

Fireproof killer whales (*Orcinus orca*): flame-retardant chemicals and the conservation imperative in the charismatic icon of British Columbia, Canada¹

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Abstract: Long-lived and high trophic level marine mammals are vulnerable to accumulating often very high concentrations of persistent chemicals, including pesticides, industrial by-products, and flame retardants. In the case of killer whales (*Orcinus orca*), some of the older individuals currently frequenting the coastal waters of British Columbia (BC) were born during the First World War, well before the advent of widespread chemical manufacture and use. BC's killer whales are now among the most polychlorinated biphenyl (PCB) contaminated marine mammals in the world. While the "legacy" PCBs have largely been banned, polybrominated diphenyl ethers (PBDEs) have recently emerged as a major concern. The endocrine-disrupting nature of these two persistent fire retardants in biota spells trouble at the top of the food chain, with increasing evidence of effects on reproductive health, the immune system, and development in exposed mammals. The heavy contamination of BC's killer whales, coupled with their long life span and high trophic level, highlights the need for a "weight of evidence" approach in research, conservation planning, and regulatory decisions. Given the global nature of contaminant dispersion, such approaches can only be effective when carried out on both national and international scales.

Résumé : Les mammifères marins qui ont une longue vie et un niveau trophique élevé sont susceptibles d'accumuler des concentrations quelquefois très considérables de produits chimiques persistants, tels que des pesticides, des sous-produits industriels et des retardateurs d'inflammation. Quelques-uns des épaulards (*Orcinus orca*) les plus âgés qui fréquentent les eaux côtières de la Colombie-Britannique (BC) sont nés durant la Première Guerre mondiale, bien longtemps avant la fabrication et l'utilisation répandues de produits chimiques. Les épaulards de BC sont maintenant parmi les mammifères marins les plus contaminés aux biphényles polychlorés (BCP) au monde. Alors que les BCP traditionnels ont en grande partie été interdits, les éthers diphényles polybromés (PBD) sont devenus un sujet important de préoccupation. La propriété qu'ont ces deux retardateurs d'inflammation persistants de perturber le système endocrinien des organismes vivants cause des problèmes au sommet de la chaîne alimentaire; il y a de plus en plus d'indications d'effets sur la santé reproductive, le système immunitaire et le développement chez les mammifères exposés. La forte contamination des épaulards de BC, associée à leur longue vie et à leur position trophique élevée, milite en faveur d'une approche fondée sur le « poids de la preuve » en recherche scientifique, en planification de la conservation et en prise de décisions réglementaires. Étant donné la dispersion des contaminants à l'échelle globale, de telles approches ne seront efficaces que si elles sont employées, tant à l'échelle nationale qu'à l'échelle internationale.

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Whales, fire, and progress

Fire, and humankind's desire to harness its potential and control its scale, has been a focal point of societies since the

advent of civilization. Whale oil, derived from the blubber of baleen and toothed whales, and seal oil, to a lesser extent, provided the combustible fuel for much of the lighting in houses, buildings, and city streets and eventually naviga-

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tional aids and railroad signals in the 16th to the late 19th centuries (Jackson 1978; Bockstoce 1995).

Commercial whaling drove many of the large whale species to the brink of extinction by the late 19th century, with some still struggling to recover today (Gulland 1976; Ross 1993; Baker and Clapham 2004). Over several centuries, the harpoon killed several million cetaceans, as whaling fleets moved from a depleted European and Arctic theatre into the largely uncharted waters of the Pacific and Indian oceans. Simply put, whale oil provided much of western society's illumination as we tapped into the energy stores that whales had accumulated during their lives in the world's oceans. The flammability of whale oil was not only evident in the myriad of lighting practices, but also by the hazards it presented during processing.

Another class of accidents to which whalers seem peculiarly liable ... is destruction by fire... [Starbuck 1964]

The switch from biological to geological fuels in the 19th century resulted from a combination of factors, including economics, the reliability of product delivery, a waning supply of whales, the Civil War in the US, catastrophic whaling fleet losses resulting from ice conditions in the 1830s and 1870s, and technological progress on several fronts (Bockstoce 1995). Two Canadian discoveries played key roles in transforming the way in which homes and cities were illuminated: the 1846 discovery by Canadian geologist Abraham Gesner of the process that produced kerosene from coal, and the first North American oil well at Oil Springs, Ontario, in 1858 (followed 1 year later by a gusher at Titusville, Pennsylvania). The end of the whaling era had begun.

