Ecological Risk Assessment for the Bay of Fundy:

DDT and Mercury

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1.0 Introduction

The Bay of Fundy supports a high diversity of marine life and is an ecologically and economically critical resource for the region. Despite its importance, little is known about whether contaminants are currently threatening the species that inhabit the Bay. It has been more than three decades since the pesticide DDT was banned for use in North America. However, it can still be found in this region because of its persistence and because it is transported long distances in air and water currents from tropical areas where it is still used. Another contaminant of concern is mercury. In contrast to DDT, mercury is present naturally in the environment but its levels in aquatic systems and biota have also been affected by human activities such as burning of fossil fuels or mining. Both DDT and the organic form of mercury concentrate through aquatic food webs to levels that can cause health problems for fish eaters or the fish themselves. The following document provides an overview on these two contaminants of concern, reviews the concentrations of DDT and Hg that have been measured in wildlife in the Bay of Fundy, and assesses whether these levels may be posing a risk to these species.

2.0 Mercury

2.1 Sources of Mercury

Mercury (Hg) is a naturally occurring element that is found in the earth's crust. A number of natural processes contribute to background levels of mercury in the atmosphere and in terrestrial and aquatic environments. Volcanic emissions, undersea vents and forest fires all constitute natural sources of mercury release (Health Canada 2009). The weathering of mercury-rich rocks can also release it into soils and aquatic environments (Health Canada 2009). Natural levels of mercury can vary from one location to the next because of differences in geology; some rocks contain more mercury than others (Environment Canada 2002). For example, deposits of mercury-rich volcanic rock result in higher background concentrations of the metal along parts of the Nova Scotian coast of the Bay of Fundy (Loring 1982).

Although mercury is released into the environment through natural processes, human activities have increased its levels well above what is naturally present. The metal has a variety of industrial applications and is often released as a result of coal-fired power generation, metal mining and metal smelting (Health Canada 2009). In addition, mercury can also be found in a number of household items including thermometers, thermostats, electrical switches and batteries (Health Canada 2011). Improper disposal of these products can result in mercury contaminated runoff from landfills, though contributions of mercury from solid waste disposal sites appear to be minimal (Slack et al. 2005). Municipal waste incineration and wastewater treatment are also sources of mercury and other heavy metals (Health Canada 2009).

Government regulation beginning mainly in the 1970s has drastically reduced mercury releases due to human activities, both around the globe and within the Bay of Fundy region (Health Canada 2011). Sunderland et al. (2010) estimate that concentrations of mercury in the waters of the Passamaquoddy Bay (an embayment of the Bay of Fundy) have decreased by 40% since the 1960s. However, despite these improvements, the long term impacts of historical sources as well as the existence of current inputs make mercury contamination an ongoing concern for the Bay region. Historical sources of mercury in the area include agricultural applications, chemicals employed by the pulp and paper industry and a number of mercury-cell chlor-alkali plants (Sunderland and Chmura 2000). A chlor-alkali plant in Dalhousie, New Brunswick continues to be the only remaining mercury cell plant operating in Canada, although concentrations of mercury released in the wastewaters from this facility are tightly controlled under the Fisheries Act. Other current point source inputs of mercury in the Bay area include atmospheric emissions from the Coleson Cove power plant as well as the Lancaster wastewater treatment plant (Hung and Chmura 2006). Sunderland et al. (2010) report that base metal smelting, municipal waste disposal, and the combustion of fossil fuels now constitute the greatest sources of mercury in Atlantic Canada. Furthermore, there is evidence to suggest that aquaculture operations within the Bay may contributing to the enrichment of sediments with organic material (material containing carbon), which enhances the production of methylmercury (MeHg), the form that concentrates up through food webs, by bacteria (Sunderland et al. 2006).

2.2 Fate of Mercury in Marine Systems

There are a variety of routes by which mercury can enter the Bay of Fundy. Outfall from municipal sewage pipes, wastewater treatment facilities, and industrial operations along the coast are some examples of direct inputs (Sunderland et al. 2006). Mercury can also be transported in the waters of the numerous rivers and their associated tributaries that drain into the Bay of Fundy. Sources of mercury within these watersheds, including runoff from municipalities, agriculture and industry, all represent more diffuse inputs of mercury into the Bay (Loring 1982). Furthermore, Hg can be transported through the air over long distances. It can then be deposited in rain or fog (wet deposition) or absorbed directly from the air into the Bay (dry deposition) (Ritchie et al. 2006). Atmospheric deposition appears to be a significant source of mercury in the Bay of Fundy region. Atlantic Canada and the Northeastern United States have been estimated to account for 12% of combined Canadian and US mercury emissions (Pilgrim et al. 2000). A recent evaluation revealed that, although direct deposition of mercury into the Bay appears to be minimal (roughly 4% of external mercury inputs), atmospheric deposition over the entire catchment accounts for up to 78% of mercury loading from river waters (Sunderland et al. 2010). Sunderland et al. (2012) estimated that the main sources of total Hg into the Gulf of Maine/Bay of Fundy are riverine inputs (5%), wastewaters from industries and municipalities (8%), atmospheric deposition (28%) and oceanographic circulation (59%).

Chronicling the fate of Hg in the environment can be a difficult task as mercury cycling involves a dynamic set of processes that are influenced by environmental factors (temperature, pH, composition of sediments, etc.). The fate of mercury in marine environments largely depends on its form; three main forms of mercury exist in salt waters: elemental (Hg^0), inorganic (Hg^{II}), and organic mercury (MeHg) (Fitzgerald et al. 2007). Mercury can be converted from one form to the other, with Hg^{II} acting as an intermediate between elemental and organic Hg. Inorganic mercury often enters marine environments through atmospheric deposition or through input from rivers (Fitzgerald et al. 2007). In the water, Hg^{II} can be converted to Hg^0 and subsequently lost back to the atmosphere through a process involving light (Ci et al. 2011). Alternatively, Hg^{II} can combine with particles in the water and undergo sedimentation. As such, the sediments of estuarine environments tend to act as mercury “sinks”, preventing a large amount from being carried to the world’s oceans (Fitzgerald et al. 2007). Within sediments, some types of bacteria can convert inorganic mercury to MeHg and this appears to be the dominant way that MeHg is formed within coastal environments such as the Bay of Fundy (Fitzgerald et al. 2007).

Organic mercury (MeHg) is the most biologically relevant form of mercury and poses the greatest concern for human health. This is because it has the ability to bioaccumulate, meaning that it can concentrate within the tissues of organisms because it is retained more efficiently than it is excreted (Environment Canada 2002). MeHg is taken up by aquatic organisms both directly from the water and from the diet. It is also subject to a process called biomagnification in which increasing concentrations are found within organisms that are higher up in the food web (organisms at higher trophic levels) (Environment Canada 2002). As a result, large, predatory fish such as shark, swordfish and tuna, can contain levels of mercury that are considered unsafe for human consumption and that are many times (up to 10^6) higher than what is measured in the water (Environment Canada 2002).

In addition to trophic level, a number of other factors have been associated with increased mercury accumulation in organisms. Greater body size and older age have been linked to increased mercury burdens in fish (Braune 1987a). This relationship presumably exists because larger fish tend to eat more contaminated prey and older organisms have had a longer period for exposure to and accumulation of environmental contaminants including mercury (Braune 1987a, Evans et al. 2005). A similar relationship between mercury accumulation and age has been found for marine mammals (Smith and Armstrong 1978, Pompe-Gotal 2009). Sex is another factor that affects the bioaccumulation of MeHg in fish. In some species, sexually mature females of the same age and comparable size contain higher levels of mercury than their male counterparts. The difference may be because females consume more Hg-contaminated prey to meet the demands of reproduction (Nicoletto and Hendricks 1987).

2.3 Effects of Mercury on Aquatic Biota and Birds

Mercury is toxic to the nervous and reproductive systems of vertebrates and is known to affect fish health. In addition to causing death if very high exposure occurs, longer-term exposure to lower amounts of mercury can impact the growth, development, reproduction and behaviour of fish (Beckvar et al. 2005). Inorganic mercury has been noted to cause frequent surfacing and sinking behaviour, erratic bursts of swimming and general inactivity in rainbow trout (Macleod and Pessah 1973). Young fish are particularly sensitive to mercury and can have decreased hatching success, developmental abnormalities and increased mortality (Huang 2011). Methylmercury is generally considered to be of even greater concern to fish health than inorganic mercury because it is readily stored and accumulated in the body (Beckvar et al. 2005, US Environmental Protection Agency (EPA) 1987). Damage to vital organs, including liver and kidney lesions, are commonly observed when fish are exposed to dietary MeHg (US EPA 1987, Mela et al. 2007). Furthermore, the nervous system is particularly susceptible to the effects of MeHg. It can cause significant damage to brain tissue and result in the disruption of normal behaviours (Berntssen 2003). Finally, MeHg is known to interfere with many different processes that are important for fish reproduction (Crump and Trudeau 2009).

The impacts of MeHg extend to other forms of wildlife as well, including fish-eating birds. In contrast to inorganic mercury, which is easily excreted, nearly all MeHg taken up in the diet is absorbed through the digestive tract (Scheuhammer 1987). The effects of mercury poisoning can be subtle and difficult to detect (Scheuhammer 1987). In birds, reproductive effects are among the more significant consequences of mercury poisoning. Female birds exposed to MeHg often produce fewer eggs; the eggs that are produced are less likely to hatch, and the hatchlings are less likely to survive (Scheuhammer 1987). Behavioural irregularities are also commonly observed in birds as mercury affects their nervous system (Environment Canada 2002). Several studies have investigated these behavioural abnormalities in loons. Adult loons exposed to MeHg often engage in irregular nesting behaviour, while chicks can become less likely to right themselves when placed on their backs, less responsive to the calls of parents and less likely to solicit rides on the backs of their mothers (Scheuhammer and Blancher 1994, Nocera and Taylor 1998, Kenow et al. 2010).

