deposition to the Strait of Georgia is currently only 2% of the quantity annually deposited by the Fraser River, but this figure may increase. Mercury can be transported long distances in the atmosphere and some of the atmospheric deposition originates with the combustion of fossil fuels, particularly coal. The growing demand for energy in Asia threatens to increase the concentration in the atmosphere over BC, which lies downwind (Pacyna and Pacyna 2002). This would increase the amount of mercury deposited directly to BC's coastal waters from the atmosphere and indirectly by the Fraser River.

Figure 5. Modelling of recent mercury deposition rates in the Strait of Georgia.



Source: Johannessen et al. 2005a. (Flux = amount of mercury flowing over a given area per unit time).

3. Secondary Indicator: Clean-up of contaminated sites in BC

This is a response indicator showing a societal response to the problem of industrial contamination. It addresses the question: What is being done about environmental contamination?

An area is considered contaminated if the site is unsuitable for specified land or water uses. Such sites may have become contaminated through spills or through deposits of chemicals during the course of commercial and industrial activity. At some locations, toxic substances in soil, surface water, and groundwater are a threat to the environment and human health. Contaminated sites can release toxic substances into the surrounding environment, infiltrating the food chain, entering the ground water, and contaminating neighbouring areas. Many sites were contaminated by past activities, up to a century or more ago, before the impact of such activities was known. Some land around Victoria's inner harbour, for example, was contaminated with metals and PAHs by a coal gasification plant in the late 1800s.

The contamination affects both land and water, and the size of contaminated sites ranges from less than a hectare to several square kilometres. The largest single source of site contamination in BC has been activities related to fossil fuels and vehicles, such as petroleum and natural gas storage and distribution, and vehicle salvage and wrecking. Heavy metals such as lead, arsenic, cadmium, and mercury are also common at contaminated sites in BC. Organic chemicals, including benzene and toluene from gasoline, occur at about two-thirds of the sites. Chlorophenols are found where wood treatment operations took place, as are benzo[a]pyrene and naphthalene from creosote. PCBs often occur at sites where heavy electrical equipment was used.

With the exception of federally managed sites, the BC Ministry of Environment is responsible for managing contaminated sites in the province and ensuring that site managers meet cleanup requirements to restore damaged lands to standards set out in the BC Environmental Management Act. These include requirements for protecting human health and improving ecosystem health by reducing or eliminating toxic materials and performing other activities to return land to a condition suitable for more general use.

Methodology and Data

The BC Ministry of Environment has been collecting information on contaminated sites since 1988. The Contaminated Sites Registry was created in 1997 to provide public access to current information on which sites are, or were, contaminated and the status of the clean-up or remediation on the site.

www.env.gov.bc.ca/epd/epdpa/contam_sites/sites_info/index.html.

Note that not all of the sites in the Registry are necessarily contaminated. Some were found to be clean after they were registered, others were contaminated at one time but were remediated before being entered in the Registry. The Registry also contains sites

awaiting complete assessment, for which the degree of contamination has not been determined.

Not all contaminated sites in the province have been submitted to the Sites Registry. Local governments, for example, can opt out of the registry program but must register a site if remediation is under way or if it is being decommissioned. Contaminated sites under federal jurisdiction (e.g., on federal Crown land) are also managed separately, but 268 of the 320 federally managed sites in BC (as of April 2005) are included in the provincial Sites Registry.

For this indicator, the set of all contaminated sites in the Ministry's three coastal regions (Vancouver Island, Lower Mainland, Skeena) was obtained from the Registry records. The proportion of the total sites was determined for which remediation is complete, in process, or for which a complete assessment is pending (Table 4 and Figure 6).

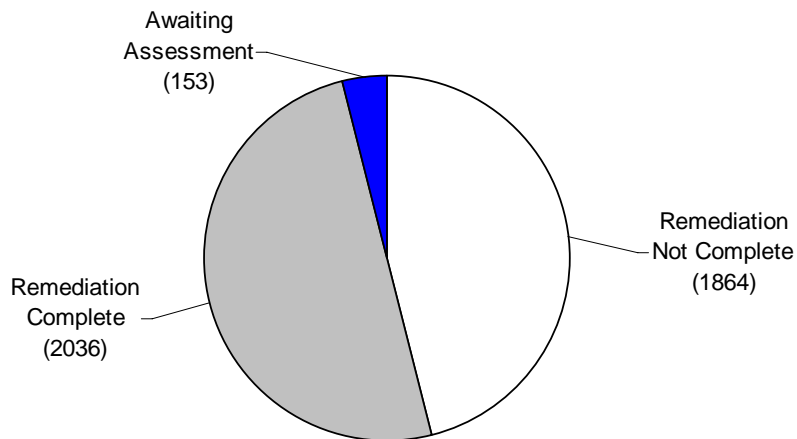
As of April 2005, 6578 sites were registered in the whole province, of which 4053 (61.6%) were in the Ministry's three coastal regions (Table 4). More information on provincial contaminated sites regulation is available at www.env.gov.bc.ca/epd/epdpa/contam_sites/.

Table 4. Number of sites in three coastal regions listed in the BC Ministry of Environment Sites Registry by remediation status (as of April 2005).

| | Vancouver Island | Lower Mainland | Skeena | Total |
|------------------------------|------------------|----------------|--------|-------|
| Remediation complete | 446 | 1517 | 73 | 2036 |
| Remediation not complete | 501 | 1101 | 262 | 1864 |
| Awaiting complete assessment | 29 | 74 | 50 | 153 |
| Total | 976 | 2692 | 385 | 4053 |

Source: BC Environment, Contaminated Sites Registry, April 2005.

Figure 6. Remediation status of contaminated sites in the coast region of BC.



Source: Data from BC Ministry of Environment Contaminated Sites Registry.

Interpretation

A total of 4053 contaminated sites have been registered in the Ministry of Environment's three coastal regions (Vancouver Island, Lower Mainland, Skeena). There is no way to know how many, as yet unregistered, contaminated sites exist in the province; however, since the registry started, 2036 sites in the coastal region have been remediated, and almost that number are in the process. In recent years, about 200 sites per year have been remediated.

Most coastal contaminated sites are located in highly populated areas: 66% in the Lower Mainland, 24% in Vancouver Island Region, and 10% in the Skeena Region. Of these sites, cleanup has been completed at 50%, and is under way at a further 46%. The remaining 4% of sites are awaiting assessment, and the degree of contamination and need for remediation is not known. In total, 96% of the contaminated sites in BC's coastal regions that were registered in the Sites Registry have been or are in the process of meeting cleanup requirements under the BC Environmental Management Act. In comparison, about 85% of the known contaminated sites in the rest of the province have been cleaned up or are in the process of remediation.

4. Key Indicator: Persistent organic pollutants in tissues of marine mammals on the BC coast

This is a pressure indicator showing the accumulation of persistent contaminants in the body tissue of whales, seals, and other organisms. It addresses the question: What is the extent of contamination from persistent organic pollutants (POPs) in the coastal environment?

POPs enter the marine food chain when organisms at the bottom of the food web, such as plankton, accumulate the contaminants from water, sediment, and food. Bottom-dwelling species with free-swimming larval stages can act like a biological "pump," moving POPs from the sediment into the water column above, where the larvae are eaten by fish. POPs are fat soluble and persistent; therefore, through biomagnification, the tissue contaminants become more concentrated as they move up the food chain to seals and killer whales (Ross et al. 2000, 2004).

Marine mammals can serve as indicator species for the presence and extent of environmental contamination because they feed high on the marine food chain. The level of contaminants in their bodies may also indicate risks to human health, especially for people who consume large amounts of fish and shellfish (Ross and Birnbaum 2003).

This indicator reports on levels of POPs in three groups of killer whales (*Orcinus orca*) inhabiting the BC coast (transients and the southern and northern populations of resident killer whales) and on harbour seals (*Phoca vitulina*). The transient whales are predators on other marine mammals, whereas both resident whale populations mainly eat fish. Because coastal killer whales travel and feed over a large area (and it is not known where

they spend the winter), contaminants found in their tissues may reflect the general state of contamination in the Pacific Ocean ecosystem.

In contrast, harbour seals are year-round residents on the coast and occupy relatively small ranges of about 20 km² (Cottrell et al. 2002), making them better indicators of contamination at a local-to-regional scale. Two of their preferred food fish, herring and hake, can be migratory, so there may be some influence on contaminant concentrations in harbour seals from sources outside the seals' immediate range (Ross et al. 2004).

Methodology and Data

Killer Whale Samples

Blubber samples were collected with biopsy darts from killer whales of both sexes and various ages in the three coastal populations. PCBs, dioxins, and furans were analyzed using high-resolution gas chromatography/high-resolution mass spectrometry (Ross et al. 2000). A slightly different grouping drawn from the same original sample set was analyzed later for PBDEs (Rayne et al. 2004). The identity of each individual sampled was confirmed using a photo identification database containing all resident and many transient whales. This provided demographic information and ensured that the same whale was not sampled twice.

Harbour Seal Samples

Tissue samples from harbour seal pups were collected in 1996 from four locations in the Strait of Georgia (Victoria, Vancouver, Crofton, and Hornby Island) and from Queen Charlotte Sound. The concentration of POPs in harbour seals increases with age, especially in males. Therefore, by sampling only pups, researchers could ensure that subjects were all the same age (three to six weeks) and that virtually all of the contaminants carried by pups come from their mothers through the placenta and in milk (Ross et al. 2004).

Total PCBs and total PBDEs in blubber of killer whales of various ages and harbour seal pups are summarized in Table 5 and shown in Figure 7.