... the whales themselves will undoubtedly be grateful for the discovery of oil which is fast superseding that hitherto supplied by themselves [Shipping List, as cited by Bockstoce (1995)]

Household and industrial lighting and heating needs grew at an ever-expanding rate in the 19th and 20th centuries, and with them came the hazards of unwanted fire. Population growth and urbanization, and the ensuing boom in wood frame building construction, provided ample tinder for the multitude of catastrophic fires that destroyed large urban tracts.

Flames! Flames! Terrible flames!
How they rise, how they mount, how they fly.
The heavens are spread with a fierce lurid glare,
Red heat is filling the earth with air,
While, mercy! Mercy! We hear the despairing ones cry.
[portion of song *Passing through the Fire* about the Great Chicago Fire of 1871 by George F. Root (1871)]

The insatiable demand for lighting, heating, and energy for domestic and industrial development led to the advent of electricity and electrical distribution grids in the early 20th century. Increasing voltage and grid complexity led to the need for an electrical insulation fluid for capacitors and transformers that was heat- and fire-resistant. This created the market foundation for the emergence in 1930 of one of the early global environmental disasters of the 20th century: the commercial development of polychlorinated biphenyls (PCBs) and the subsequent contamination of aquatic food webs the world over.

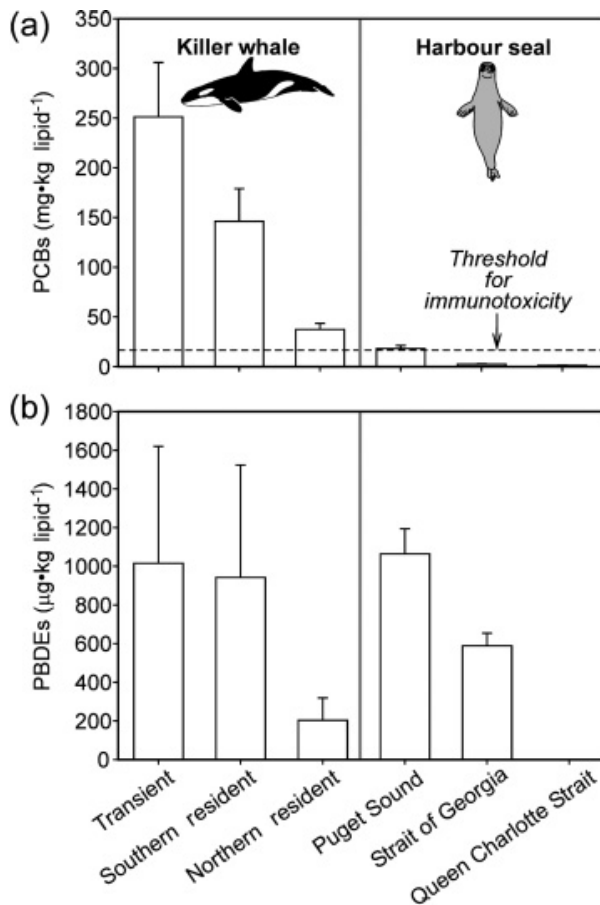
An estimated 1.3 million tonnes of PCBs were produced worldwide between 1930 and 1993, approximately half of which took place in the US and 97% of which were used in the northern hemisphere (Breivik et al. 2002a). Although largely used as heat-resistant oils in electric transformers and capacitors and as industrial-scale hydraulic fluids and lubricants, PCBs were also used in a variety of other minor applications, including specialized lubricants, plasticizers, ink solvents, adhesives, sealants, and lighting ballast (Addison 1986). Approximately 1.4% of global PCB production is thought to have escaped into the environment (Breivik et al. 2002b). The toxicity and widespread occurrence of PCBs led to their being banned in Canada and the US in 1977 and in most other industrialized nations in the late 1970s and early 1980s.

With approximately 400 Canadians dying in fires every year and \$1.7 billion in direct fire-related losses (International Association for the Study of Insurance Economics 2004), reducing the risk of fires remains to this day a public safety priority. A number of new fire- and heat-resistant and -retardant chemicals were developed either to replace the PCBs in their former applications or to serve in other products or uses. Polybrominated diphenyl ethers (PBDEs) represent one such product, which was developed to resist fire as an additive component in consumer electronics, automobile plastics, foam upholstery in furniture, carpets, and some textiles (Birnbaum and Staskal 2004). PBDEs have been sold as three commercial products (penta, octa, and deca formulations) and have rapidly gained popularity as an effective means to meet fire-control regulations for product flammability from the 1970s onward.