Methylmercury toxicity in fish-eating mammals (including humans) is similar to that in birds. It acts as a neurotoxicant and is readily absorbed, whereas inorganic mercury is largely excreted (Environment Canada 2002). Studies of laboratory animals reveal that short-term (acute) effects of mercury toxicity include convulsions and tingling or loss of feeling (paresthesia) (Environment Canada 2002). Long-term (chronic) exposure to MeHg can result in impaired coordination and reaction time along with various other motor deficits (Environment Canada 2002). Methylmercury can also cause birth defects and learning deficits in the offspring of mammals (Environment Canada 2002). Information from accidental mercury poisoning confirms these effects in humans. Victims of mercury poisoning due to industrial mercury emissions in Minamata, Japan experienced vision and hearing loss, loss of coordination and speech disturbances. Children born to mothers in the area had increased incidence of birth defects, physical and mental developmental delays, and an increased incidence of cerebral palsy (Eisler 1987).

3.0 DDT

3.1 Sources of DDT

DDT, dichlorodiphenyltrichloroethane, first became popular in the 1940s for use as an insecticide. The organochlorine compound functions as a potent neurotoxin in insects. Due to its efficacy and relatively low cost, DDT became the gold standard for crop protection and for disease control. However, concerns over its persistence and potential threat to human and wildlife health would eventually lead to bans on its use beginning in the 1970s (Commission for Environmental Protection (CEC) 2003). In 2001, DDT was recognized as one of twelve persistent organic pollutants (POPs) by the

Stockholm Convention, an international agreement aimed at reducing human exposure to environmental toxicants. DDT is no longer used in Canada or the U.S., though it continues to be applied as a malaria vector control agent in some parts of the world (CEC 2003). The World Health Organization (WHO) endorses the use of the chemical to control populations of mosquitoes, black flies, and other malaria carrying insects. DDT is commonly applied indoors in most parts of sub-Saharan Africa (WHO 2011). Though alternatives exist, none of these chemicals offer the same long-term protection of DDT (WHO 2011).

DDT can still be found within the Bay of Fundy region due to past uses of the pesticide. The pesticide was part of an aggressive aerial spraying program conducted by the New Brunswick government to combat a growing problem with spruce budworm. The worm feeds on species of spruce and pine tree and outbreaks of the pest can pose a considerable threat to the health of a forest (May 1978). DDT was the exclusive focus of the spraying program headed by the province between the years of 1952-1963 (Kerswill 1967). In 1957, the operation encompassed nearly 5.7 million acres of New Brunswick forest, a considerable increase from an original 186, 000 acres (Kerswill 1967). Improper spraying techniques often resulted in areas receiving a higher dosage than was intended. For example, in 1962, some areas received an application of 2 1/4 lbs of DDT per acre instead of an intended ¼ lb, the level that was chosen to prevent harm to fisheries and aquatic life (Macdonald 1962, Kerswill 1967). Similar spraying programs were also conducted in northern Maine (Kerswill 1967). In addition, DDT was commonly applied to potato and cereal crops in New Brunswick, leading to the accumulation of the pesticide in sediments of the Saint John River watershed (Stewart et al. 1977). In Nova Scotia, DDT was commonly applied to apple orchids in the early 1960s to prevent damage to crops by the codling moth (Pickett 1949). It is highly persistent in sediments and soils and can be found in the environment many decades after its original application.

Though DDT is no longer used in Canada, there still remain a variety of routes by which DDT may enter the Bay of Fundy. Residual DDT and related breakdown products

(DDE and DDD) can be transported through runoff waters from agricultural and urban areas (Stewart et al. 1977, Zhang et al. 2010). As with mercury, atmospheric transport also plays a significant role in the distribution of DDT within the environment. DDT can be carried through the atmosphere and subsequently deposited in rain water or through dry deposition (Brun et al. 2008, Zhang et al. 2010). Volatilization of DDT from soils has been observed 23 years after application of the pesticide formulas commonly used in agriculture (Spencer et al. 1996). Urban centers appear to be other potential sources of DDT to the atmosphere (Sun et al. 2006). Additionally, because DDT can be transported through the air over long distances, global as well as regional sources of DDT must also be considered. Long range transport of DDT to North America likely occurs from sources in Asia where the chemical is still being used to control insect vector driven diseases (Guglielmo et al. 2009). Due to global restrictions on its use, atmospheric concentrations of DDT are in general decline both in North America and around the world (Brun et al. 2008, Shunthirasingham 2010).

3.2 Fate of DDT in Marine Systems

Once mobilized in the environment, DDT and its associated decomposition products tend to move from land to the atmosphere and from the atmosphere into oceans (Woodwell et al. 1971)**.** DDT released into the global environment tends to concentrate in Polar regions because low temperatures keep it from evaporating again and trap it in water, soil and sediment (Wania and Mackay 1993). DDT is poorly soluble in water, and in marine environments the majority of the pesticide tends to accumulate in sediments and aquatic biota (Cramer 1973, Environment Canada 1997). In sediments, DDT is converted into DDE, a compound of even higher persistence, by methane and sulfur producing microorganisms (Quensen et al. 1998). DDE can also enter marine systems directly via precipitation; it is found in the atmosphere as a result of volatilization from soils (where similar bacterially-mediated degradation occurs) and photodegradation (breakdown by light) of DDT in the air and can be washed out in rainfall (Maugh 1973). DDD and DDMU are other degradation products that occur, to a lesser extent, in marine systems as a result of the same processes (Quensen et al. 1998).

The half-life of DDT in the marine environment has been estimated at three years; however, DDT can persist for decades within sediments (Oliver et al. 1989, Connell et al. 2002).

DDT and its breakdown products dissolve better in fats than in water and therefore accumulates within the tissues of living organisms. As a result, like MeHg, they are also susceptible to the processes of bioaccumulation and biomagnification. Accumulation in fish tissues occurs through direct absorption from the water (bioconcentration) and also from the diet (Environment Canada 1997, Wang and Wang 2005). Highest concentrations tend to be found within fatty tissues such as the liver, red muscle, and gonads and in fattier species (Environment Canada 1997). DDT is fairly persistent within fish, having a half-life ranging from 112 days to one year in lake and brook trout respectively (Environment Canada 1997). Dietary exposure to DDT and other chlorinated compounds appears to be of greater significance in organisms of higher tropic levels (Duursma et al. 1986). Therefore, organisms at the top of marine food webs, such as marine mammals, are likely to accumulate the highest levels of DDT (Environment Canada 1997). Large, fish-eating birds are also known to contain high levels of DDT residues (Environment Canada 1997).

Lipid content, age and sex are all interacting factors that influence the accumulation of DDT in wildlife. Since DDT is fat-seeking, organisms that have high levels of body fat, such as whales and other marine mammals, have the capacity to accumulate greater levels of the pesticide (Tanabe et al. 1994, Environment Canada 1997). Generally, accumulation of DDT tends to increase with age, as older organisms have been exposed to environmental contaminants for longer periods of time (Environment Canada 1997). However, this trend can vary between sexes. Levels of DDT in males tend to be higher than in females because females are capable of eliminating fatty deposits (and the associated DDT) through various processes associated with reproduction. For example, marine mammals can transfer DDT to their young during lactation because the milk is high in fats (Tanabe et al. 1994). A similar relationship has been observed in species of fish as females eliminate lipid soluble contaminants through egg production (Larson et al. 1993).

3.3 Effects of DDT on Aquatic Biota and Birds

Toxicological impacts of DDT exposure in fish include impaired growth, behavioural abnormalities and reproductive effects (Beckvar 2005). Decreased feeding behaviour as well as excitable and nervous quivering have been observed in bass, salmon, and rainbow trout exposed to DDT (Surber 1946). High short-term doses can also result in a loss of coordinated movement and inflammation of the gills (Lingaraja et al. 1979). At very high concentrations, DDT can also be directly lethal to fish. Studies conducted during the 1962 spraying operation in New Brunswick saw 100% mortality of salmon and trout (parr and fry stages) caged within the lower Cains River, Miramichi, within two weeks of the program start date. The death rate was attributable to DDT as the highest mortality rate for any one species and life stage caged at a reference site (not within the spraying area) was 7%. Additionally, evaluation of the cumulative effects of DDT spraying over multiple years found reduced numbers of salmon fry and parr at sites within the Cains and Renous Rivers when compared to pre-spray records (Macdonald 1962).

Though DDT can also be lethal to fish-eating birds if very high exposures occur, the more recent effects on avian species from dietary DDT exposure are likely to be reproductive in nature (Environment Canada 1997). One well documented effect observed in birds is egg shell thinning. DDT interferes with the deposition of calcium during egg shell formation, resulting in thinner shells. The fragile eggs are easily damaged during incubation, often resulting in a decreased number of hatchlings (US Fish and Wildlife Service 2011). In the 1960s and 70s, DDT was responsible for dramatic declines in populations of osprey and peregrine falcon in Canada and the U.S. Fortunately, due to conservation measures and the ban of DDT use, these populations have since recovered (Government of New Brunswick; Department of Natural Resources 2011, US Fish and Wildlife Service 2011). Additional reproductive effects of DDT exposure in birds include reduced gonad size and decreased egg production by females (Environment Canada 1997).