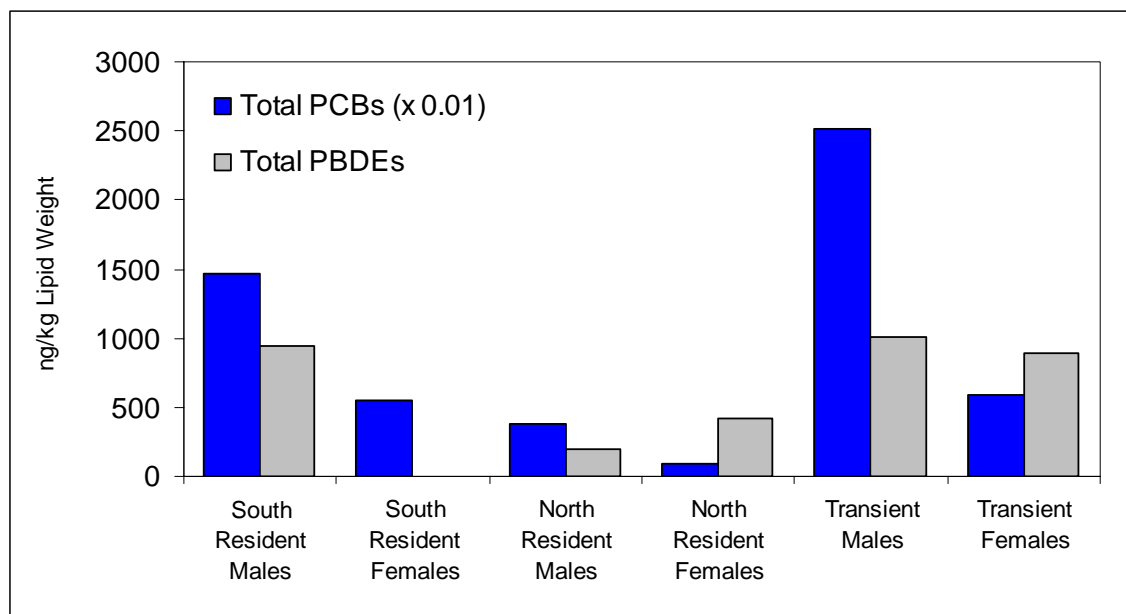
Total PCBs, dioxins and furans in seals and northern resident killer whales are shown in Figure 8.

Table 5. Total PCBs and PBDEs in the blubber of killer whales of various ages and harbour seal pups (3–6 weeks old) on the coast of BC.

| Species | Population sampled | Total PCBs (ng/g) | | Total PBDEs (ng/g) | | Data source |
|----------------------|-----------------------|-------------------|--------|--------------------|-----|--------------------------|
| | | Mean (n) | SD | Mean (n) | SD | |
| Killer whales | S. resident males | 146,300 (4) | 32,700 | 942 (5) | 291 | PCBs: Ross et al. 2000 |
| | S. resident females | 55,400 (2) | 19,300 | – | – | |
| | N. resident males | 37,400 (8) | 6,100 | 203 (13) | 58 | PBDEs: Rayne et al. 2004 |
| | N. resident females | 9,300 (9) | 2,800 | 415 (8) | 338 | |
| | Transient males | 251,200 (5) | 54,700 | 1,015 (6) | 302 | |
| | Transient females | 58,800 (5) | 20,600 | 885 (7) | 353 | |
| Harbour seals (pups) | Strait of Georgia | 2,475 (31) | 174 | – | – | Ross et al. 2004 |
| | Queen Charlotte Sound | 1,143 (5) | 262 | – | – | |

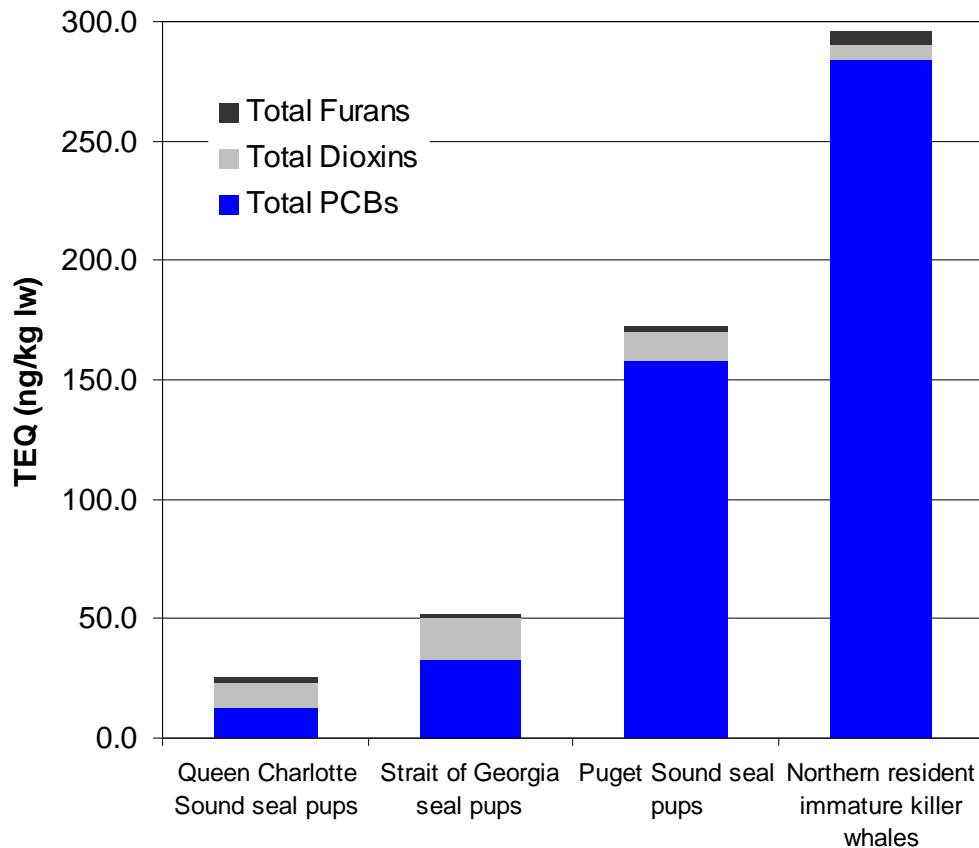
Note: Concentrations are in ng/g (ppb) lipid weight. PCB data on whales are for adults only; PBDE data for whales include some juveniles.

Note: Large variances about the means (SD = standard deviation) are due to the wide age range in whales sampled as well as the small sample sizes.

Figure 7. PCB and PBDE concentrations (ng/kg lipid weight) in male and female killer whales from three populations (southern and northern residents, transients).

Sources: Compiled from Rayne et al. 2004 and Ross et al. 2000, 2004.

Figure 8. Total equivalent concentrations (TEQs) of PCBs, dioxins, and furans in harbour seal pups from Queen Charlotte Sound, Strait of Georgia, and Puget Sound and immature northern resident killer whales (ng/kg lipid weight).



Source: Ross et al. 2000.

Interpretation

Differences in tissue contamination between the three groups of killer whales were consistent with differences in their diet and geographic range. Transient whales were the most contaminated, likely because they feed at a higher trophic level by consuming other marine mammals. Transients are wide-ranging and do not appear to frequent the more industrialized areas of the coast, therefore it is not likely that they occupy more contaminated waters than resident whales.

Although PCB concentrations in northern resident whales were high compared to marine mammals from other parts of the world, they were the least contaminated of the three BC groups of whales. Although they have a similar diet, the more contaminated southern population likely eats more contaminated prey from the industrialized areas of BC and Washington state (Ross et al. 2000). The health risk to killer whales of this concentration of contaminants is not known, but they are among the most contaminated marine mammals in the world (Ross et al. 2004). A high tissue concentration of PCBs was one

reason for the recent classification of southern resident killer whales as Endangered under Canada's Species at Risk Act.

These studies show that PCB concentrations in killer whales are roughly 100 times higher than PBDE concentrations (Rayne et al. 2004), likely reflecting the earlier period of use of PCBs relative to PBDEs. PCBs may also accumulate to a greater extent because PCB molecules are smaller than PBDE molecules, and small size favours more rapid uptake. The most common PBDE congener, BDE-47 (one of the more toxic forms of PBDE), accounts for about 60–75% of the PBDEs in killer whales (Rayne et al. 2004) and is widely distributed in environmental samples (Gill et al. 2004). In all three killer whale populations, males were more contaminated with PCBs (but not PBDEs) than females. This is consistent with the fact that females shed some of their body burden of POPs through the birth and lactation of each calf, whereas males would continue to accumulate contaminants throughout their life.

As shown in Figure 8, accumulated PCBs appear to be more of a risk for harbour seal pups than dioxins and furans. There were differences in tissue concentrations between seal pups born in Queen Charlotte Sound, the Strait of Georgia, and Puget Sound. It appears that contamination in the diet of adult harbour seals is reflected in regional differences in contamination of their pups, with the order of contamination being Puget Sound > Strait of Georgia > Queen Charlotte Sound (Ross et al. 2004). This is consistent with the finding that herring from the Southern Strait of Georgia have lower contaminant concentrations than herring from the central and southern portions of Puget Sound (O'Neill and West 2005).

The burden of mixed contaminants can impair the immune system in seals, possibly facilitating outbreaks of disease and mass mortality (de Swart et al. 1996). Although the health risk to whales from these contaminants is unknown, concentrations in most whales were higher than those likely to cause immunotoxicity in harbour seals (Ross et al. 2000). Research has shown that Vitamin A concentrations in harbour seals in the Strait of Georgia and Puget Sound has been affected by exposure to contaminants (Simms et al. 2000), whereas contaminant concentrations in Strait of Georgia animals are still below those likely to cause immunotoxicity (Ross et al. 2004).

Few data are available on contaminant trends over time in BC's marine mammals, but the PCB concentrations in harbour seals from southern Puget Sound dropped rapidly after controls on the release of PCBs were imposed in 1970s. Levels stabilized after the mid-1980s (Ross et al. 2004), reflecting the persistence of PCBs already in the environment, continued seepage from old sources, and new inputs through atmospheric delivery from distant sources.

Overall, this indicator shows that POPs released in the environment continue to accumulate through the food webs and recycle in ecosystems long after measures intended to curtail emission from industrial sources have taken effect. POPs, such as PCBs, dioxins, and furans, that have been the target of regulation and pollution control efforts for decades, are still accumulating in whale and seal tissues. Adding to the tissue contamination of "legacy" POPs, are the PBDEs, which appear to have been

accumulating within the marine food chain since the 1970s. They have emerged only recently as an environmental concern, in much the same way that PCBs were considered 30 years ago.