The extensive use of the "lighter" penta- and octa-PBDE products may explain much of the observed environmental contamination, as degrading upholstery or foam may have released PBDE congeners into the environment (Hale et al. 2002). Although these two products were removed from the European market in 1998 and are currently being withdrawn from the North American market, the continued use of the deca-PBDE formulation ensures that PBDE releases into the environment will continue during the 21st century. Although the "heavier" deca-PBDE product (consisting largely of BDE-209) was thought by some to be immobile in the environment and resistant to degradation (Hardy 2002, 2005), recent research would suggest otherwise, raising the spectre of increased exposures in biota as PBDE use continues (Law et al. 2003; Watanabe and Sakai 2003; Hites et al. 2005).

Although the PCBs and PBDEs differ in production histories and applications, they also possess remarkable similarities (Figs. 1a, 1b). Both consist of two halogenated phenyl rings with up to 209 theoretically possible congeners, and both represent chemical classes that are lipophilic. PBDEs differ from PCBs in the bound halogen (Br for PBDEs and Cl for PCBs) and in the presence of an ether linkage between the two phenyl rings of PBDEs. These chemical features greatly influence their persistence and fate in the environment, their propensity to partition into aquatic food webs, and their relative vulnerability to metabolic elimination by biota. Positive correlations between PCBs and PBDEs in air and in biota support the notion that these two chemical classes share some general anthropogenic source features

Fig. 1. British Columbia's killer whales (*Orcinus orca*) and harbour seals (*Phoca vitulina*) are exposed to high levels of the regulated polychlorinated biphenyls (PCBs) and moderate levels of the emerging polybrominated diphenyl ethers (PBDEs). (a) The "legacy" of the PCBs persists to this day, with British Columbia's killer whales among the most contaminated marine mammals in the world. These killer whales are at risk for PCB-related effects, as their contamination levels easily surpass established toxicity thresholds, including one for immunotoxicity in harbour seals ($17 \text{ mg}\cdot\text{kg}^{-1}$). (b) The exponentially increasing PBDEs levels are found in British Columbia's marine mammals, presenting an emerging health risk to high trophic level biota for the 21st century. (Data from Ross et al. (2000, 2004), Rayne et al. (2004), and P.S. Ross (unpublished data).)



and some of the physico-chemical properties that govern their behaviour in the environment and in aquatic food webs (Manchester-Neesvig et al. 2001; Strandberg et al. 2001).

And while high trophic level marine mammals and other wildlife have been contaminated by often very high concentrations of numerous persistent organic pollutants (POPs), the PCBs and the PBDEs provide an opportunity to evaluate two major contaminant classes of ecotoxicological concern today. With PCBs largely banned but omnipresent as legacy environmental contaminants and PBDEs currently being used, these chemicals provide a contrasting tale of production histories, having both been designed as stable, heat-resistant and fire-retardant products. In fact, the very features that conferred commercial benefits in their slated applications

spelled trouble for many marine mammal species, as both PCBs and PBDEs possess congeners that do not readily breakdown in the environment and are amplified in aquatic food chains. Such persistence and bioaccumulative properties are frowned upon by regulators, particularly when the contaminants in question are endocrine-disrupting.

Source, transport, and fate of PCBs and PBDEs in the Northeast Pacific Ocean

The source, transport, and fate of PCBs have been well described in over 30 years of research on Great Lakes, Arctic, and Baltic Sea ecosystems and food webs (Strandberg et al. 1998; Muir et al. 1999; Morrison et al. 2002). The PBDEs are less well understood, although an accumulating literature and a structural similarity to PCBs provide a basis for at least an elementary characterization of source, transport, and fate for these currently used flame retardants (Boon et al. 2002; de Wit 2002; Zhu and Hites 2004).

It is a combination of physical, chemical, and biological functions that drive source, transport, and fate processes, delivering with great efficiency those chemicals that possess certain properties to high trophic level biota. In this way, those contaminants that fall within an optimal physico-chemical "window", namely those with an octanol–water partitioning coefficient $\sim \log K_{o/w}$ 4.0–7.0, are amplified in food webs (Metcalf and Metcalfe 1997; Meylan et al. 1999; Christensen et al. 2005). The fundamental processes at the root of contaminant "amplification" in environmental compartments have been described as "solvent switching" and "solvent depletion", reflecting chemical partitioning toward equilibrium in the environment and a solvent-reducing process in biota that retain increasing POP concentrations in lipid stores, respectively (Macdonald et al. 2002).

The octanol–water partitioning coefficients for the PCBs span a lower range ($\log K_{o/w}$ 4.46–8.18 for PCBs 1 through 209) than their PBDE counterparts ($\log K_{o/w}$ 4.74–10.33 for PBDEs 1 through 209) (Hawker and Connell 1988; Braekvelt et al. 2003), conferring a slightly different spectrum of interactive properties on the movement of these two contaminant classes in the environment.