DDT is only moderately toxic to mammals; however, long-term exposure to relatively low doses of DDT can cause a wide range of effects (Environment Canada 1997). DDT is a known endocrine disrupter, meaning that it can interfere with the body's natural hormones (Cooper and Kavlock 1997). As such, DDT and DDT related compounds can cause a number of developmental and reproductive abnormalities. Reduced birth weight, reduced fertility and premature labour have all been observed in species of rat and mice (Environment Canada 1997). DDT likely has carcinogenic effects as well. Laboratory studies with rodents have demonstrated that dietary exposure to DDT is associated with the development of tumours, primarily in the liver (Environment Canada 1997). Epidemiological studies of humans suggest that exposure to chlorine-containing pesticides, including DDT, is associated with decreased semen quality, testicular cancer, menstrual cycle abnormalities, and altered timing of sexual development. Most of these effects were observed at levels above general exposure to the public in Europe and North America (Toft et al. 2004).

4.0 Methods - Ecological Risk Assessment Using Risk Quotients

A literature search was completed for DDT and Hg concentrations in the Bay of Fundy (up to June 2015). The data were compiled from published journal articles, reports from the St. Andrew's Biological Station (SABS, Fisheries and Oceans Canada (DFO)) library, student theses, and personal communications for unpublished data. The data were compiled into an Excel spreadsheet including, but not limited to, the following parameters: concentration (wet and/or dry weight, and variance), tissue type, species common name, species latin name, length, weight, depth, % moisture, % lipid, year sampled, and reference. If more than one publication had information for the same species, tissue and contaminant at the same site, average results from all sources were reported for that location. Because DDT consists of several different compounds, we reported total DDT (TDDT) as the sum of *p,p'*-DDD, *p,p'*- DDE, *p,p'* -DDT, *o,p'*-DDD, *o,p'*-DDE and *o,p'*-DDT when these individual chemicals were measured. Mercury concentrations were reported either as total Hg (both organic and inorganic forms) or as MeHg.

A second literature search of chronic (longer-term) and acute (shorter-term) toxicity data for marine and freshwater species exposed to Hg and DDT was completed (see Appendix A). Toxicity studies were evaluated for their robustness (methods, test design, data quality, results) (for details see form in Appendix B) and only those studies with a rating of "satisfactory" or higher were included in Tables 1 and 2. In addition, the studies that were included for the risk quotient (RQ) calculations described below were only those that had measured tissue concentrations of the exposed organisms. Chronic toxicity studies for marine species are limited. For some calculations, studies for freshwater organisms were used for the RQ calculations. To calculate risk for the fishes, the most sensitive endpoint of all fish studies was determined and applied for all fish species in the Bay of Fundy. The same process was used for birds. No chronic Hg toxicity data were available for lobsters and marine mammals. Similarly for DDT, studies on its chronic toxicity were not found for invertebrates. For this reason, Hg and DDT data were reported for some species but toxicity-based risk quotients (RQs) were not calculated. Some sediment studies were also reported here but no measurements of Hg and DDT in the water of the Bay of Fundy were found.

Due to the obvious data gaps in the chronic toxicity tests for marine species, a table of various Canadian guidelines was created (Table 3). This table includes the following guidelines for Hg: Water Quality Guidelines for the protection of aquatic life for total mercury (THg) and for methylmercury (MeHg); Canadian Tissue Residue Guidelines for the protection of wildlife consumers of aquatic biota for MeHg; Interim Sediment Quality Guidelines for THg and Probable Effect Levels for THg in sediment. For DDT the guidelines were: Interim Sediment Quality Guidelines for DDD; Probable Effect Level for DDT in sediments; Probable Effect Level for DDE in sediments; Probable Effect Level for DDD in sediments; and Canadian Tissue Residue Guidelines for the protection of wildlife consumers of aquatic biota for total DDT (TDDT). The Tissue Residue

Guidelines for organisms and the Interim Sediment Guidelines and the Probable Effect Levels for sediment were used to calculate RQs for these guidelines (see below).

To assess whether or not DDT and Hg may be affecting the health of organisms living in the Bay of Fundy, we calculated several different types of RQs. An RQ >1 suggests some risk whereas an RQ of 1 or lower indicates little risk. In this assessment we compared the average concentration or tissue residue (TR) in Bay of Fundy species to what was measured in a similar species and caused an effect during a chronic lab toxicity test. Whenever possible, data from marine experiments were used to calculate the RQ. In some cases due to limited information, results from freshwater chronic exposures were used. The Low Effect Residue (LER) from the most sensitive endpoint of the chronic toxicity studies was divided by 10 (an assessment factor), and then used as the denominator for the RQ calculations.

$$RQ_{LER/10} = TR/(LER/10)$$

An assessment factor of 10 is used to account for some of the uncertainty in applying lab data to field organisms and in extrapolating from one species to another. TR is the tissue residue for that species and tissue found in the Bay of Fundy. RQs for the No Effect Residue were also calculated using RQ_{NER}=TR/NER although these calculations would be much more of a "worst case scenario" than the RQs calculated using LER data. For this reason, the evaluation done herein focussed on the results for the $RQ_{LER/10}$ and are hereafter referred to as RQs. The Tissue Residue Guidelines (TRG) were used to determine risks for those species that may be eaten by birds or marine mammals by dividing the tissue concentrations in the prey species by the TRG (RQ_{TRG}=TR/TRG; referred to as RQ_{TRG} in the text). For sediments, the Interim Sediment Quality Guidelines were used to calculate RQs as follows: RQ_{ISQG}= SC/ISQG (note: SC=sediment concentration; referred to as RQ_{ISQG} in the text). For Probable Effect Level the calculation was: RQ_{PEL}=SC/PEL (referred to as RQ_{PEL} in the text). Only Hg data were available for sediments in the Bay of Fundy.

5.0 Results - Mercury Concentrations and Risk in the Bay of Fundy

5.1 Mercury in Birds

Data for mercury concentrations in birds were found for 24 species collected between 1978 and 2006, and mainly included THg in feather, egg, liver, kidney and muscle tissues (see Table 4 and Appendix C). In one study where both THg and MeHg were measured in birds, similar concentrations of both were found across several bird species but THg was typically higher than MeHg (Bond and Diamond 2009b). Summary data are shown in Table 4 whereas data from individual studies are given in Appendix C. For feather tissues, the lowest MeHg and THg were found for Arctic tern (791 and 891 ng/g ww or ppb, respectively) and the highest MeHg and THg in Leech's storm petrel (5330 and 4855 ng/g ww, respectively). Using this tissue and both THg and MeHg data, RQs were > 1 for Atlantic puffin, Common murre, Common tern, Leech's storm-petrel and Razorbill (range from 1.01 to 5.46) but not for Arctic tern (RQs of 0.81 and 0.91 for MeHg and THg, respectively)(Table 4). Eggs were mainly measured for THg and concentrations ranged from 40 to 891 ng/g ww in Glossy ibis to Leech's storm-petrel, respectively. The RQs calculated for eggs indicated that all but one species (Glossy ibis) were above 1. For Herring gull, eggs from one location (Kent Island) had an RQ > 1 (1.35) but at other locations the RQs were 0.63 and 0.96 (Manawagonish Island and Gulf of Maine). Burgess et al. (2013) show that THg concentrations in herring gull eggs have decreased at Manawagonish Island, New Brunswick over 36 years (1972-2008). Similarly, in Sunderland et al. (2012) the THg concentrations in the Double-crested cormorant at Manawagonish Island, New Brunswick decreased over the same period of time. THg in kidney tissues were lowest in Black-legged kittiwake (242 ng/g ww) and highest in Double-crested cormorant (5345 ng/g ww).

All 9 bird species had RQs > 1 for this kidney tissue (range 1.47 to 32.39) and these RQs tended to be higher than those for the other tissues. Similar results were found for liver tissues where there was a large range in THg concentrations (225 to 7048 ng/g ww) but all species had RQs > 1 (range 1.69 to 52.99). Total Hg in bird muscle was typically lower than for other tissues and ranged from 37 ng/g ww in Black-legged kittiwake to 606 ng/g ww in Double-crested cormorant. Nonetheless, most species still had RQs > 1 (Arctic tern, Herring gull, Black guillemot, Common eider duck, Common tern and Double-crested cormorant) whereas three had RQs < 1 (Black-legged kittiwake, Red-necked phalarope, Bonaparte's gull). In summary, the Hg RQs indicated that there is a risk for all species except Glossy ibis (RQ of 0.48), but the risk varied depending on the tissue examined. Birds at lowest risk are Arctic tern (average RQ of 1.97), Black-legged kittiwake (average RQ of 1.58) and Red-necked phalarope (RQ of 1.14), and at the highest risk are Leech's storm-petrel (average RG of 9.3) and Double-crested cormorant (average RQ of 19.5).

5.2 Mercury in Marine Mammals

Only one study was found with Hg concentrations in marine mammals and it was from the late 1960s. Total Hg was measured in muscle of harbour porpoises and the average concentration was 1054 ng/g ww (Table 5). No toxicity studies were available for marine mammals to calculate whether this level of exposure posed a risk or not.