POPS IN A TERRESTRIAL MAMMAL: GRIZZLY BEAR

A recent study of grizzly bears in BC found that the coastal bears carry more POPs in their tissues and have a different group of contaminants than the grizzlies that live in the interior (Christensen et al. 2005). Spawning Pacific salmon make up a large proportion of the fall diet of coastal bears, whereas the diet of interior grizzlies consists mainly of plant material with no salmon.

The salmon diet of coastal bears was found to be the source for about 80% of the PBDE congeners, 90% of the PCBs, and 70% of the organochlorine pesticides found in the body tissue of the bears. Although levels of contaminants in the bears were low and it is not known what effect this might have on their health, the study does show the significance of salmon as a transport for contaminants from the North Pacific Ocean.

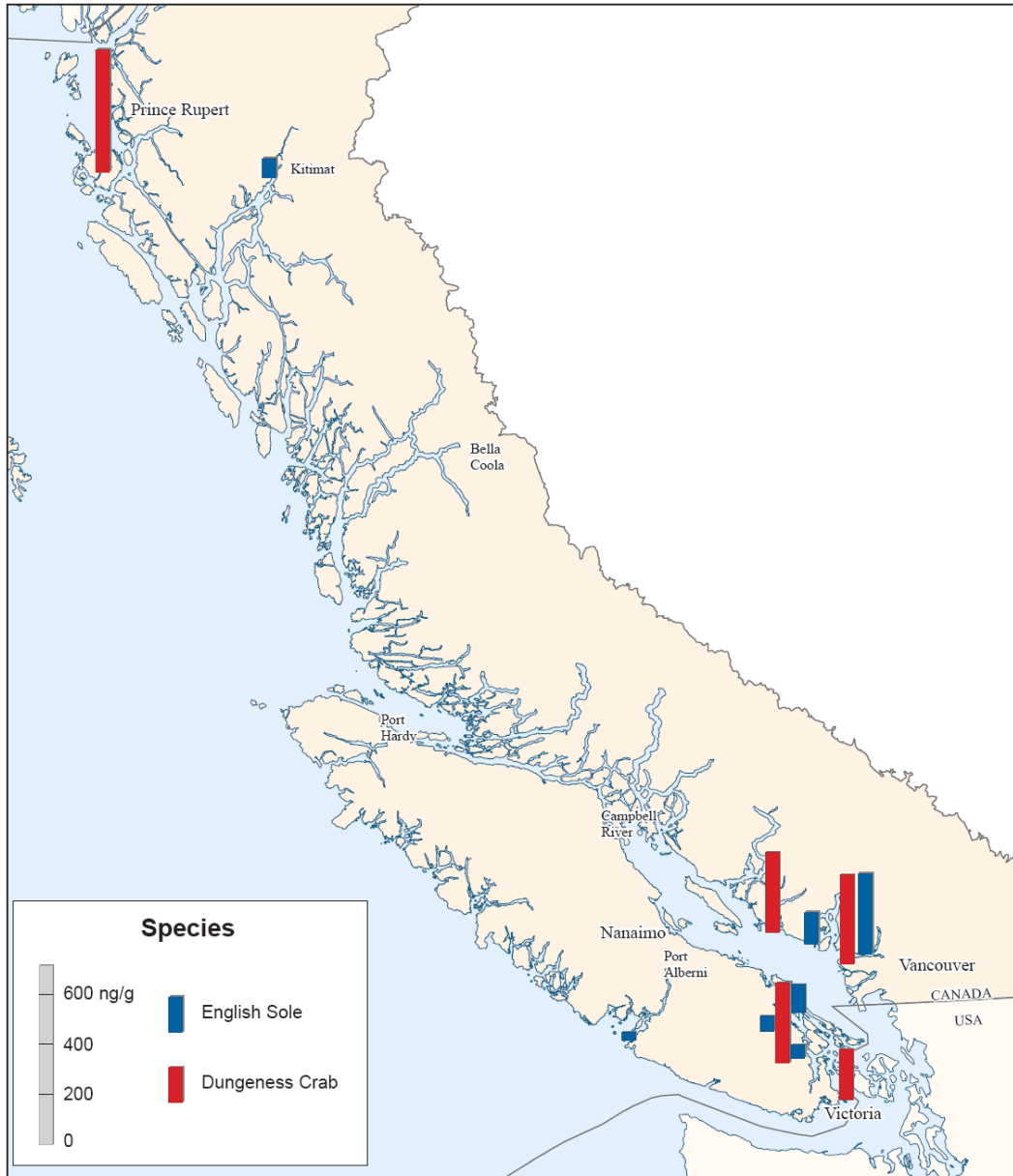
Supplementary Information: PBDEs in other organisms

PBDE levels are rapidly increasing worldwide as a consequence of exponential increases in production (Ikonomou et al. 2002). In North America, the environment and people are 10 or more times as contaminated with PBDEs as in Europe, and studies show that the level of contamination has doubled every 4-6 years (Hites 2004). An assessment of harbour seal "food baskets" (20 species of prey) showed that PBDEs are now ranked as the number three POP in the Strait of Georgia food web, after PCBs and DDT (Cullon et al. 2005).

In BC, studies on the PBDE concentrations in animals from the Strait of Georgia between 1991 and 2000, show variable results, in part because concentrations in the environment have been increasing over the period the samples were collected (Figure 9). Ikonomou et al. (2002) report:

- Dungeness crab collected from sites in the Strait of Georgia and at Kitimat and Prince Rupert had 4–420 ng/g lipid weight in tissues of their hepatopancreas.
- English sole collected from the same areas and time period had 22–310 ng/g lipid weight in the liver tissue.

Figure 9. PBDE concentrations (ng/g lipid weight) in tissues of Dungeness crab and English sole in coastal samples.



Source: Ikonomou et al. 2002.

5. Secondary Indicator: Long-term trends in persistent organic pollutants in great blue heron and cormorant eggs

This is a pressure indicator and in some respects a response indicator. It shows the pressure from POPs in the environment on key marine bird species, while also showing the effects of societal efforts to respond to the problem. It addresses the questions: What is the impact of persistent pollutants in the environment? Are efforts to protect the environment from industrial pollutants effective?

Hérons and cormorants can be used as indicators of the health of the coastal environment because they occur over a wide geographic range. They also feed primarily on small forage fish whose contaminant levels reflect local conditions. The coastal populations of herons do not migrate, and cormorants do not migrate outside the region.

Since herons and cormorants feed mainly on small fish, any changes in the amount of contaminants entering the food chain are rapidly reflected in contaminant levels in their prey, and subsequently in bird eggs. Spatial and temporal patterns in environmental contamination can be evaluated by measuring contaminant concentrations in eggs.

Methodology and Data

The Canadian Wildlife Service began monitoring toxin levels in the eggs of great blue herons (*Ardea herodias*) in 1977 and double-crested cormorants (*Phalacrocorax auritus*) in 1970. Environment Canada has used these data from the Georgia Basin to establish an indicator of contamination in birds that feed higher on the food chain (c.f., Environment Canada 2005a,b,d).

More than 20 great blue heron colonies were sampled, beginning in the 1980s. Three colonies, located in the Georgia Basin, were chosen for in-depth monitoring. The University of British Columbia colony reflected general non-point source urban pollution; the Nicomekl colony represented a rural habitat, and the Crofton colony reflected point-source pollution from a pulp and paper mill. Samples were maintained in a tissue bank, which allowed them to be reanalyzed and trends to be assessed retrospectively as new contaminants were identified. Compounds for analysis have included industrial organochlorines (PCBs, dioxins and furans, and PBDEs) and organochlorine pesticides (DDE, chlordane, dieldrin).

Data were obtained from eggs sampled by the Canadian Wildlife Service. No more than one egg was removed from any nest. The egg contents were placed in glass jars with a chemically cleaned foil liner between the lid and jar, or wrapped in aluminum foil and then frozen until analysis.

In some years, eggs from individual colonies were pooled before analysis by combining equal volumes from each egg to make one sample. In other years, individual eggs were analyzed (Elliott et al. 2001; Turle and Collins 1992).