Structural differences and a larger molecular mass relative to PCBs is thought to impede heavier PBDE congeners from ready transport across biological membranes and, hence, uptake by biota (Kelly et al. 2004). Although low levels of BDE-209 in aquatic biota would tend to support this notion, detection of relatively high levels of this heavy congener in terrestrial food web dependent Peregrine falcons (*Falco peregrinus*) in Sweden (Lindberg et al. 2004) and Vancouver Island marmots (*Marmota vancouverensis*) and interior grizzly bears (*Ursus arctos horribilis*) in British Columbia (Lichota et al. 2004; Christensen et al. 2005) indicate that it is mobile in the environment, does partition into terrestrial food webs, and is taken up by biota. And, although the highly brominated PBDE congeners (e.g., BDE-209) would tend not to fall within the optimal $\log K_{o/w}$ window for amplification in aquatic food webs, their debromination, as has been shown in carp (*Cyprinus carpio*), can lead to the formation of lighter congeners (Stapleton et al. 2004) that are

more mobile and can readily accumulate in aquatic food webs.

The atmosphere plays a key role in the (rapid) dispersion of POPs away from source. PCB and PBDE signatures in sediments and in biota reveal that source regions tend to be “heavier” (i.e., more halogenated) than sink regions (Muir et al. 1996; Ross et al. 2004; Ueno et al. 2004), consistent with the global fractionation processes that favour the transport of the more volatile chemicals into remote regions of the world (Wania and Mackay 2001). Wandering air masses are credited with the delivery of PCBs and other POPs into the Arctic and Antarctic (Risebrough et al. 1976; Muir et al. 1992). Prevailing winds from the west and southwest spread air pollutants across the Pacific Ocean (Blais et al. 1998; Jaffe et al. 1999; Li et al. 2002), with wet and dry deposition of pollutants onto the surface waters of the ocean providing a route of entry into its pelagic food web constituents (Ewald et al. 1998; Guruge et al. 2001; Ueno et al. 2004).

The migratory movements of various fish, birds, and other biota can translocate POPs from one region to another. In the Northeast Pacific Ocean, tens of millions of salmon scavenge food web borne contaminants in the open ocean and deliver these to their natal streams as they return to spawn in Alaska, British Columbia, Oregon, and California. Sockeye salmon (*Oncorhynchus nerka*) were shown to import PCBs from the North Pacific Ocean into Alaskan lakes, introducing not only PCBs into the local watershed and its food web, but also a “heavier” pattern of PCBs that differed markedly from that of the PCB congeners deposited through atmospheric processes (Ewald et al. 1998; Krümmel et al. 2003). We recently attributed 70% of the organochlorine pesticides, 90% of the PCBs, and 85% of the “lighter” PBDEs in coastal (maritime) grizzly bears in British Columbia to their consumption of salmon returning from the Pacific Ocean, highlighting the effectiveness of this biological transport process between marine and terrestrial ecosystems (Christensen et al. 2005).

Several POP classes have been detected in killer whales (*Orcinus orca*) in the Northeast Pacific Ocean, including PCBs, dioxins, furans, organochlorine pesticides, and the flame retardants polybrominated biphenyls (PBBs), polychlorinated naphthalenes (PCNs), and PBDEs (Ross et al. 2000; Ylitalo et al. 2001; Rayne et al. 2004), reflecting net accumulation from ingested prey. Pacific salmon are critical to British Columbia’s resident killer whales, comprising an estimated 92% of their annual diet (65% consisting of the long-lived, high trophic level Chinook salmon, *Oncorhynchus tshawytscha*), with the remaining 8% thought to be a mix of other fish, including rockfish (*Sebastes* spp.), lingcod (*Ophiodon elongatus*), halibut (*Hippoglossus stenolepis*), and herring (*Clupea pallasii*) (Ford et al. 1998).

The very high PCB levels and moderate PBDE levels observed in these killer whales are with little doubt due to a combination of what might be described as “local” (via non-salmonid prey) and “global” or “background” (via salmon as prey) sources. Salmon accumulate most of their POP burden at sea, highlighting the potential for a trans-Pacific bridge for contaminant transport (O’Neill et al. 1998). Despite widely cited reports of contaminants in wild Pacific salmon, PCB (~20 µg·kg wet weight⁻¹) and PBDE (~4 µg·kg wet

weight⁻¹) concentrations were relatively low in returning 2002 adult Chinook salmon (Hites et al. 2004a, 2004b) when compared with some of the remaining locally residing killer whale prey items. For example, rockfish species in an urban Puget Sound site had a PCB concentration of approximately 158 µg·kg wet weight⁻¹ (West 1997).