5.3 Mercury in Fish

Mercury data were found for 18 fish species from the Bay of Fundy (Table 5 and Appendix C). Both THg and MeHg were measured and almost all analyses were done on whole bodies rather than on specific tissues. The exceptions were muscle tissues measured for Atlantic herring, harbor pollock, herring brit, longhorn sculpin, spiny dogfish and witch flounder, for which THg was measured in both whole bodies and muscle or in muscle only. For Atlantic herring, THg concentrations in muscle and whole body were similar (8.92 and 8.94 ng/g ww, respectively). The species that had the lowest THg concentrations were herring brit, harbor pollock, Atlantic herring, pollock, winter flounder and mackerel at 4, 5.0, 8.9, 18.7, 21.1 and 22 ng/g ww, respectively. In contrast, fishes with the highest THg concentrations in whole bodies were spiny dogfish, swordfish, bluefin tuna and thresher shark at 99, 416, 565 and 1472 ng/g ww. Because THg in fish is mainly MeHg, the trends for the MeHg data were the same with pollock, winter flounder and mackerel having the lowest concentrations and dogfish, swordfish, bluefin tuna and thresher shark having the highest MeHg concentrations. The RQs calculated for fishes were done for both THg and MeHg but the results were almost identical for both forms of this contaminant (Table 5). The fishes with the lowest risk (RQ = or < 1) from Hg were herring brit, harbour pollock, Atlantic herring, winter flounder, pollock, mackerel, yellowtail flounder, white hake and Atlantic salmon. Cod and haddock had an RQ < 1 for MeHg (0.8 and 0.61, respectively) and above 1 for THg (1.2 and 1.08, respectively). Witch flounder and longhorn sculpin had RQ ranges greater than 1 for THg (ranging from 1.00 to 13.33 and 1.7 to 4.0 respectively). All other fishes consistently had RQs above 1 for both THg and MeHg and included herring, cunner, spiny dogfish, swordfish, bluefin tuna and thresher shark. The latter two species had the highest RQs (tuna – 15, shark – 43 for MeHg). In summary, over half of the fish species for which there are measurements of Hg had RQs above 1, suggesting some risk to their health.

For fish-eating wildlife (birds, mammals), the RQs that were calculated using Tissue Residue Guidelines also indicated some risk for species living in the Bay of Fundy. Typically the fish that had $RQ_{LER/10}$ > 1 also had RQ_{TRG} > 1. Birds and marine mammals feeding on cod, herring, cunner, spiny dogfish, swordfish, bluefin tuna and thresher shark may be at risk from Hg toxicity because the RQ_{TRG} values were all > 1 (Table 5).

5.4 Mercury in Invertebrates

American lobster and blue mussels were measured for THg in the digestive gland or soft tissues, respectively (Table 6). In lobster, THg was 60 ng/g ww but no toxicity data were available to calculate an RQ. In contrast, mussel average THg ranged from 29 to 34 ng/g ww across sites and the RQs were consistently well below 1 (0.08 to 0.09)(Table 6). In summary, there does not appear to be a risk for mussels to Hg but data are insufficient to evaluate risk for other invertebrates in the Bay of Fundy.

The RQs that were calculated for species that may consume lobster or mussels were all at or below 1 for mussels (0.8-0.9) and most other invertebrates and above 1 for lobsters (1.8) and soft-shelled clams (1.5) (Table 6). This suggests that there may be a risk for the species that feed on lobsters and clams.

5.5 Mercury in Sediments

Some mercury data for sediments in the Bay of Fundy are available from a number of studies. Most have measured THg and these concentrations ranged from 10 to 238 ng/g dw (Table 6). MeHg concentrations were lower at 0.4 and 0.7 ng/g dw. Using the Interim Sediment Quality Guideline, only 1 of 17 THg concentrations exceeded this guideline for sediments from the St. Croix Estuary and the RQ_{ISQG} for this site was 1.83 (Table 6). No sediments exceeded the Probable Effects Level of 700 μg/kg (ng/g dw). The Hg data currently available indicate almost no risk to sediment-dwelling organisms from exposure to this pollutant.

6.0 Results - DDT Concentrations and Risk in the Bay of Fundy

6.1 DDT in Birds

Far fewer measurements of DDT than Hg have been made for birds from the Bay of Fundy and most results are from the 1970s or early 1990s. The data that were available for 13 bird species were for a range of tissues including egg, liver, muscle, fat and carcass (Table 7 and Appendix D). In specific tissues, the highest DDT concentrations were found in fats because DDT has a tendency to accumulate in fatty tissues and the lowest concentrations were measured in liver and muscle. As an example, liver and muscle tissues of the Double-crested cormorant had 8.4 and 4.2 μg/g ww TDDT, respectively, whereas eggs and fats had concentrations of 19 and 162 μg/g ww TDDT, respectively. Whole birds (carcasses) of Black-bellied plover, Dunlin, Greater yellowlegs, Lesser yellowlegs, Semipalmated plover, Semipalmated sandpiper and Short-billed dowtcher were also measured and ranged from 0.09 (semipalmated sandpiper) to 6.19 (lesser yellowlegs) μg/g lipid weight. The only chronic toxicity data for which residues were available were for eggs and, for this reason, RQs were only calculated for bird eggs from the Bay of Fundy. Of the four species with egg data, only Double-crested cormorant had an RQ > 1 (4.13) (Table 7). Black duck, Guillemot and Herring gull had RQs ranging from 0.33 to 0.95. In summary, the TDDT data from the Bay of Fundy indicates that some bird species may be at risk from exposure to this pesticide if current

DDT concentrations are similar. However, DDT data are limited and older (especially for the species shown to be at greatest risk), and the limited toxicity data make it difficult to thoroughly assess risks.

6.2 DDT in Marine Mammals

Like Hg, the information on DDT in marine mammals is very limited for the Bay of Fundy. Total DDT has been measured in Common seal liver (1970s) and blubber from Harbor porpoises and North Atlantic right whales (late 1980s). These studies show higher TDDT in Common seal (12.6 μg/g ww) and lower TDDT in blubber from porpoises and whales (6.34 and 0.12 μg/g ww, respectively)(Table 8, Appendix D). No chronic toxicity data for marine mammals was available and, as a result, RQs could not be calculated for these three species.

6.3 DDT in Fish

Nine species of fish from the Bay of Fundy have been measured for DDT in muscle, viscera, liver and whole bodies and all of these data are from the 1970s. For muscle samples the lowest TDDT concentrations were found in plaice, ocean perch and white hake (0.01, 0.03 and 0.03 μg/g ww, respectively) and the highest TDDT concentrations were found in sea raven and white shark at 0.32 and 0.48 μg/g ww, respectively; viscera for sea raven was similar to muscle at 0.30 μg/g ww (Table 8). Whole body herring had TDDT of 0.06 μg/g ww. Highest TDDT concentrations were measured in white shark liver at 441 μg/g ww. Calculations of RQs indicated that muscle or whole body tissues for all species but shark were below 1 (Table 8, Appendix D). White shark muscle had an RQ of 1.32 suggesting some risk to this species. In summary, the older data that are available for the Bay of Fundy indicate that there was little risk from DDT toxicity to all but 1 of the 9 fish species studied. It is not clear whether a potential risk to white shark currently remains but this warrants some study.

The risk quotients for fish consumers (using Tissue Residue Guidelines) indicated that all species and tissues except plaice were above 1 (Table 8). Highest RQs were found for mackerel (10), bluefin tuna (11), sea raven (21-22) and white shark (34 in muscle). Most notably, white shark liver had an RQ of 31,500. In summary, almost all fish measured for TDDT had RQs above 1. However, it is important to note that these data are at least 3 decades old so the current risk to fish-eating birds and mammals is not known.

6.4 DDT in Invertebrates

With the exception of one study on American lobster hepatopancreas done in the 1980s, all other TDDT data for invertebrates from the Bay of Fundy is from the late 1960s. Clams (*Mya arenaria*), mussels (*Mytilus edulis*) and scallops (*Placopecten magellanicus*) were analyzed and had TDDT concentrations below detection limits, 0.09 and 0.03 µg/g ww, respectively (Table 9). Lobster hepatopancreas had TDDT concentrations of 1.59 µg/g ww. Overall, it is difficult to assess the current risk of DDT to Bay of Fundy invertebrates because only older residue data are available and no relevant toxicity data could be found with which to calculate RQs.

For consumers of these invertebrates, RQ calculations using Tissue Residue Guidelines indicated that DDT in all species but clams posed some risk to invertebrate-eating wildlife with RQs of 6.4 and 2.1 for mussels and scallops and an RQ of 113 for lobsters (Table 9).

7.0 Summary

This report presents concentrations of Hg and DDT in birds, marine mammals, fish, invertebrates, and sediments for the Bay of Fundy and a preliminary risk assessment of these contaminants using the Risk Quotient approach. This approach compares the concentrations of Hg or DDT in the tissues of the organisms to what is known to cause effects in the lab after longer-term exposures and for a similar species, or to what may cause effects in sediment-dwelling organisms or predators.

Using the data that were available, we found that Hg poses a risk to all but one species of bird and about half of the fish species. It was not possible to evaluate risk of Hg for marine mammals and invertebrates. In comparison, 25% (1 of 4) of the bird species and 11% of the fishes (1 of 9) may be at risk from DDT toxicity. It was not possible to calculate RQs for marine mammals and invertebrates. In balance, Hg appears to be a greater threat than DDT to the Bay of Fundy wildlife although there are only more recent data for the former contaminant.

For wildlife that eat fish or invertebrates, some risks were also found for Hg and DDT. For Hg, half of the fish species, lobster and soft shelled clams (but not mussels) may affect the health of their predators because of elevated concentrations of this pollutant. For DDT, 8 of 9 fish species and 3 of 4 invertebrate species had an $RQ_{TRG} > 1$, suggesting some risks for their consumers.