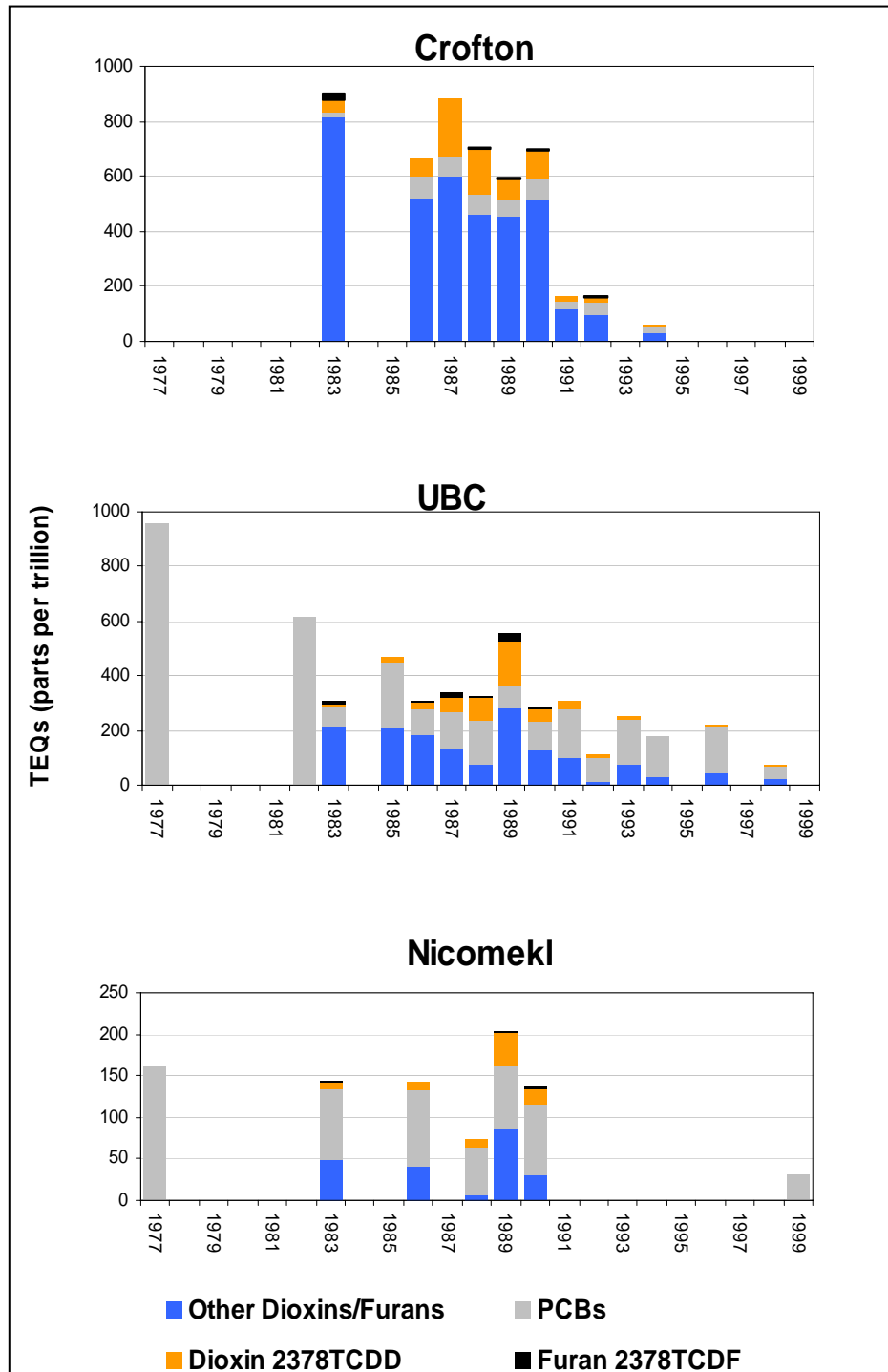
The levels of contaminants were determined by Environment Canada's national laboratories, using methods of the period. Conversion factors were used to compare early concentrations with later values to take into account refinements in analytical methods over the period these data were collected. For example, conversion factors were applied to concentrations of PCBs measured as PCB 1260 (early 1970s) and PCB 1254/1260 (1973–1987) to convert them to the sum of PCB congeners (after 1987). After 1987, total PCBs were measured as the summed concentrations of the individual congeners present.

Concentrations of PCBs, dioxins, and furans are displayed as toxic equivalent concentrations (TEQs) on a wet weight basis, derived using standard conversion factors. (Note: these TEQs are not strictly comparable to data reported in the previous indicator for marine mammals, which were based on lipid weight).

Figure 10 shows the different patterns of contamination with PCBs, dioxins, and furans in the eggs of the three test colonies of great blue herons. The levels of the most toxic dioxin (2,3,7,8-TCDD) and furan (2,3,7,8-TCDF) congeners are shown separately along with the total other congeners of dioxins and furans.

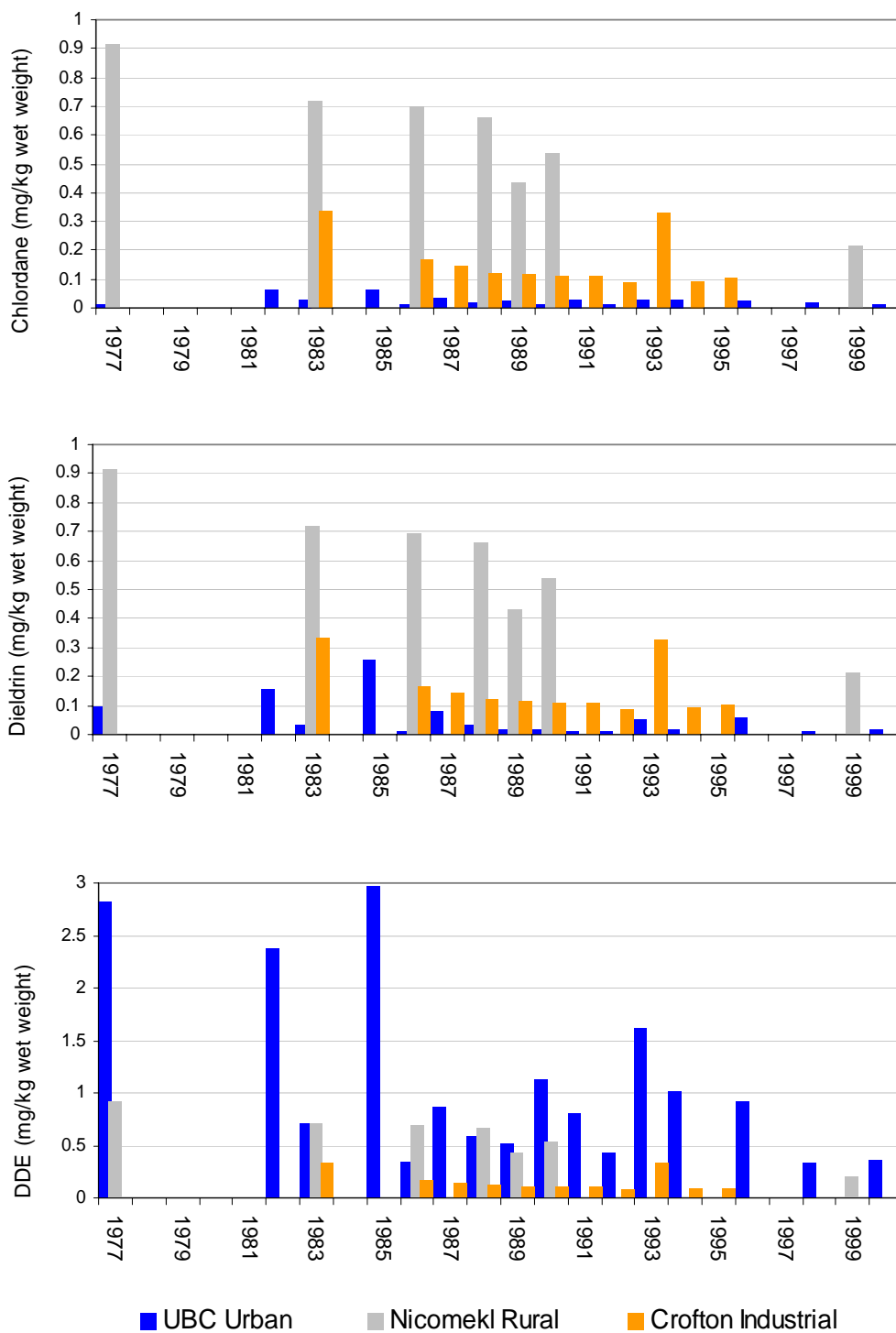
Figure 11 shows the levels of contamination in the three colonies from three organochlorine pesticides that are no longer in use in Canada: DDE (breakdown product of DDT), dieldrin, and chlordane.

Figure 10. Pattern of toxic equivalents (TEQs in parts per trillion, ppt, wet weight) of industrial organochlorines in great blue heron eggs from colonies at UBC, Nicomekl, and Crofton.



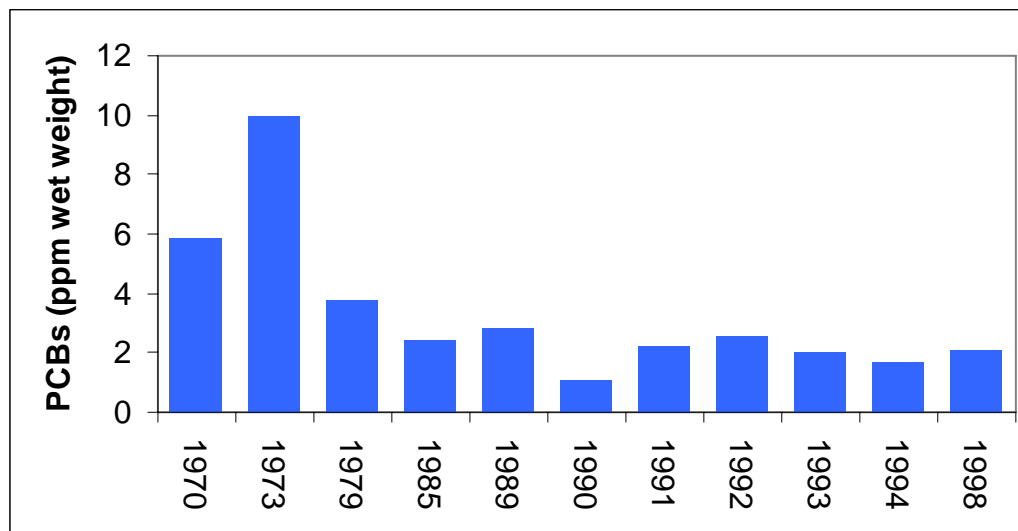
Source: Canadian Wildlife Service (Environment Canada 2005b).
 Note: Only PCBs were measured in 1977 and 1982 at UBC; in 1977 at the Nicomekl colony.

Figure 11. Concentrations of organochlorine pesticides (mg/kg wet weight) in great blue heron eggs from the Georgia Basin (UBC, Nicomekl, and Crofton).



Source: Environment Canada 2005b.

Figure 12. PCBs in double-crested cormorant eggs from Mandarte Island (southern Strait of Georgia) (parts per million, ppm, wet weight of PCBs). Values are arithmetic means of the sum of all PCB compounds.



Source: Canadian Wildlife Service 2000 (Environment Canada 2005d).

Interpretation

PCBs, Dioxins, and Furans

Monitoring shows that PCB, dioxin, and furan concentrations have been decreasing in heron eggs (Figure 10) since the late 1970s and early 1980s. Until 1990, the Crofton heron colony was most contaminated by dioxins and furans coming from the nearby (point source) Crofton pulp and paper mill. The UBC colony, which forages in the Fraser River estuary, showed more exposure to PCBs, which would originate from a variety of non-point and point sources. These differences in pattern reflect differences in local impact between a pulp mill and a large urban centre. The lower levels of contaminants from the rural colony (Nicomekl) is consistent with what would be expected for a colony exposed to lower deposition of POPs overall.