These results highlight the potentially skewed importance of locally residing prey items as a contaminant source to resident killer whales. The threefold higher levels of PCBs and fivefold higher PBDE levels in southern resident killer whales compared with their like-eating northern resident counterparts may, in fact, attest to such a local influence, where the southern residents must contend with the industrial coastal waters of southern British Columbia and northern Washington State. Studies of nonmigratory harbour seals (*Phoca vitulina*) (Ross et al. 2004) and their prey (Cullon et al. 2005) support the idea that Puget Sound represents a regional PCB “hotspot” and possible contaminant source to southern resident killer whales.

In addition to proximity to source, the biology and ecology of marine mammals will shape the extent to which they are exposed to POPs. The marine mammal eating transient killer whales are more PCB-contaminated than their fish-eating resident counterparts, reflecting their higher trophic level (Ross et al. 2000). Males are more PCB-contaminated than females, reflecting the transfer of fat-soluble contaminants to offspring via transplacental and lactational transfer to calves. Long lives lead to a long exposure history and a lifetime accumulation scenario, with male killer whales living up to 50 years and females up to 85 years (Olesiuk et al. 1990). And while the metabolic capacities of killer whales and other marine mammals appear to preferentially eliminate dioxin-like compounds, they are less able to eliminate the mono-ortho PCBs and other globular (nonplanar)-structured POPs (Tanabe et al. 1988; De Swart et al. 1995; Ross et al. 2000). Little is known about the metabolic ability of marine mammals towards PBDEs, but a lack of sex-based differences may reflect a reduced reproductive transfer by females to their offspring or nonsteady state environmental (dietary) conditions as PBDE concentrations continue to increase in long-lived marine mammals (Lebeuf et al. 2004; Rayne et al. 2004).

Effects of PCBs and PBDEs on killer whale health

Four major factors impede the direct documentation of causal relationships between exposure to environmental contaminants and adverse health effects in marine mammals: (i) the abundance of potentially toxic chemicals that have been detected at often high concentrations in marine mammals; (ii) the many natural confounding factors that can complicate interpretation of both contaminant concentrations and health end points in marine mammals, such as age, sex, condition, diseases, reproductive cycles, and stress; (iii) the many anthropogenic factors that can influence the health of marine mammals other than endocrine-disrupting contaminants, including biological pollution (pathogen transfer), fishing pressures (reduction of prey base), noise and disturbance, and climate change; and (iv) the often insurmountable

legal, ethical, or logistical challenges associated with obtaining samples from large mammals in an aquatic or semi-aquatic environment.

However, the conserved nature of many physiological end points does provide a foundation for interspecies extrapolation among vertebrates, where laboratory rodents often serve as a basis for the evaluation of risks to humans of a given chemical (Descotes 2003). Similarly, extrapolation and a “weight of evidence” approach provides a basis for characterizing the risk of contaminant-related adverse health effects in marine mammals and enables a compilation of data from studies using *in vitro* cell culture techniques, laboratory animal models, captive feeding experiments of a marine mammal cohort, and field research using biomarkers or epidemiology (Ross 2000). While extrapolation among species provides an opportunity for a more all-encompassing weight of evidence, observations must incorporate possible interspecies differences in sensitivity to the effects of a given chemical (Hahn 1998).

The dichlorodiphenyltrichloroethane (DDT)-associated eggshell thinning and extirpation of fish-eating birds from large tracts of North America and Europe in the 1960s and 1970s is with little doubt the best-described example of population-level impacts of a single chemical in a wildlife species. The mechanism underlying this occurrence was ultimately resolved through a combination of field and captive dosing studies (Hickey and Anderson 1968; Wiemeyer and Porter 1970; Lundholm 1997). Documenting the health-related impacts of other POP classes has been more challenging, reflecting the highly complex mixtures to which free-ranging animals are now exposed and the multiple levels of possible effects.

At such high concentrations in British Columbia’s killer whales, the PCBs are perhaps the most troubling of contaminants in terms of toxicological risk (Ross et al. 2000). The toxicity of PCBs is multifaceted but can be functionally grouped into “dioxin-like” and “non-dioxin-like” mechanisms of action. The former mechanism involves binding of the planar and mono-ortho PCB congeners to the aryl hydrocarbon receptor (AhR) in vertebrates (Hahn 1998), which leads to induction of certain enzymes and a number of effects, perhaps most notably immunotoxicity. The latter group of mechanisms comprises a suite of effects, including the PCB metabolite associated disruption of thyroid hormone (thyroxine) and vitamin A through a competitive binding to their shared transport complex in circulation (primarily transthyretin – retinol binding protein or TTR–RBP) (Rolland 2000; Simms and Ross 2001).