There are several limitations of what has been done in this report. First, relevant lab toxicity data were not always available for the biota in the Bay of Fundy or for similar species. More specifically, toxicity data for marine mammals does not exist and, for many studies, concentrations of DDT or Hg in tissues of the test animals were not available. The limited data that were available for birds, fishes and invertebrates often did not include measurements for the same tissues (e.g., feathers, liver or muscle for birds). Also, it was necessary to use toxicity data for one species and apply these data to all species examined in the Bay. This approach leads to some uncertainty because one species may be less sensitive than another to the effects of DDT or Hg. The second limitation is for the contaminant data for the Bay of Fundy. Much less information is available for DDT than Hg and many measurements are for animals collected from the 1970s and 1980s.

Given that DDT use in North America was banned in the early 1970s, it is likely that current concentrations (and associated risks) are lower than those calculated herein. Similar trends could be expected for the older Hg data as well given that some controls on Hg emissions in North America were instituted in the 1980s. It is clear from the results of this study that more current assessments of contaminants in Bay of Fundy wildlife is needed, with perhaps the exception of Hg in some birds (data are available for 2006 for several species). However, all fish data and the limited invertebrate data are at least a decade old.

Table 1: Mercury chronic toxicity data for fish and birds

Common name	Species name	Group	Life Stage	Mode	Hg conc.	Hg form	Test duration	NOEC	LOEC	Effect Observed	Tissue type	Low Effect Residue (LER) (µg/g)	Reference
Walleye	*Stizostedion vitreum*	Fish	J	F	0.1 µg/g	MeHg	6 m	<0.04 µg/g	0.1 µg/g	decrease in GSI (more so in males) , testicular atrophy in low [MeHg]	whole	0.25	Friedmann et al. 1996
Striped mullet	*Mugil cephalus*	Fish	A	AQ	1 µg/L	MeHg	7 d	0 ug/L	1 µg/L	Fin regeneration-Development	whole	0.3	Weis and Weis 1978
Mussels	*Perna viridis*	Invert	-	AQ	25 µg/L	$HgCl_2$	14 d	Control	25 µg/L	Significant increases in: NH4N excretion Significant decrease in filtration rate, scope of growth and growth efficiency	whole	3.76	Krishnakumar 1990
Mallard ducks	*Anas platyrhynchos*	Bird	A/E	F	0.1 µg/g	MeHg	3 gen	Control	0.5 µg/g	Increased # eggs laid outside of nest, fewer sound eggs, decreased response to maternal call, longer approach time with respect to maternal call	egg	0.83	Heinz 1979
Mallard ducks	*Anas platyrhynchos*	Bird	A/E	F	0.1 µg/g	MeHg	3 gen	Control	0.5 µg/g	Increased # eggs laid outside of nest, fewer sound eggs, decreased response to maternal call, longer approach time with respect to maternal call	liver	1.33	Heinz 1979

J = Juvenile, A = Adult, E = Eggs, F = Food, AQ = Aqueous, m = months, d = days, gen = generations, conc. = concentration

Table 1 (continued): Mercury chronic toxicity data for fish and birds

Common name	Species name	Group	Life Stage	Mode	Hg conc.	Hg form	Test duration	NOEC	LOEC	Effect Observed	Tissue type	Low Effect Residue (LER) (µg/g)	Reference
Mallard ducks	*Anas platyrhynchos*	Bird	A/E	F	0.1 µg/g	MeHg	3 gen	Control	0.5 µg/g	Increased # eggs laid outside of nest, fewer sound eggs, decreased response to maternal call, longer approach time with respect to maternal call	kidney	1.65	Heinz 1979
Mallard ducks	*Anas platyrhynchos*	Bird	A/E	F	0.1 µg/g	MeHg	3 gen	Control	0.5 µg/g	Increased # eggs laid outside of nest, fewer sound eggs, decreased response to maternal call, longer approach time with respect to maternal call	muscle	0.77	Heinz 1979
Mallard ducks	*Anas platyrhynchos*	Bird	A/E	F	0.1 µg/g	MeHg	3 gen	Control	0.5 µg/g	increased # eggs laid outside of nest, fewer sound eggs, decreased response to maternal call, longer approach time with respect to maternal call	brain	0.59	Heinz 1979
Mallard ducks	*Anas platyrhynchos*	Bird	A/E	F	0.1 µg/g	MeHg	3 gen	Control	0.5 µg/g	increased # eggs laid outside of nest, fewer sound eggs, decreased response to maternal call, longer approach time with respect to maternal call	feathers	9.76	Heinz 1979

J =Juvenile, A = Adult, E = Eggs, F = Food, AQ = Aqueous, m = months, d = days, gen = generations, conc. = concentration

Table 2: DDT chronic toxicity data for fish and birds

Common name	**Species name**	**Group**	**Life Stage**	**Avg. size (g)**	**Test conditions**	**Food DDT form**	**Test duration (days)**	**NOEC mg/kg**	**LOEC mg/kg**	**Effect**
Black ducks	*Anas rubripes*	Birds	-	-	Food /corn	DDE	136 d	Control	10	Eggshell thinning, survival
Chinook salmon	*Onchorhynchus tshawytscha*	Fish	Fingerling	0.61g	Food	DDT	7d	6.4	37.5	Death

Common name	**Species name**	**Group**	**DDD Low Effect Residue (LER) µg/g-ww**	**DDE Low Effeect Residue (LER) µg/g-ww**	**DDT Low Effect Residue (LER) µg/g-ww**	**TDDT Low Effect Residue (LER) µg/g-ww**	**Total Residue Notes**	**Reference**
Black ducks	*Anas rubripes*	Birds	-	46	-	-	DDE	Longcore 1971
Chinook salmon	*Onchorhynchus tshawytscha*	Fish	0.71	0.1	2.84	3.65	DDD + DDE + *o,p* DDT	Buhler et al. 1969

Table 3: Water quality and tissue residue guidelines for mercury and DDT

Name	Form	Marine (ng/L)	Freshwater (ng/L)	Marine (µg/L)	Freshwater (µg/L)	LOAEL Marine (µg/L)	LOAEL Freshwater (µg/L)	Marine µg/kg	Notes	References
Water Quality Guidelines Hg for the protection of aquatic life (Environment Canada 2007)	THg	16	26	0.016	0.026	0.16	0.26	-	Guideline LOAEL/10	CCME Canadian Water Quality Guidelines for the Protection of Aquatic Life
Water Quality Guidelines Hg for the protection of aquatic life (Environment Canada 2007)	MeHg	NRG	4	NRG	0.004	NRG	0.04	-	Guideline LOAEL/10	CCME Canadian Water Quality Guidelines for the Protection of Aquatic Life
Canadian Tissue Residue Guidelines for the protection of wildlife consumers of aquatic biota (Environment Canada 2000)	MeHg	-	-	-	-	-	-	33.0	diet ww	CCME Canadian Tissue Residue Guidelines for the protection of wildlife consumers of aquatic biota
Interim Sediment Quality guidelines (ISQGs)	THg	-	-	-	-	-	-	130	marine dw	CCME Sediment Quality Guideline for the Protection of Aquatic Life
Probable Effect Levels (PELs)	THg	-	-	-	-	-	-	700	marine dw	CCME Sediment Quality Guideline for the Protection of Aquatic Life
Interim Sediment Quality guidelines (ISQGs)	DDT	-	-	-	-	-	-	1.19	dw	CCME Canadian Sediment Quality Guidelines for the Protection of Aquatic Life

NRG - No Recommended Guideline, ww - wet weight, dw - dry weight

Table 3 (continued): Water quality and tissue residue guidelines for mercury and DDT

Name	Form	Marine (ng/L)	Freshwater (ng/L)	Marine (µg/L)	Freshwater (µg/L)	LOAEL Marine (µg/L)	LOAEL Freshwater (µg/L)	Marine µg/kg	Notes	References
Interim Sediment Quality guidelines (ISQGs)	DDE	-	-	-	-	-	-	2.07	dw	CCME Canadian Sediment Quality Guidelines for the Protection of Aquatic Life 1999
Interim Sediment Quality guidelines (ISQGs)	DDD	-	-	-	-	-	-	1.22	dw	CCME Canadian Sediment Quality Guidelines for the Protection of Aquatic Life 1999
Probable Effect Levels (PELs)	DDT	-	-	-	-	-	-	4.77	dw	CCME Canadian Sediment Quality Guidelines for the Protection of Aquatic Life 1999
Probable Effect Levels (PELs)	DDE	-	-	-	-	-	-	374	dw	CCME Canadian Sediment Quality Guidelines for the Protection of Aquatic Life 1999
Probable Effect Levels (PELs)	DDD	-	-	-	-	-	-	7.81	dw	CCME Canadian Sediment Quality Guidelines for the Protection of Aquatic Life 1999
Canadian Tissue Residue Guidelines for the protection of wildlife consumers of aquatic biota (Environment Canada 1997)	TDDT	-	-	-	-	-	-	14	diet ww	Canadian Tissue Residue Guidelines for the Protection of Wildlife consumers of Aquatic Biota 1997