The double-crested cormorant colony at Mandarte Island, in the southern Strait of Georgia, shows a similar history of PCB exposure (Figure 12). The mean concentration in eggs fell from 10 mg/kg wet weight in 1973 to less than 2 mg/kg in 1998. This colony occupies a larger geographic range than the UBC herons, feeding over most of the Strait of Georgia and Puget Sound in the fall and winter. The contaminant levels in their eggs would indicate broader contaminant levels over the region.

In 1987, studies on the impact of elevated dioxin and furan concentrations on heron chick development found that the Crofton heron colony failed to produce chicks and had very high dioxin/furan toxicity in their eggs (Elliott et al. 2001). In 1988, embryos and chicks

had significantly higher levels of organochlorine detoxifying enzymes in their livers, reduced growth rates, and other signs of dioxin exposure (e.g., Hart et al. 1991). Reproductive success improved in later years at the Crofton colony. This is likely related to the decrease in levels of dioxin/furan concentrations in the eggs coincident with the sharp drop in dioxin and furan releases from the Crofton pulp mill after 1989 (see Table 1, Indicator 1) (Sanderson et al. 1994).

The drop in PCB concentration in the eggs coincides with the timing of controls on PCB release in North America, starting in 1977. The relative contribution of dioxins and furans to the total TEQ has decreased substantially in response to these restrictions. Residues continue to persist, however, from past deposition and ongoing atmospheric deposition of PCBs (Wilson et al. 1996). Added to the residues from older POPs released into the environment is the more recent threat of PBDEs, with concentrations increasing rapidly in the eggs of both herons and cormorants (Elliott et al. 2005). Mean concentration of PBDEs were 455 ng/g wet weight in the heron eggs from the UBC colony in 2002. At some locations, concentrations of the most toxic PBDEs (the penta-dibromodiphenyl ether forms) in fish are approaching levels potentially toxic to fish-eating birds (Elliott et al. 2005).

Organochlorine Pesticides

Monitoring shows that concentrations of DDE, dieldrin, and chlordane have been declining over time in heron eggs (Figure 11). As with the industrial contaminants described above, there were local differences in concentration. The lowest concentrations were in the Crofton colony, which would be expected because these pesticides were not typically used in, nor are a product of, pulp mill processes. The greatest concentrations of all chemicals were in eggs from the UBC colony, representing urban non-point sources. All of these pesticides originally had urban uses for insects in buildings as well as in horticultural and agricultural areas.

In birds, the impact of organochlorine pesticides is apparent in thinner eggshells, which are easily damaged, lowering reproductive success. Studies on heron eggs showed that shells were consistently thinner between 1977 and 1986 than they were before 1947; shell thickness also has increased at all three colonies (UBC, Crofton, and Nicomekl) since 1987 (Harris et al. 2002), which coincides with the general reduction in input of these pesticides to the environment.

The decline in chlorinated pesticide concentrations in heron eggs reflects legislation banning or severely restricting Canadian production and use. Permits for the use of chlordane were suspended in 1985, with the exception of use to control subterranean termites (discontinued in 1995). The sale of dieldrin was heavily restricted in the mid-1970s, with its last registered use in Canada in 1984. Although DDT was banned in 1969, the breakdown product, DDE, is still present in wildlife at toxicologically significant concentrations. These residues likely come from atmospheric transport from regions where DDT is still used for insect control, as well as from persistent residues in BC soils and sediments from past agricultural use.

Supplementary Information: Pesticides in the Lower Fraser Valley

Pesticides are used to control or eliminate insect pests, weeds, and fungi in the forestry, agricultural, and urban landscapes. Active ingredients in pesticides range from non-toxic to extremely toxic; some are active only at the time of application (e.g., soaps and horticultural oils), others persist for months or more in the environment. In 2003, the most recent year for which statistics are available, 999,767 kg of pesticide active ingredients were sold or used in BC (not including wood treatment chemicals or most domestic-label products; ENKON 2005).

In 2003, studies conducted by Environment Canada detected 34 currently used pesticides in air samples near Abbotsford in the lower Fraser Valley. Although use of most organochlorine pesticides (which are POPs) has been restricted, both lindane and endosulfan remain in use and were detected at low concentrations along with several organochlorine pesticides that are no longer used. Endosulfan peaked at 380 pg/m³ during the summer, at around the time it was applied to crops.

Other, non-chlorinated, but persistent pesticides found were the herbicide atrazine and the organophosphate insecticide malathion. Atrazine peaked at 150 pg/m³ in air but was not detected in rainwater. Malathion was present at the highest concentration, 2500 pg/m³. Atmospheric concentrations of both malathion and atrazine at Abbotsford were the highest measured across Canada in 2003 (Blanchard et al. 2005).

WHAT IS HAPPENING IN THE ENVIRONMENT?

A wide range of contaminants are detectable in BC's coastal environment, originating from a variety of human activities. The contaminants described in the indicators included PCBs, dioxins and furans, PBDEs, mercury, and DDE, all of which are persistent and bioaccumulative, and PAHs and some organophosphorus pesticides, which are less persistent. Many of the contaminants were discharged locally into the environment in the past, through practices that are socially and legally unacceptable today, but which were widespread during the first half of the last century.

Overall, the environmental concentrations of PCBs, dioxins and furans, mercury, DDE, and other organochlorine pesticides have fallen. This is a direct result of regulatory controls on release of these substances to the environment that were instituted in the 1970s–1990s. Except at contaminated sites, the concentrations found in the air, water, and general environment are low. Through bioaccumulation and biomagnification, however, much higher concentrations of persistent chemicals appear in animals near the top of the marine food chain, such as marine birds, seals, and killer whales.

PCBs, dioxins, furans, and some pesticides are called “legacy POPs” because residues remain in the environment long after actions were taken to eliminate the main commercial and industrial sources of these chemicals. For animals at the top of the marine food chain, PCB residues are still the most toxicologically significant of all POPs, even though almost 30 years has elapsed since release of PCBs was prohibited

throughout North America. Residues of DDE are still significant more than 35 years after DDT was banned in Canada. These POPs will persist in the coastal environment for decades, gradually diminishing in toxicological importance until they reach equilibrium with new inputs.

PBDEs represent a new class of persistent contaminants that have become a focus of concern within the last 5–10 years because levels in the environment have been rising rapidly. PBDE concentrations in the breast milk of Vancouver area mothers doubled every two and half years throughout the 1990s (Ryan et al. 2002). Concentrations have been doubling in marine mammals about every 5 years (Hites 2004) and in marine birds about every 5.7 years (Elliott et al. 2005).

With the withdrawal of pentaBDE and octaPBDE from the market in Europe and North America, a decrease in environmental concentrations would be expected, similar to the decrease seen in other POPs after controls were instituted. All PBDEs are currently being assessed under CEPA, but only decaBDE is still in use. It is distributed in commercial and consumer products to a greater extent than PCBs were, and there is evidence that it can break down into the more toxic penta- and octaBDE forms.

WHY IS IT HAPPENING?

With thousands of substances in use, it may be surprising that so few have emerged as environmental issues. Those that do, including the examples in this paper, share a similar history. At the outset, they were discovered, produced, and used freely as improvements to commerce and society, without regard for their transport or fate within the environment. Their potential hazard began to be recognized as residues accumulated in people and wildlife near the top of the food chain. Toxicological studies were followed a few years later by campaigns to reduce applications and, ultimately, by controls to restrict or eliminate manufacture, use, release, and disposal. Because there are often many possible sources for POPs and other contaminants, it is a challenge to make the links between the source and the effects to evaluate the success of mitigation efforts.

The Canadian Environmental Protection Act of 1988, and its subsequent amendment in 1999, were passed to prevent environmental pollution. Part of the intent of CEPA (1999) is to protect human health and the environment from substances in use. Despite regulations and voluntary controls, however, there are still sources of continuing, low input of contaminants to the provincial environment:

- Use of decaBDEs in furniture, textiles, plastics, and other consumer goods.
- Local, industrial, and other activities continue to produce small amounts of contaminants. Examples are burning salt-laden wood waste in boilers and trash in backyard burn barrels; both activities produce dioxins and furans (see “What You Can Do”).
- Breakdown processes in soils and sediments that are still releasing contaminants from past uses (such as DDT).

- Accidental releases or spills of controlled pollutants. In the case of PCBs, there remains a risk that they may be released from controlled storage and landfill sites and from electrical equipment still in use.
- Transport over long distances in the atmosphere from parts of the world where they are still in use. Scientists suspect that the slow rate of decrease for DDE in wildlife, despite the long-time ban on the parent pesticide throughout North America, is due to long-range transport from other continents as well as the slow release of residues from past use.

WHY IS IT IMPORTANT?

All of the substances discussed in this paper are threats to human health and the environment. Human health risks have been monitored and controlled through regulatory protection of food, limits on exposure in the workplace, and limits on exposure from consumer and industrial products and other activities. Because the risks have been recognized retrospectively, however, they have been hard to manage for chemicals already released into the environment. For example, the fact that organochlorine pesticides caused eggshell thinning in birds was an unexpected impact, far removed from where the hazards might have been expected. More recently, harbour seals continue to show demonstrable physiological effects from POPs in their body tissue, and the cumulative effect of POPs in killer whales is a serious concern.

The costs of contamination, both to the economy and to the environment, have not been calculated. Economic costs range from loss of livelihood, loss of markets, and loss of harvest (e.g., closures of shellfish harvesting areas) to direct costs for cleaning up thousands of contaminated sites, replacing commercial and industrial equipment, and paying for monitoring services and regulatory systems.

WHAT IS BEING DONE ABOUT INDUSTRIAL CONTAMINANTS?

Internationally, Canada was the first country to ratify the Stockholm Convention on Persistent Organic Pollutants, in 2001. The objective of this international agreement is to protect human health and the environment from persistent organic pollutants. It calls for the elimination, immediately or in the long term, of some of the world's most dangerous chemicals: PCBs, dioxin and furans, hexachlorobenzene, and the pesticides DDT, dieldrin, aldrin, endrin, chlordane, heptachlor, toxaphene, and mirex.