The larger molecular mass and shape of PBDEs are thought to reduce binding affinity to the AhR such that the risk of dioxin-like effects attributed to PBDEs is more limited than those attributed to their PCB counterparts (Chen et al. 2001). However, as with the PCBs, PBDE metabolites appear to be potent competitors for thyroxine in circulation, where they displace this hormone from its carrier complex (Meerts et al. 2000) or increase elimination via upregulation of phase II uridinediphosphate-glucuronosyltransferase (UDGPT) enzyme activities (Zhou et al. 2001). In addition, there is evidence for other endocrine-disrupting influences for both PBDE parent congeners and some of their metabolites, including estrogenicity (Meerts et al. 2001).

In short, both PCBs and PBDEs can, through their parent congeners and their metabolites, alter thyroid and vitamin A economy and affect neurological development, reproductive development, and immune function in exposed mammals (Meerts et al. 2001; Hallgren and Darnerud 2002; Kuriyama et al. 2005). While both similarities and dissimilarities have been observed in the mechanisms of toxic injury between PCBs and PBDEs, more research is needed to reveal the extent to which the documented toxicological mechanisms for PCBs provide a basis for characterizing the risk of toxicity associated with the less documented PBDEs.

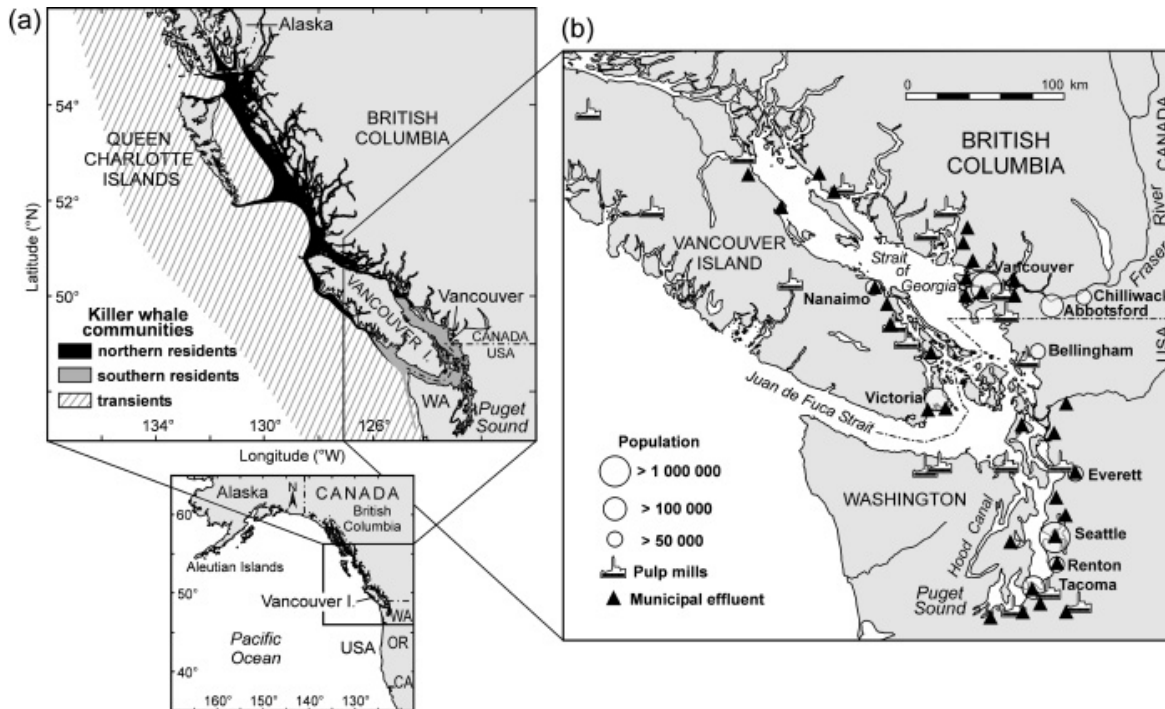
While laboratory animal studies provide an important mechanistic basis for assessing toxicological effects attributable to particular chemicals, they do not fully capture the “real world” (complex mixture) exposure scenarios of marine mammals. In two captive studies, harbour seals fed fish caught from more POP contaminated waters caused reproductive impairment and immunotoxicity (Reijnders 1986; De Swart et al. 1994; Ross et al. 1996). In addition, circulating vitamin A and thyroid hormone concentrations were reduced in the exposed seals (Brouwer et al. 1989; De Swart et al. 1994). PCBs represented the dominant POP in these studies, and laboratory rats exposed to the same contaminant profiles provided additional evidence of a PCB involvement in the effects (Ross et al. 1997).