NRG - No Recommended Guideline, ww - wet weight, dw - dry weight

Table 4: Average mercury concentrations in birds and risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	$RQ_{LERs/10}$	Year	Reference	Sources of LER
Arctic tern	Bird	breast feathers	MeHg	9760	791	0.81	2006	Bond and Diamond 2009b	Bouton et al. 1999, Heinz 1979
Atlantic puffin	Bird	breast feathers	MeHg	9760	1634	**1.67**	2006	Bond and Diamond 2009b	Bouton et al. 1999, Heinz 1979
Common murre	Bird	breast feathers	MeHg	9760	1249	**1.28**	2006	Bond and Diamond 2009b	Bouton et al. 1999, Heinz 1979
Common tern	Bird	breast feathers	MeHg	9760	1619	**1.66**	2006	Bond and Diamond 2009b	Bouton et al. 1999, Heinz 1979
Leech's storm-petrel	Bird	breast feathers	MeHg	9760	5330	**5.46**	2006	Bond and Diamond 2009b	Bouton et al. 1999, Heinz 1979
Razorbill	Bird	breast feathers	MeHg	9760	1073	**1.10**	2006	Bond and Diamond 2009b	Bouton et al. 1999, Heinz 1979
Arctic tern	Bird	breast feathers	THg	9760	891	0.91	2006	Bond and Diamond 2009b	Bouton et al. 1999, Heinz 1979
Atlantic puffin	Bird	breast feathers	THg	9760	1805	**1.85**	2006	Bond and Diamond 2009b	Bouton et al. 1999, Heinz 1979
Common murre	Bird	breast feathers	THg	9760	987	**1.01**	2006	Bond and Diamond 2009b	Bouton et al. 1999, Heinz 1979
Common tern	Bird	breast feathers	THg	9760	1380	**1.41**	2006	Bond and Diamond 2009b	Bouton et al. 1999, Heinz 1979

Table 4 (continued): Average mercury concentrations in birds and risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	$RQ_{LERs/10}$	Year	Reference	Sources of LER
Leech's storm-petrel	Bird	breast feathers	THg	9760	4855	**4.97**	2006	Bond and Diamond 2009b	Bouton et al. 1999, Heinz 1979
Razorbill	Bird	breast feathers	THg	9760	1404	**1.44**	2006	Bond and Diamond 2009b	Bouton et al. 1999, Heinz 1979
Leech's storm-petrel	Bird	egg	MeHg	830	1008	**12.14**	2006	Bond and Diamond 2009b	Bouton et al. 1999, Heinz 1979
Arctic tern	Bird	egg	THg	830	220	**2.64**	2005-06	Bond and Diamond 2009b	Bouton et al. 1999, Heinz 1979
Atlantic puffin	Bird	egg	THg	830	234.25	**2.82**		Goodale et al. 2008, Bond and Diamond 2009a, Neil Burgess, pers. comm.	Bouton et al. 1999, Heinz 1979
Black guillemot	Bird	egg	THg	830	520	**6.27**	1998, 2001-06	Goodale et al. 2008	Bouton et al. 1999, Heinz 1979
Common eider	Bird	egg	THg	830	294	**3.54**	1998, 2001-06	Goodale et al. 2008, Bond and Diamond 2009a	Bouton et al. 1999, Heinz 1979
Common murre	Bird	egg	THg	830	247	**2.98**	2005-06	Bond and Diamond 2009a	Bouton et al. 1999, Heinz 1979
Common tern	Bird	egg	THg	830	124	**1.49**	1998, 2001-06	Goodale et al. 2008, Bond and Diamond 2009a	Bouton et al. 1999, Heinz 1979

Table 4 (continued): Average mercury concentrations in birds and risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	$RQ_{LERs/10}$	Year	Reference	Sources of LER
Double-crested cormorant	Bird	egg	THg	830	280	**3.37**	1998, 2001-06	Goodale et al. 2008	Bouton et al. 1999, Heinz 1979
Double-crested cormorant	Bird	egg	THg	830	166	**2.00**	2008	Neil Burgess, pers. comm.	Bouton et al. 1999, Heinz 1979
Glossy ibis	Bird	egg	THg	830	40	0.48	1998, 2001-06	Goodale et al. 2008	Bouton et al. 1999, Heinz 1979
Herring gull	Bird	egg	THg	830	80	0.96	1998, 2001-06	Goodale et al. 2008	Bouton et al. 1999, Heinz 1979
Herring gull	Bird	egg	THg	830	112	**1.35**	2008	Neil Burgess, pers. comm.	Bouton et al. 1999, Heinz 1979
Least tern	Bird	egg	THg	830	150	**1.81**	1998, 2001-06	Goodale et al. 2008	Bouton et al. 1999, Heinz 1979
Leech's storm-petrel	Bird	egg	THg	830	890.8	**10.73**	1998, 2001-06	Goodale et al, 2008, Bond and Diamond 2009a	Bouton et al. 1999, Heinz 1979
Leech's storm-petrel	Bird	egg	THg	830	531.0	**6.40**	2008	Neil Burgess, pers. comm.	Bouton et al. 1999, Heinz 1979
Piping plover	Bird	egg	THg	830	240	**2.89**	1998, 2001-06	Goodale et al. 2008	Bouton et al. 1999, Heinz 1979
Razorbill	Bird	egg	THg	830	433.7	**5.22**	1998, 2001-06	Goodale et al. 2008, Bond and Diamond 2009a	Bouton et al. 1999, Heinz 1979

Table 4 (continued): Average mercury concentrations in birds and risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	$RQ_{LERs/10}$	Year	Reference	Sources of LER
Willet	Bird	egg	THg	830	100	**1.20**	1998, 2001-06	Goodale et al. 2008	Bouton et al. 1999, Heinz 1979
Arctic tern	Bird	kidney	THg	1650	453	**2.75**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Black guillemot	Bird	kidney	THg	1650	491	**2.98**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Black-legged kittiwake	Bird	kidney	THg	1650	242	**1.47**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Bonaparte's gull	Bird	kidney	THg	1650	418	**2.53**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Common eider	Bird	kidney	THg	1650	358	**2.17**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Common tern	Bird	kidney	THg	1650	1505	**9.12**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Double-crested cormorant	Bird	kidney	THg	1650	5345	**32.39**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Herring gull	Bird	kidney	THg	1650	352	**2.13**	1988	Elliot et al. 1992	Bouton et al. 1999, Heinz 1979

Table 4 (continued): Average mercury concentrations in birds and risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	$RQ_{LERs/10}$	Year	Reference	Sources of LER
Herring gull	Bird	kidney	THg	1650	300	**1.82**	1988	Elliot et al. 1992	Bouton et al. 1999, Heinz 1979
Herring gull	Bird	kidney	THg	1650	350	**2.12**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Leech's storm-petrel	Bird	kidney	THg	1650	1213	**7.35**	1988	Elliot et al. 1992	Bouton et al. 1999, Heinz 1979
Arctic tern	Bird	liver	THg	1330	470	**3.53**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Black guillemot	Bird	liver	THg	1330	513	**3.86**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Black-bellied plover	Bird	liver	THg	1330	475	**3.57**	1990-91	Braune and Noble 2009	Bouton et al. 1999, Heinz 1979
Black-legged kittiwake	Bird	liver	THg	1330	372	**2.80**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Bonaparte's gull	Bird	liver	THg	1330	450	**3.38**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Common eider	Bird	liver	THg	1330	987	**7.42**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979

Table 4 (continued): Average mercury concentrations in birds and risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	$RQ_{LERs/10}$	Year	Reference	Sources of LER
Common tern	Bird	liver	THg	1330	1249	**9.39**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Double-crested cormorant	Bird	liver	THg	1330	3520	**26.47**	1988	Elliot et al. 1992	Bouton et al. 1999, Heinz 1979
Double-crested cormorant	Bird	liver	THg	1330	7048	**52.99**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Dunlin	Bird	liver	THg	1330	832	**6.25**	1990-91	Braune B and Noble G, 2009	Bouton et al. 1999, Heinz 1979
Greater yellowlegs	Bird	liver	THg	1330	809	**6.08**	1990-91	Braune B and Noble G, 2009	Bouton et al. 1999, Heinz 1979
Herring gull	Bird	liver	THg	1330	503	**3.78**	1988	Elliot et al. 1992	Bouton et al. 1999, Heinz 1979
Herring gull	Bird	liver	THg	1330	227	**1.71**	1988	Elliot et al. 1992	Bouton et al. 1999, Heinz 1979
Herring gull	Bird	liver	THg	1330	482	**3.62**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Leech's storm-petrel	Bird	liver	THg	1330	2419	**18.19**	1988	Elliot et al. 1992	Bouton et al. 1999, Heinz 1979

Table 4 (continued): Average mercury concentrations in birds and risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	$RQ_{LERs/10}$	Year	Reference	Sources of LER
Lesser yellowlegs	Bird	liver	THg	1330	818	**6.15**	1990-91	Braune and Noble 2009	Bouton et al. 1999, Heinz 1979
Red-necked phalarope	Bird	liver	THg	1330	225	**1.69**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Semipalmated plover	Bird	liver	THg	1330	356	**2.68**	1990-91	Braune and Noble 2009	Bouton et al. 1999, Heinz 1979
Semipalmated sandpiper	Bird	liver	THg	1330	703	**5.28**	1990-91	Braune and Noble 2009	Bouton et al. 1999, Heinz 1979
Short-billed dowitcher	Bird	liver	THg	1330	535	**4.02**	1990-91	Braune and Noble 2009	Bouton et al. 1999, Heinz 1979
Arctic tern	Bird	muscle	THg	770	89.0	**1.16**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Black guillemot	Bird	muscle	THg	770	113	**1.47**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Black-legged kittiwake	Bird	muscle	THg	770	37.0	0.48	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Bonaparte's gull	Bird	muscle	THg	770	75.0	0.97	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979