In 2000, Canada established the Canada Persistent Organic Pollutants Fund with a commitment of \$20 million to help developing countries and those with economies in transition deal with POPs. The fund is administered by UNEP and the World Bank and assists countries to implement the Stockholm Convention. As of mid-2005, about half the funds had been dispersed to over 70 projects.

In Canada, the Canadian Environmental Protection Act (CEPA), includes procedures for investigating and assessing substances, and for regulating substances that are, or might

become, toxic as defined by CEPA. CEPA requires the Minister of the Environment and the Minister of Health to “categorize” all of the approximately 23,000 substances on the Domestic Substances List (DSL). This is a prioritization process to identify substances on the DSL that should be subject to screening assessments and potential control if they are found to be toxic. The program is intended to identify those substances with the greatest potential for exposure or that are persistent or bioaccumulative and inherently toxic to humans and other living organisms. All substances on the DSL are to be categorized by late 2006. This has led to the creation of two Priority Substances Lists, comprising a total of 69 substances or groups of substances. Under CEPA provisions, these 69 substances have been assessed to determine if they are toxic and whether there should be controls over production and release to the environment. A CEPA Environmental Registry provides information to the public and supports public participation in environmental decisions (www.ec.gc.ca/CEPARegistry/). The Registry provides access to toxic chemical assessments, inventories of substances and their toxicity, the industries they are associated with, and proposed management strategies and regulations.

At the provincial level, an updated BC Environmental Management Act (2003) replaced two earlier pieces of provincial legislation, bringing provisions from both into one statute. New provisions for toxic substances and waste management were introduced, including changes affecting contaminated sites, hazardous wastes, and waste discharge regulations. New sediment criteria incorporated in the Contaminated Sites Regulation provide both a health measurement tool related to sediment concentrations for several of the substances mentioned in this paper and a strong inducement for industry to avoid creating future contaminated sites. The act covers a broad range of environmental regulations (see www.qp.gov.bc.ca/statreg/stat/E/03053_00.htm).

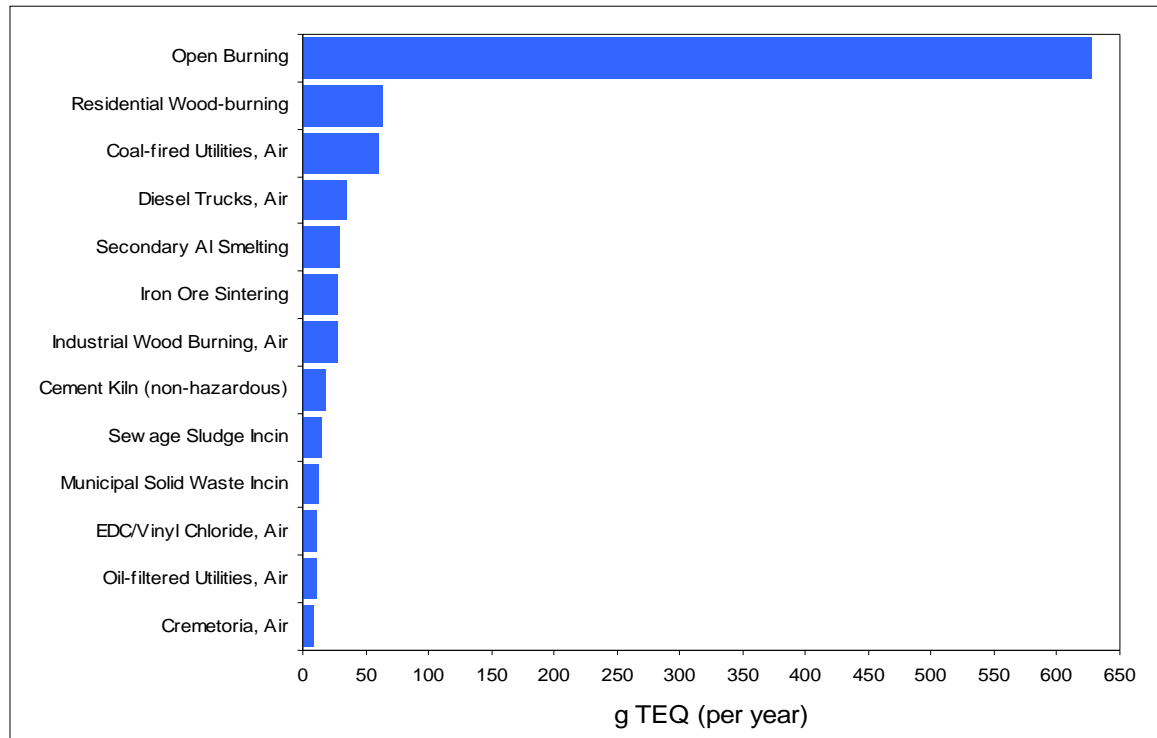
Under the Canada-wide Environmental Standards Sub-Agreement between federal and provincial Environment Ministers, coastal pulp and paper mills must reduce their atmospheric emission of dioxins and furans created by burning wood waste to less than 500 pg TEQ/m³ by 2006 (CCME 2001). The atmospheric emissions are a byproduct of burning sawmill waste from logs transported in salt water booms. Because salt contains chlorine, when salt-laden wood is burned in pulp mill boilers, it creates small quantities of dioxins and furans. Currently, BC has eight coastal mills with boilers that must meet the criteria under this standard.

WHAT CAN YOU DO?

One source of POPs and other contaminants in the environment is under the control of the individual. The US Environmental Protection Agency (EPA) found that the largest source of dioxin and furan emissions in the US is now the uncontrolled burning of household trash (backyard burning). It found that burning the trash from as few as two average households releases as much dioxins and furans into the environment in a year as a modern full-scale municipal waste incinerator handling the garbage of 40,000–120,000 households (Lemieux et al. 2000). The EPA estimates of the total dioxin and furan load to the environment from backyard burning in the US are 10 times higher than the total emissions from the next highest sources—residential wood burning and coal-fired

utilities (FNB and IOM 2003; see Figure 13). The health risk for people from this source may be exposure through food. Dioxins released into the air settle on plants, which are then eaten by meat and dairy animals. The animals store the dioxins in fatty tissue and people are exposed to the chemicals when they consume the meat and dairy products.

Figure 13. Dioxin emitted in 2004 to air by source in the US (g TEQ per year).



Source: US EPA and National Academy of Sciences.

Note: In Canada, other dioxin sources (e.g., slash burning and burning in beehive burners), may be relatively more important as dioxin sources than in the US.

This source of contaminants would be eliminated by halting the practice of burning waste in open piles or burn barrels. Because this is primarily an issue in rural areas where there is often less access to recycling or waste disposal services, there is an important role for regional districts and communities to provide accessible transfer stations and drop-off boxes, and to develop recycling programs for rural households.

Many municipalities ban open burning as part of local smoke control regulations. Where such burning is still permitted, however, individuals can help solve this problem:

- Buy recyclable and reusable products and avoid disposable items. Buy in bulk and avoid excess packaging.
- Have waste picked up by a licensed waste removal company or take it to a local landfill or transfer station.

- Separate recyclables and drop them off at a local recycling centre. For information on recycling or disposal options, call the BC Recycling Hotline at 1-800-667-4321.
- Compost yard and garden waste, food, and leaves. Rent a chipper or hire a service to chip brush and wood to make mulch.
- Never burn toxic materials (e.g., tires, plastics, construction and demolition waste, treated and painted wood, or rubber).

For more information on avoiding backyard burning, see wapwww.gov.bc.ca/air/particulates/bbsgiyea.html.

This is only one area in which individual actions can affect the quality of the environment. For additional information on preventing pollution, see The Canadian Pollution Prevention Information Clearinghouse (CPPIC). This searchable database of references and links aims to share scientific information and give Canadians access to information on pollution prevention. The clearinghouse is supported by Environment Canada and offers more than 1500 resources for business, government, educators, and individuals. (www.ec.gc.ca/cppic/en/index.cfm)

References

- ATSDR (Agency for Toxic Substances and Disease Registry). 1998. Hazard summary, revised Jan. 2000. Toxicological profile for chlorinated dibenzo-p-dioxins. Public Health Service, US Dep. of Health and Human Services, Atlanta, GA. www.epa.gov/ttn/atw/hlthef/dioxin.html.
- Blanchard, P., T. Harner, Y. Yao, L. Tuduri, W. Belzer, and Y. Ryu. 2005. Atmospheric levels of current-use pesticides in BC. Environment Canada workshop presentation on Canadian Current Use Pesticides.
- Bodaly, R.A., and R.J.P. Fudge. 1999. Uptake of mercury by fish in an experimental boreal reservoir. *Arch. Env. Contam. Toxicol.* 37:103-109.
- Brigham, M.E., D.P. Krabbenhoft, M.L. Olson, and J.F. DeWild. 2002. Methyl mercury in flood-control impoundments and natural waters of Northwestern Minnesota, 1997–99. *Water Air Soil Pollut.* 138:61-78.
- Bright, D.A., and D.V. Ellis. 1990. A comparative survey of imposex in northeast Pacific neogastropods (Prosobranchia) related to tributyltin contamination, and choice of a suitable bioindicator. *Can. J. Zool.* 68:1915-1924.
- Burd, B. 2005. Inventory, data analyses and selection of environmental health indicators for sediment contaminants and/or organic enrichment in B.C. Rep. by Ecostat Research Ltd. for Minist. Water, Land and Air Protect., Victoria, BC. 74pp.
- CCME (Canadian Council of Ministers of Environment). 1999 (updated through December 2004). *Environmental Quality Guidelines, Vol.2.* Winnipeg, MN.