Evidence from the field comes in part from observations of abnormalities in marine mammal populations inhabiting contaminated coastal regions, including skeletal malformations, adrenal lesions, and reproductive impairment in Baltic Sea harbour, ringed (*Phoca hispida*), and grey (*Halichoerus grypus*) seals (Helle et al. 1976; Mortensen et al. 1992; Olsson et al. 1994). Elevated POP concentrations in harbour porpoises (*Phocoena phocoena*) that died from infectious diseases compared with those that died from trauma suggests that contaminant-associated immunotoxicity is affecting immunocompetence (Jepson et al. 1999). A lack of recovery in the contaminated St. Lawrence beluga whales (*Delphinapterus leucas*) (since the 1962 protection of this population), in combination with observations of disease-associated mortalities, has been partly attributed to POPs and other environmental contaminants (De Guise et al. 1995; Lebeuf et al. 2004).

In addition, field evidence derived from correlations between POP concentrations and a number of end points provides important linkages between the field and more controlled laboratory or captive study scenarios. For example, as observed in some captive feeding and laboratory animal studies, contaminant-associated alterations have been observed in a number of marine mammal populations, including alterations in thyroid hormone and vitamin A (Simms et al. 2000; Jenssen et al. 2003; Braathen et al. 2004), cytochrome P450 enzymes (Nyman et al. 2003), and immune function (Lahvis et al. 1995; Levin et al. 2005). In most cases, confounding factors have been eliminated through study design or by statistical means, but critical evaluation of the “correlation” is needed to ensure that the relationship is causal (Ross et al. 2003; Ross 2004).

Carrying out toxicological studies in free-ranging killer whales is fraught with the obvious challenges listed above. However, despite the lack of mechanistic evidence relating the high POP levels in British Columbia’s killer whales to an

Fig. 2. British Columbia's reproductively isolated killer whale (*Orcinus orca*) communities include the marine mammal eating transients (threatened) and the fish-eating northern (threatened) and southern (endangered) residents (Ford et al. 1998). During their summer feeding season (May–October; see panel *a*), the approximately 89 members of the southern resident community frequent the relatively industrialized Georgia Basin – Puget Sound (see panel *b*), where they feed on returning salmon and can also feed on locally contaminated prey.



observable adverse health effect, there is reason for concern (see Fig. 1). Contaminant-related disruption of several end points in free-ranging harbour seals in local killer whale habitat indicates that POP levels are of biological concern in the coastal food chains of British Columbia and Washington State. In addition, PCB concentrations in northern resident, southern resident, and transient killer whales readily exceed established thresholds for effects of PCBs on reproduction in harbour seals ($25 \text{ mg}\cdot\text{kg}^{-1}$) (Boon et al. 1987) and river otters (*Lontra canadensis*) ($7.5 \text{ mg}\cdot\text{kg}^{-1}$) (Kihlstrom et al. 1992), immune function in harbour seals ($17 \text{ mg}\cdot\text{kg}^{-1}$) (De Swart et al. 1994; Ross et al. 1995), and endocrine effects (thyroid hormone and vitamin A) in river otters ($4 \text{ mg}\cdot\text{kg}^{-1}$) (Smit et al. 1996) and harbour seals ($17 \text{ mg}\cdot\text{kg}^{-1}$) (De Swart et al. 1994; Ross et al. 1995). Extrapolation and a weight of evidence approach therefore imply a significant health risk associated with current PCB burdens in British Columbia's killer whales communities.

Fireproof killer whales and the quest for habitat protection

British Columbia's killer whales face a number of conservation threats. The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) has listed the southern resident killer whales as "endangered", the northern residents as "threatened", and the transients as "threatened" under the terms of the Species at Risk Act (SARA). The southern residents face three major conservation concerns: (i) noise and disturbance associated with heavy vessel traffic; (ii) reduced

availability of preferred prey (Chinook salmon) because of habitat loss, fishing pressures, and climate change; and (iii) high levels of endocrine-disrupting contaminants, such as PCBs (Environment Canada 2005). Each of these is thought to present a real risk to the health and viability of the reproductively isolated southern resident killer whales, such that the SARA-enlisted recovery strategy must address a daunting set of tasks. An initial assessment of the southern resident killer whale community revealed a reduced annual population growth rate compared with that of its northern counterpart (1.3% compared with 2.9%; Olesiuk et al. 1990), which could be due to higher POP concentrations, food limitation, and (or) disturbance from boats. Each of these three stressors on their own can be viewed as a health risk (Ross et al. 2000; Erbe 2002; Williams et al. 2002), but the combination of all three could threaten population viability.