Table 4 (continued): Average mercury concentrations in birds and risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	$RQ_{LERs/10}$	Year	Reference	Sources of LER
Common eider	Bird	muscle	THg	770	153	**1.99**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Common tern	Bird	muscle	THg	770	166	**2.16**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Double-crested cormorant	Bird	muscle	THg	770	606	**7.87**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Herring gull	Bird	muscle	THg	770	101	**1.31**	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979
Red-necked phalarope	Bird	muscle	THg	770	46.0	0.60	1978-84	Braune 1987b	Bouton et al. 1999, Heinz 1979

Table 5: Average mercury concentrations in fish and marine mammals and their risk quotients (RQ) for the Bay of Fundy

Common Name	Group	Tissue type	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	RQ_{TRG}	$RQ_{LERs/10}$	Year	Reference	Sources of LER
Harbour porpoises	Mammal	muscle	THg	-	1054	-	-	1969-77	Gaskin et al. 1979	-
Atlantic herring	Fish	muscle	THg	300	8.92	0.3	0.30	1981	Braune 1987a	Weis and Weis 1978
Atlantic herring	Fish	whole body	THg	300	8.94	0.3	0.30	1981	Braune 1987a	Weis and Weis 1978
Atlantic salmon	Fish	muscle	THg	300	29.2	0.9	1.0	1994-1997	Sunderland et al. 2012	Weis and Weis 1978
Bluefin tuna	Fish	whole body	MeHg	300	495.7	**15.0**	**16.52**	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Bluefin tuna	Fish	whole body	THg	300	564.9	**17.1**	**18.83**	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Cod	Fish	whole body	MeHg	300	27.1	0.8	0.90	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Cod	Fish	whole body	THg	300	35	**1.1**	**1.17**	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Cunner	Fish	whole body	MeHg	300	75.3	**2.3**	**2.51**	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Cunner	Fish	whole body	THg	300	79.7	**2.4**	**2.66**	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Haddock	Fish	whole body	MeHg	300	18.3	0.6	0.61	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978

Table 5 (continued): Average mercury concentrations in fish and marine mammals and their risk quotients (RQ) for the Bay of Fundy

Common Name	Group	Tissue type	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	RQ_{TRG}	$RQ_{LERs/10}$	Year	Reference	Sources of LER
Haddock	Fish	whole body	THg	300	32.3	1.0	**1.08**	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Harbor pollock	Fish	muscle	THg	300	5.0	0.2	0.17	1978-1984	Braune and Gaskin 1987	Weis and Weis 1978
Herring	Fish	whole body	MeHg	300	40.3	**1.2**	**1.34**	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Herring	Fish	whole body	THg	300	47.5	**1.4**	**1.58**	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Herring brit	Fish	muscle	THg	300	4	0.1	0.13	1978-1984	Braune and Gaskin 1987	Weis and Weis 1978
Longhorn sculpin	Fish	muscle	THg	300	55-132	**1.7-4.0**	**1.83-4.40**	1970	Sunderland et al. 2012	Weis and Weis 1978
Mackerel	Fish	whole body	MeHg	300	17.4	0.5	0.58	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Mackerel	Fish	whole body	THg	300	22	0.7	0.73	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Pollock	Fish	whole body	MeHg	300	15.4	0.5	0.51	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Pollock	Fish	whole body	THg	300	18.7	0.6	0.62	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Spiny dogfish	Fish	whole body	MeHg	300	83.9	**2.5**	**2.80**	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Spiny dogfish	Fish	whole body	THg	300	99.3	**3.0**	**3.31**	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978

Table 5 (continued): Average mercury concentrations in fish and marine mammals and their risk quotients (RQ) for the Bay of Fundy

Common Name	Group	Tissue type	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	RQ_{TRG}	$RQ_{LERs/10}$	Year	Reference	Sources of LER
Spiny dogfish	Fish	muscle	THg	300	250	**7.6**	**8.33**	1993, 2007	Forsythe2008, Sunderland et al. 2012	Weis and Weis 1978
Swordfish	Fish	whole body	MeHg	300	294	**8.9**	**9.80**	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Swordfish	Fish	whole body	THg	300	416.4	**12.6**	**13.88**	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Thresher shark	Fish	whole body	MeHg	300	1426	**43.2**	**47.56**	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Thresher shark	Fish	whole body	THg	300	1472	**44.6**	**49.08**	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
White hake	Fish	whole body	MeHg	300	24	0.7	0.80	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
White hake	Fish	whole body	THg	300	29.5	0.9	0.98	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Winter flounder	Fish	whole body	MeHg	300	15.2	0.5	0.51	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Winter flounder	Fish	whole body	THg	300	21.1	0.6	0.70	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Witch flounder	Fish	muscle	THg	300	33-440	1.00-**13.33**	**1.10-14.67**	1970	Sunderland et al. 2012	Weis and Weis 1978
Yellowtail flounder	Fish	whole body	MeHg	300	23	0.7	0.78	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978
Yellowtail flounder	Fish	whole body	THg	300	26.9	0.8	0.90	2001-02	Gareth Harding, unpublished data	Weis and Weis 1978

Table 6: Average mercury concentrations in sediments and invertebrates and their risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type / Depth	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	RQ_{TRG}	$RQ_{LERs/10}$	RQ_{ISQG}	RQ_{PEL}	Year	Reference	Sources of LER	Notes
American lobster	Invert	digestive gland	THg	-	60.0	**1.8**	-	-	-	2001	Chou et al. 2004	-	-
American lobster	Invert	muscle	THg	-	29	0.88	0.08	-	-	1990-1996	Sunderland et al. 2012	-	-
Amphipod	Invert	-	THg	-	11.7	0.35	-	-	-	1983, 2001	Braune and Gaskin 1987, Sunderland et al. 2012	-	-
Amphipod	Invert	-	MeHg	-	1.06	0.03	-	-	-	2011	Sizmur et al. 2013	-	-
Amphipod	Invert	-	THg	-	4.1	0.1	-	-	-	2011	Sizmur et al. 2013	-	-
Blue mussel	Invert	soft tissue	THg	3760	32.5	1.0	0.09	-	-	2001	Chou et al. 2004	Krishnakumar et al. 1990	20-30 mm
Blue mussel	Invert	soft tissue	THg	3760	33.6	1.0	0.09	-	-	2001	Chou et al. 2004	Krishnakumar et al. 1990	30-50 mm
Blue mussel	Invert	soft tissue	THg	3760	29.2	0.9	0.08	-	-	2001-2007-08	Chou et al. 2004, LeBlanc et al. 2009, LeBlanc et al. 2009b	Krishnakumar et al. 1990	50+ mm
Blue mussel	Invert	-	THg	3760	26.7	0.81	0.09	-	-	1998	Sunderland et al. 2012	Krishnakumar et al. 1990	size unkown
Copepod	Invert	-	THg	-	4.3	0.13	-	-	-	1981	Braune 1987a	-	-

Table 6 (continued): Average mercury concentrations in sediments and invertebrates and their risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type/ Depth	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	RQ_{TRG}	$RQ_{LERs/10}$	RQ_{ISQG}	RQ_{PEL}	Year	Reference	Sources of LER	Notes
Euphausiids (*M. novegica*)	Invert	-	THg	-	6.0	0.18	-	-	-	1980-1983	Braune and Gaskin 1987	-	-
Euphausiids (*T. inermis*)	Invert	-	THg	-	3.0	0.09	-	-	-	1980-1983	Braune and Gaskin 1987	-	-
Oyster	Invert	-	THg	-	16.0	0.48	-	-	-	1998	Sunderland et al. 2012	-	-
Periwinkle	Invert	-	THg	-	30.0	0.91	-	-	-	1996-1998	Sunderland et al. 2012	-	-
Phytoplankton (25-63 µm)	Invert	-	THg	-	2.8	0.08	-	-	-	2000-2002	Sunderland et al. 2012	-	-
Polychaete (*Nephtys sp.)*	Invert	-	THg	-	7.3	0.22	-	-	-	1983-1984, 2001	Braune 1987a, Braune and Gaskin 1987, Sunderland et al. 2012	-	-
Polychaete (*Capitellidae sp.)*	Invert	-	MeHg	-	0.58	0.02	-	-	-	2011	Sizmur et al. 2013	-	-
Polychaete (*Capitellidae sp.)*	Invert	-	THg	-	5.3	0.16	-	-	-	2011	Sizmur et al. 2013	-	-
Polychaete (*Glyceridae* sp.)	Invert	-	MeHg	-	1.32	0.04	-	-	-	2011	Sizmur et al. 2013	-	-
Polychaete (*Maldanidae sp*.)	Invert	-	MeHg	-	6.76	0.20	-	-	-	2011	Sizmur et al. 2013	-	-