- CCME (Canadian Council of Ministers of Environment). 2001. Canada-wide standards for dioxins and furans. Winnipeg, MN. 12pp.
- Christensen, J.R., M. MacDuffee, R.W. MacDonald, M. Whitarcar and P.S. Ross. 2005. Persistent organic pollutants in British Columbia grizzly bears: Consequences of divergent diets. *Environ. Sci. Technol.* 39:6952-6960.
- Clarkson, T.W. 1997. The toxicology of mercury. *Crit. Rev. Clin. Lab. Sci.* 34:369-403.
- Clotfelter, E.D., A.M. Bell, and K.R. Levering. 2004. The role of animal behaviour in the study of endocrine-disrupting chemicals. *Anim. Behav.* 68:465-476.
- Cottrell, P.E., S.J. Jeffries, B. Beck, and P.S. Ross. 2002. Growth and development in free-ranging harbour seals (*Phoca vitulina*) in British Columbia. *Marine Mammal Sci.* 18(3):721-733.
- Cullon, D.L., S.J. Jeffries, and P.S. Ross. 2005. Persistent organic pollutants (POPs) in the diet of harbour seals (*Phoca vitulina*) inhabiting Puget Sound, Washington (USA) and the Strait of Georgia, British Columbia (Canada): A food basket approach. *Environ. Sci. Technol.* in press.
- DEFRA (UK Department for Environment, Food, and Rural Affairs). 2003. Endocrine disruption in the marine environment programme (EDMAR). www.defra.gov.uk/environment/chemicals/hormone/report.htm.
- DFO (Department of Fisheries and Oceans). 2000. Dungeness crab coastal fisheries. DFO Science Stock Status Report C6-14.
- de Swart, R.L., P.S. Ross, J.G. Vos, and A.D.M.E. Osterhaus. 1996. Impaired immunity in harbour seals (*Phoca vitulina*) exposed to bioaccumulated organic contaminants: Review of a long term feeding study. *Environ. Health Perspect.* 104(Suppl.4):823-828.
- Eisler, R. 1987. Polycyclic aromatic hydrocarbon hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish. Wildl. Serv. Biol. Rep. 85(1.11). 81pp.
- Elliott, J.E., M.L. Harris, L.K. Wilson, P.E. Whitehead, and R.J. Norstrom. 2001. Monitoring temporal and spatial trends in polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) in eggs of great blue heron (*Ardea herodias*) on the coast of British Columbia, Canada, 1983-1998. *Ambio* 30:416-428.
- Elliott, J.E., L.K. Wilson, and B. Wakeford. 2005. Polybrominated diphenyl ether trends in eggs of marine and freshwater birds from British Columbia, Canada. *Environ. Sci. Technol.* in press.
- ENKON Environmental Ltd. 2005. Survey of pesticide use in British Columbia, 2003. Environment Canada and BC Minist. Environ., Victoria. 135pp.
- Environment Canada. 1990. First priority substances list (PSL1): Polychlorinated dibenzodioxins and polychlorinated dibenzofurans. www.ec.gc.ca/substances/ese/eng/psap/PSL1_dioxins.cfm.

- Environment Canada. 1992. Reference method for the determination of polychlorinated dibenzo-para-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) in pulp and paper mill effluents. Rep. EPS 1/RM/19.
- Environment Canada. 2005a. Dioxin/furan levels: An indicator of toxic contaminants in coastal BC. Environment Canada, Pacific and Yukon Region.
www.ecoinfo.ec.gc.ca/env_ind/region/dioxinfuran/dioxin_e.cfm.
- Environment Canada. 2005b. Toxins in great blue heron eggs: An indicator of contaminants in the Georgia Basin. Available from
www.ecoinfo.ec.gc.ca/env_ind/region/gbhtoxin/gbhtoxin_e.cfm.
- Environment Canada. 2005c. Coastal mill furan loadings. Unpubl. data, Enforcement and Emergencies, Environmental Protection, Vancouver, BC.
- Environment Canada. 2005d. PCBs in cormorant eggs: An indicator of toxic contaminants in the southern Strait of Georgia.
www.ecoinfo.ec.gc.ca/env_ind/region/cormorant/pcbs_e.cfm.
- EPA (US Environmental Protection Agency). 2002. Ambient aquatic life water quality criteria for tributyltin (TBT), draft. Office of Water. EPA 822-B-02-001.
- FNB and IOM (Food and Nutrition Board and Institute of Medicine). 2003. Dioxins and dioxin-like compounds in the food supply: Strategies to decrease exposure. National Academies Press. 318pp. [Data cited from Appendix A].
- Garrett, C. 2004. Priority substances of interest in the Georgia Basin: Profiles and background information on current toxics issues. Technical Supporting Document of the Canadian Toxics Work Group Puget Sound/Georgia Basin International Task Force. GBAP Publ. No.EC/GB/04/79.
- GESAMP (Group of Experts on Scientific Aspects of Marine Pollution). 1983. Report of the 13th Session, Geneva, Switzerland, 28 February–4 March 1983. IMO/FAO/UNESCO/WMO/WHO/IAEA/UN/UNEP Joint Group of Experts on Scientific Aspects of Marine Pollution. Reports and Studies No. 18, 50pp. WMO, Geneva, Switzerland.
- Gill, U., I. Chu, J.J. Ryan, and M. Feeley. 2004. Polybrominated diphenyl ethers: Human tissue levels and toxicology. *Rev. Environ. Contam. Toxicol.* 183:55-97.
- Gouin, T., and T. Harner. 2003. Modelling the fate of the polybrominated diphenyl ethers. *Environ. Int.* 29:717-724.
- Harris, M.L., J.E. Elliott, R.W. Butler, and L.K. Wilson. 2002. Reproductive success and chlorinated hydrocarbon contamination of resident great blue herons (*Ardea herodias*) from coastal British Columbia, Canada, 1977 to 2000. *Environ. Pollut.* 121:207-227.
- Hart, L.E., K.M. Cheng, P.E. Whitehead, K.M. Shah, R.J. Lewis, S.R. Rushkowski, R.W. Blair, D.C. Bennett, S.M. Bandiera, R.J. Norstrom, and G.D. Bellward. 1991. Dioxin contamination and growth and development in great blue heron embryos. *J. Toxicol. and Environ. Health.* 32:331-344.

- Health Canada. 2004a. Factsheet: PBDEs (polybrominated diphenyl ethers), Health Canada Online. www.hc-sc.gc.ca/english/media/releases/2004/pbde.htm.
- Health Canada. 2004b. Factsheet: Polybrominated diphenyl ethers (PBDEs) in fish. Health Canada Food Program. www.hc-sc.gc.ca/food-aliment/cs-ipc/fr-ra/e_pbde_fish.html.
- Health Canada. 2004c. Factsheet: Fish and Seafood Survey – 2002. Health Canada Food Program. www.hc-sc.gc.ca/food-aliment/cs-ipc/fr-ra/e_seafood_survey.html.
- Health Canada. 2004d. Flame retardants in Canadian breast milk raise concerns. CBC Health and Science News. 8 June 2004. www.cbc.ca/story/science/national/2004/06/07/pbdes040607.html.
- Hempel, P. (ed.). 2000. High altitude POPs and alpine predators. Science & the Environment Bulletin. Environment Canada. www.ec.gc.ca/science/sandenov00/article1_e.html.
- Hites, R.A. 2004. Polybrominated diphenyl ethers in the environment and people: A meta-analysis of concentrations. Environ. Sci. Technol. 38(4):945-956.
- Hites, R.A. J.F. Foran, D.O. Carpenter, M.C. Hamilton, B.A. Knuth, S.J. Schwage. 2004. Global assessment of organic contaminants in farmed salmon. Science 303:226-229
- Ikonomou, M.G., S. Rayne, M. Fischer, M.P. Fernandez, and W. Cretney. 2002. Occurrence and congener profiles of polybrominated diphenyl ethers (PBDEs) in environmental samples from coastal British Columbia. Chemosphere 46:649-663.
- IMO (International Maritime Organization). 2001. Antifouling Systems Convention. www.imo.org/Conventions/mainframe.asp?topic_id=529.
- Irwin, R.J., M. VanMouwerik, L. Stevens, M.D. Seese, and W. Basham. 1997. Environmental Contaminants Encyclopedia. National Park Service, Water Resources Division, Fort Collins, CO.
- Johannessen, S.C., R.W. Macdonald, and R.M. Eek. 2005a. Historical trends in mercury sedimentation and mixing in the Strait of Georgia, British Columbia. Environ. Sci. Technol. 39:4361-4368.
- Johannessen, S.C., M.C. O'Brien, K.L. Denman, and R.W. Macdonald. 2005b. Seasonal and spatial variations in the source and transport of sinking particles in the Strait of Georgia, British Columbia, Canada. Mar. Geol. 216:59-77.
- Kuriyama, S.N., C.E. Taisness, K. Grotze, and I. Chahoud. 2005. Developmental exposure to low-dose PBDE-99: effects on male fertility and neurobehaviour in rat offspring. Environ. Health Perspect. 113:149-154.
- Laflamme, D. 2005. Polybrominated diphenyl ether (PBDE) flame retardants and human health. Presented at the Puget Sound Georgia Basin Research Conference, 29-31 March 2005, Seattle, WA.
- Lemieux P.M., C.C. Lutes, J.A. Abbott, and K.M. Aldous. 2000. Emissions of polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans from the