Resident killer whales are heavily exposed to POPs through the consumption of prey such that their habitat might be generically viewed in terms of "local" and "global" sources of contamination. Local habitat for the southern resident killer whales can be described as the area frequented by the approximately 89 individuals of J, K, and L pods during their summer feeding season (~May through October; see Fig. 2a; Ford et al. 2000). Identifying killer whale prey species that reside in this region and working towards targets for prey tissue contaminant concentrations that would protect killer whales from adverse health effects represents an important local (regional) mitigative strategy. This area, the Georgia Basin – Puget Sound transboundary region, is at the

Table 1. While source controls and regulatory efforts led to rapid initial declines in polychlorinated biphenyl (PCB) concentrations in biota, the largely unregulated polybrominated diphenyl ethers (PBDEs) are increasingly exponentially in fish, seabirds, and marine mammals in British Columbia (BC), Canada, and Washington State, USA.

Contaminant	Change (%)	Species	Site	Time frame	Reference
PCBs	-73	Double-crested cormorant eggs	Strait of Georgia	1970–1985	Elliott et al. 1989
PCBs	-92	Harbour seals	Puget Sound	1972–1997	Calambokidis et al. 2001
PBDEs	1200	Mountain whitefish	Columbia River (BC)	1992–2000	Rayne et al. 2003
PBDEs	3640	Great blue heron eggs	Vancouver	1983–2002	Elliott et al. 2005
PBDEs	6950	Harbour seals	Puget Sound	1984–2003	P.S. Ross, unpublished data

Note: Double-crested cormorants, *Phalacrocorax auritus*; harbour seals, *Phoca vitulina*; mountain whitefish, *Prosopium williamsoni*; great blue herons, *Ardea herodias*.

receiving end of emissions and waste discharges from the activities of 7.5 million human inhabitants (see Fig. 2b). On the global scale, the growing Pacific Rim economies ensure that contaminants from as many as 2 billion humans will enter the North Pacific Ocean via direct or indirect means. While daunting, working towards the enactment of the Stockholm Convention, which provides a global framework for phasing out the top 12 POPs (including PCBs), and adding the PBDEs to this protocol would serve to protect remote aquatic food chains from contamination and help protect high trophic level wildlife.

Source control and regulatory efforts can be effective at reducing inputs of POPs into local killer whale habitat, as evidenced by temporal trend profiles in biota. The PCBs and PBDEs provide a contrasting picture of contamination by flame-retardant chemicals (Table 1). Although source controls and regulations led to a sharp drop in PCBs in Puget Sound harbour seals during the 1970s and 1980s (Calambokidis et al. 2001), currently used PBDEs have been increasing exponentially in biota from the Great Lakes (Zhu and Hites 2004), the Arctic (Ikonomou et al. 2002), and British Columbia (Rayne et al. 2003; Elliott et al. 2005). Although PCBs remain the top-ranked POP class in coastal food webs in the Puget Sound and Georgia Basin (as documented in composite “food baskets” for harbour seals), PBDEs were second- and third-ranked in these two basins, respectively (Cullon et al. 2005). With the continued use of deca-BDEs in Canada and the US, their degradation into more mobile and bioaccumulative forms, and their doubling time of 3–5 years observed in the environment, PBDEs are slated to become one of the pre-eminent POPs of the century.

Conclusions

The killer whale habitat in British Columbia is changing, reflecting the arrival of European and Asian hunters, whalers, and fishers in the 18th and 19th centuries and the rapid subsequent colonization of the region. Local habitat is noisier, busier, and more contaminated, and the North Pacific Ocean is warmer and increasingly viewed as a sink for environmental contaminants from a rapidly developing Asian economy. Protecting the killer whale food chain must be central to conservation efforts such that contaminants of concern to either the health of salmon or the health of resident killer whales are identified and mitigative measures adopted.

Contaminant source, transport, and fate functions will not remain static; in an ocean where warmer waters and “fishing down the food chain” have altered food web structure, contaminant pathways are likely to change, posing immense challenges to those attempting to study or mitigate contaminant concerns (Pauly et al. 1998; Schindler 2001; Macdonald et al. 2003).

The demand for a fuel source for lighting in the past resulted in the depletion of the world’s large cetaceans and a conservation dilemma that persists to this day. It is perhaps ironic that the whale blubber used historically to fuel the flames of our lights is now contaminated with chemicals that were designed to prevent the unwanted ignition and spread of fire.

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