- open burning of household waste in barrels. *Environ. Sci. and Technol.* 3:377-384.
- Li, Y.-F., and R.W. Macdonald. 2005. Sources and pathways of selected organochlorine pesticides to the Arctic and the effect of pathway divergence on HCH trends in biota: A review. *Sci. Tot. Environ.* 342:87-106.
- Lomax, D.P., L.L. Johnson, W.T. Roubal, J.E. West, and S.M. O'Neill. 2001. Should we be concerned about estrogenic compounds in Puget Sound waterways? Adult male English sole collected near urban areas may provide the answer. *Proc. Puget Sound Res. Conf.*, Bellevue, WA, 12-14 Feb. 2001.
- Macdonald, R.W., and E.A. Crecelius. 1994. Marine sediments in the Strait of Georgia, Juan de Fuca Strait and Puget Sound: What can they tell us about contamination? Pp. 101-137 in R.C.H. Wilson, R.J. Beamish, F. Aitkens, and J. Bell (eds.). *Review of the marine environment and biota of Strait of Georgia, Puget Sound and Juan de Fuca Strait. Can Tech. Rep. Fish. Aquat. Sci. Rep.* 1948.
- Macdonald, R.W., D. Mackay, Y.-F. Li, and B. Hickie. 2003. How will global climate change affect risks from long-range transport of persistent organic pollutants? *Human Ecol. Risk Assess.* 9:643-660.
- Maguire, R.J. 2000. Review of the persistence, bioaccumulation and toxicity of tributyltin in aquatic environments in relations to Canada's toxic substances management policy. *Wat. Qual. Res. J. Can.* 35:633-679.
- Matthiessen, P., Y. Allen, J. Bignell, J. Craft, S. Feist, G. Jones, I. Katsiadaki, M. Kirby, F. Robertson, S. Scott, C. Stewart, and J. Thain. 2000. Studies of endocrine disruption in marine fish: Progress with the EDMAR programme. *ICES CM* 2000/S:08.
- McDonald, T.A. 2002. A perspective on the health risks of PBDEs. *Chemosphere* 46:745-755.
- Mehrle, P.M., D.R. Butler, B.B. Little, L.M. Smith, J.D. Petty, P.H. Peteman, D.L. Stalling, G.M. De Gaeve, J.J. Coyle, and W.J. Adams. 1988. Toxicity and biocincentration of 2,3,7,8-tetrachlorodibenzo-p-dioxin and of 2,3,7,8-tetrachlorodibenzo-p-furan in rainbow trout. *Environ. Toxicol. Chem.* 7:47-62.
- O'Neill, S.M., and J.E. West. 2005. Persistent organic contamination in whole bodies of Pacific herring (*Clupea pallasii*) in Puget Sound, Washington: Evidence of environmental segregation of stocks based on contaminant levels and patterns of contamination. Presented at the Puget Sound/Georgia Basin Research Conference, 29-31 March 2005, Seattle, WA.
- Pacyna, E.G. and J. M. Pacyna. 2002. Global emission of mercury from anthropogenic sources in 1995. *Water, Air and Soil Pollution* 137:149-165.
- PMRA (Pest Management Regulatory Agency). 2002. Special Review Decision: Tributyltin antifouling paints for ship hulls. PMRA Special Review Decision, SRD2002-01, June 2002. www.pmra-arla.gc.ca/english/pubs/srd-e.html.

- Rahman, F., K.H. Langford, M.D. Scrimshaw, and J.N. Lester. 2001. Polybrominated diphenyl ether (PBDE) flame retardants. *Sci. Tot. Environ.* 275:1-17.
- Rayne, S., and M.G. Ikonomou. 2002. Reconstructing source polybrominated diphenyl ether congener patterns from semi-permeable membrane devices in the Fraser River, British Columbia, Canada: comparison to commercial mixtures. *Environ. Sci. Technol.* 21:2292-2300.
- Rayne, S., M.G. Ikonomou, P.S. Ross, G.M. Ellis, and L.G. Barrett-Lennard. 2004. PBDEs, PBBs, and PCNs in three communities of free-ranging killer whales (*Orcinus orca*) from the Northeastern Pacific Ocean. *Environ. Sci. Technol.* 38:4293-4299.
- Reitsem, T.J., J.A.J. Thompson, P. Scholtens, and J.T. Spickett. 2002. Further recovery of Northeast Pacific neogastropods from imposex related to tributyltin contamination. *Mar. Pollut. Bull.* 44:257-261.
- Ross, P.S., and L.S. Birnbaum. 2003. Integrated human and ecological risk assessment: A case study of persistent organic pollutants (POPs) in humans and wildlife. *Human Ecol. Risk Assess.* 9:303-324.
- Ross, P.S., G.M. Ellis, M.G. Ikonomou, L.G. Barret-Lennards and R.F. Addison. 2000. High PCB concentrations in free-ranging Pacific killer whales, *Orcinus orca*: Effects of age, sex and dietary preference. *Mar. Poll. Bull.* 40:504-515.
- Ross, P.S., S.J. Jeffries, M.B. Yunker, R.F. Addison, M.G. Ikonomou, and J.C. Calamabokidis. 2004. Harbour seals (*Phoca vitulina*) in British Columbia, Canada, and Washington State, USA, reveal a combination of local and global polychlorinated biphenyl, dioxin, and furan signatures. *Environ. Toxicol. Chem.* 23:157-165.
- Ryan, J.J. 2004. Polybrominated diphenyl ethers (PBDEs) in human milk.; occurrence worldwide. 3rd Int. Workshop on Brominated Fire Retardants, Toronto, ON, 6-9 June 2004.
- Ryan, J.J., B. Patry, P. Mills, and G. Beaudoin. 2002. Body burdens and food exposure in Canada for polybrominated diphenyl ethers (PBDEs). *Organohalogen Compounds* 51:226-229.
- Safe, S. 1993. Toxicology, structure-function relationship, and human and environmental health impacts of polychlorinated biphenyls: Progress and problems. *Environ. Health Perspect.* 100:259-268.
- Sanderson, J.T., J.E. Elliott, R.J. Norstrom, P.E. Whitehead, L.E. Hart, K.M. Cheng, and G.D. Bellward. 1994. Monitoring biological effects of polychlorinated dibenzo-p-dioxins, dibenzofurans and biphenyls in great blue heron chicks (*Area herodias*) in British Columbia. *J. Toxicol. Environ. Health.* 41:435-450.
- Simms, W., S.J. Jeffries, M.G. Ikonomou, and P.S. Ross. 2000. Contaminant-related disruption of Vitamin A dynamics in free-ranging harbour seal (*Phoca vitulina*) pups from British Columbia, Canada, and Washington State, USA. *Environ. Toxicol. Chem.* 19:2844-2849.

- Stapleton, H.M., N.G. Dodder, J.H. Offenber, M.M. Schantz, and S.A. Wise. 2004. Polybrominated diphenyl ethers in house dust and clothes dryer lint. *Environ. Sci. Technol.* 38:5306-5311.
- Stewart, C., and J.A.J. Thompson. 1997. Vertical distribution of butyltin residues in sediments of British Columbia harbours. *Environ. Technol.* 18:1195-1202.
- Turle, R., and B. Collins. 1992. Validation of the use of pooled samples for monitoring of contaminants in wildlife. *Chemosphere* 25:463-469.
- Waldichuk, M. 1983. Pollution in the Strait of Georgia: A review. *Can. J. Fish. Aquat. Sci.* 40:1142-1167.
- West, J., S. O'Neill, D. Lomax, and L. Johnson. 2001. Implications for reproductive health in rockfish (*Sebastes* spp.) from Puget Sound exposed to polychlorinated biphenyls. *Proc. Puget Sound Res. Conf.*, Bellevue, WA, 12-14 Feb. 2001.
- Wilford, B.H., T. Harner, J. Zhu, M. Shoeib, and K.C. Jones. 2004. Passive sampling survey of polybrominated diphenyl ether flame retardants in indoor and outdoor air in Ottawa, Canada: Implications for sources and exposure. *Environ. Sci. Technol.* 38:5312-5318.
- Wilson, L.K., J.E. Elliott and P.E. Whitehead. 1996. Chlorinated compounds in wildlife from the Fraser River Basin. Pacific and Yukon Region, Canadian Wildlife Service, Environmental Conservation Branch. *Tech. Rep. Ser.* 251. 73pp.
- Wilson, R.C.H., and S.J. McKinnon. 2003. The Point Grey ocean disposal site, 1975-2000: A 25-year review. *Rep. prep. by 2WE Associates Consulting Ltd. for the Disposal at Sea Program, Environment Canada, Pacific and Yukon Region.* 87pp + Apps.
- Wolfe, M.F., S. Schwarzbach, R.A. Sulaiman. 1998. Effects of mercury on wildlife: A comprehensive review. *Environ. Toxicol. Chem.* 17:146-160.
- Van den Berg, M., L. Birnbaum, A.T.C. Bosveld, B. Brunström, P. Cook, M. Feeley, J.P. Giesy, A. Hanberg, R. Hasegawa, S.W. Kennedy, T. Kubiak, J.C. Larsen, F.X.R. van Leeuwen, A.K.D. Liem, C. Nolt, R.E. Peterson, L. Poellinger, S. Safe, D. Schrenk, D. Tillitt, M. Tysklind, M. Younes, F. Wærn, and T. Zacharewski. 1998. Toxicity equivalency factors for PCBs, PCDDs, PCDFs for humans and wildlife. *Environ. Health Perspect.* 106:775-792.
- Yunker, M.B. 2000. Sediment PAH signatures in the IONA receiving environment. Development of a receiving environment monitoring approach to liquid waste management: Progress Workshop 2, Greater Vancouver Regional District, Burnaby, BC.
- Yunker, M.B., and R.W. Macdonald. 2003a. Alkane and PAH depositional history, sources and fluxes in sediments from the Fraser River Basin and Strait of Georgia, Canada. *Organic Geochem.* 34:1429-1454.
- Yunker, M.B., and R.W. Macdonald. 2003b. Petroleum biomarker sources in suspended particulate matter and sediments from the Fraser River Basin and Strait of Georgia, Canada. *Organic Geochem.* 34:1525-1541.

- Yunker, M.B., W.G. Cretney, and M.G. Ikonomou. 2002. Assessment of chlorinated dibenzo-p-dioxin and dibenzofuran trends in sediment and crab hepatopancreas from pulp mill and harbor sites using multivariate- and index-based approaches. *Environ. Sci. Technol.* 36:1869-1878.
- Yunker, M.B., R.W. Macdonald, R. Brewer, S. Sylvestre, T. Tuominen, and M. Sekela. 2000. Assessment of natural and anthropogenic hydrocarbon inputs using PAHs as tracers. The Fraser River Basin and Strait of Georgia 1987-1997. Fraser River Action Plan / Burrard Inlet Environmental Action Program, Environment Canada, Vancouver, BC. 128pp.
- Zala, S.M., and D.J. Penn. 2004. Abnormal behaviours induced by chemical pollution: A review of the evidence and new challenges. *Animal Behav.* 68:649-664.