Table 6 (continued): Average mercury concentrations in sediments and invertebrates and their risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type/ Depth	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	RQ_{TRG}	$RQ_{LERs/10}$	RQ_{ISQG}	RQ_{PEL}	Year	Reference	Sources of LER	Notes
Polychaete (*Maldanidae sp.*)	Invert	-	THg	-	7.5	0.23	-	-	-	2011	Sizmur et al. 2013	-	-
Polychaete (*Paraonidae sp.*)	Invert	-	MeHg	-	2.93	0.09	-	-	-	2011	Sizmur et al. 2013	-	-
Polychaete (*Paraonidae sp.*)	Invert	-	THg	-	9.8	0.30	-	-	-	2011	Sizmur et al. 2013	-	-
Polychaete (*Spionidae sp.*)	Invert	-	MeHg	-	3.45	0.10	-	-	-	2011	Sizmur et al. 2013	-	-
Polychaete (*Spionidae sp.*)	Invert	-	THg	-	19.1	0.58	-	-	-	2011	Sizmur et al. 2013	-	-
Sea urchin	Invert	-	THg	-	26.7	0.81	-	-	-	1996-1997	Sunderland et al. 2012	-	-
Soft shelled clam	Invert	-	THg	-	50.0	**1.52**	-	-	-	1998	Sunderland et al. 2012	-	-
Sediment core, push core-Passamaquoddy Bay	Sediment	0-12 cm	MeHg	-	0.4	-	-	0.00	0.00	2001	Sunderland et al. 2004	-	dry weight
Sediment core, push core- Head of St. Croix Estuary	Sediment	0-12 cm	MeHg	-	0.7	-	-	0.01	0.00	2001	Sunderland et al. 2004	-	dry weight
Sediment-Belliveau Point	Sediment	-	THg	-	15.00	-	-	0.12	0.03	1997-02	Hung and Chmura 2006	-	dry weight
Sediment-Bocabeb	Sediment	-	THg	-	57.50	-	-	0.44	0.02	1997-02	Hung and Chmura 2006	-	dry weight

Table 6 (continued): Average mercury concentrations in sediments and invertebrates and their risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type/ Depth	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	RQ_{TRG}	$RQ_{LERs/10}$	RQ_{ISQG}	RQ_{PEL}	Year	Reference	Sources of LER	Notes
Sediment-Bocabeb	Sediment	-	THg	-	57.50	-	-	0.44	0.02	1997-02	Hung and Chmura 2006	-	dry weight
Sediment- Cape Enrage	Sediment	-	THg	-	10.5	-	-	0.08	0.01	1997-02	Hung and Chmura 2006	-	dry weight
Sediment- Dipper Harbour	Sediment	-	THg	-	27.5	-	-	0.21	0.04	1997-02	Hung and Chmura 2006	-	dry weight
Sediment-Gooseberry Cove	Sediment	18-67 cm	THg	-	17.5	-	-	0.13	0.02	2001	Chou et al. 2004	-	dry weight
Sediment- Inner Musquash Harbour	Sediment	8-12 cm	THg	-	12.5	-	-	0.10	0.08	2001	Chou et al. 2004	-	dry weight
Sediment-Lorneville	Sediment	-	THg	-	29.5	-	-	0.23	0.02	1997-02	Hung and Chmura 2006	-	dry weight
Sediment-Mouth of Musquash Harbour	Sediment	34-60 cm	THg	-	10	-	-	0.08	0.04	2001	Chou et al. 2004	-	dry weight
Sediment- Saint John Harbour	Sediment	-	THg	-	30	-	-	0.23	0.04	2001	Chou et al. 2004	-	dry weight
Sediment- St. Martins	Sediment	-	THg	-	22.5	-	-	0.17	0.03	1997-02	Hung and Chmura 2006	-	dry weight

Table 6 (continued): Average mercury concentrations in sediments and invertebrates and their risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type/ Depth	Hg form	Low Effect Residue (LER) ng/g-wet	Hg ng/g-wet	RQ_{TRG}	$RQ_{LERs/10}$	RQ_{ISQG}	RQ_{PEL}	Year	Reference	Sources of LER	Notes
Sediment – Wood Point	Sediment	-	THg	-	20.5	-	-	0.16	0.03	1997-02	Hung and Chmura 2006	-	dry weight
Sediment core, push core-Passamaquoddy Bay	Sediment	0-12 cm	THg	-	60.2	-	-	0.46	0.09	2001	Sunderland et al. 2004	-	dry weight
Sediment core, push core- Head of St. Croix Estuary	Sediment	0-12 cm	THg	-	238	-	-	**1.83**	0.34	2001	Sunderland et al. 2004	-	dry weight
Sediment, gravity core-Passamaquoddy Bay	Sediment	0-30 cm	THg	-	47.0	-	-	0.36	0.07	2001	Sunderland et al. 2004	-	dry weight
Sediment, gravity core-Passamaquoddy Bay	Sediment	40-100 cm	THg	-	10.7	-	-	0.08	0.02	2001	Sunderland et al. 2004	-	dry weight
Sediment, gravity core- Head of St. Croix River	Sediment	0-30 cm	THg	-	38.6	-	-	0.30	0.06	2001	Sunderland et al. 2004	-	dry weight
Sediment, gravity core- Head of St. Croix River	Sediment	40-80 cm	THg	-	18.4	-	-	0.14	0.03	2001	Sunderland et al. 2004	-	dry weight

Table 7: Average DDT concentrations in birds and their risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type Tested	Low Effect Residue (LER) µg/g-wet	ΣDDT µg/g-lipid	ΣDDT µg/g-wet	$RQ_{LER/10}$	Year (s)	Fish eating?	Sources of Data	Sources of LER
Double-crested cormorant	Bird	abdominal fat	-	-	162	-	1970s	Y	Zitko and Choi 1971	-
Herring gull	Bird	abdominal fat	-	-	1.70	-	1970s	Y	Zitko and Choi 1971	-
Black-bellied plover	Bird	carcass	-	0.49	-	-	1990-91		Braune and Noble 2009	-
Dunlin	Bird	carcass	-	0.17	-	-	1990-91		Braune and Noble 2009	-
Greater yellowlegs	Bird	carcass	-	0.50	-	-	1990-91	Y	Braune and Noble 2009	-
Lesser yellowlegs	Bird	carcass	-	6.19	-	-	1990-91	Y	Braune and Noble 2009	-
Semipalmated plover	Bird	carcass	-	3.79	-	-	1990-91		Braune and Noble 2009	-
Semipalmated sandpiper	Bird	carcass	-	0.09	-	-	1990-91		Braune and Noble 2009	-
Short-billed dowitcher	Bird	carcass	-	0.67	-	-	1990-91		Braune and Noble 2009	-
Black duck	Bird	egg	4.60	-	1.50	0.33	1970		Zitko and Choi 1971	Longcore 1971
Double-crested cormorant	Bird	egg	4.60	-	19.0	**4.13**	1970s	Y	Zitko et al. 1972, Zitko and Choi 1971	Longcore 1971
Guillemot	Bird	egg	4.60	-	4.35	0.95	1970s	Y	Zitko and Choi 1971	Longcore 1971
Herring gull	Bird	egg	4.60	-	4.25	0.92	1970s	Y	Zitko V., 1972	Longcore 1971
Double-crested cormorant	Bird	liver	-	-	4.16	-	1970s	Y	Zitko and Choi 1971	-
Greater shearwater	Bird	liver	-	-	0.86	-	1974	Y	Gaskin et al. 1978	-
Herring gull	Bird	liver	-	-	2.08	-	1970s	Y	Zitko et al. 1972, Zitko and Choi 1971	-
Sooty shearwater	Bird	liver	-	-	0.26	-	1974	Y	Gaskin et al. 1978	-

Table 7 (continued): Average DDT concentrations in birds and their risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type Tested	Low Effect Residue (LER) µg/g-wet	ΣDDT µg/g-lipid	ΣDDT µg/g-wet	$RQ_{LER/10}$	Year (s)	Fish eating?	Sources of Data	Sources of LER
Double-crested cormorant	Bird	muscle	-	-	8.40	-	1970s	Y	Zitko and Choi 1971	-
Greater shearwater	Bird	muscle	-	-	1.05	-	1974	Y	Gaskin et al. 1978	-
Herring gull	Bird	muscle	-	-	2.07	-	1970s	Y	Zitko et al. 1972, Zitko and Choi 1971	-
Sooty shearwater	Bird	muscle	-	-	0.34	-	1974	Y	Gaskin et al., 1978	-
Double-crested cormorant	Bird	subcutaneous fat	-	-	164	-	1970s	Y	Zitko and Choi 1971	-
Herring gull	Bird	subcutaneous fat	-	-	26.0	-	1970s	Y	Zitko and Choi 1971	-

Table 8: Average DDT concentrations in fish and marine mammals and their risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type	Low Effect Residue (LER) µg/g-wet	ΣDDT µg/g-wet	$RQ_{TRG,DDT}$	$RQ_{LER/10}$	Year (s)	Reference	Sources of LER
Common seal	Mammal	liver	-	12.65	**903.2**	-	1970s	Zitko 1975	-
Harbour porpoises	Mammal	blubber	-	6.34	**452.9**	-	1989-91	Westgate et al. 1997	-
North Atlantic right whale	Mammal	blubber	-	0.12	**8.3**	-	1988-89	Woodley 1991	-
Bluefin tuna	Fish	muscle	3.65	0.15	**10.7**	0.41	1970s	Zitko et al. 1972	Buhler et al. 1969
Cod	Fish	muscle	3.65	0.04	**2.9**	0.11	1970s	Zitko 197, Zitko and Choi 1971	Buhler et al. 1969
Herring	Fish	whole body	3.65	0.06	**4.3**	0.16	1970s	Zitko V., 1972	Buhler et al. 1969
Mackerel	Fish	muscle	3.65	0.14	**10.0**	0.38	1970s	Zitko et al. 1972, Zitko and Choi 1971, Zitko 1971	Buhler et al. 1969
Ocean perch	Fish	muscle	3.65	0.03	**2.1**	0.08	1970s	Zitko et al. 1972	Buhler et al. 1969
Plaice	Fish	muscle	3.65	0.01	0.7	0.03	1970s	Zitko et al. 1972	Buhler et al. 1969
Sea raven	Fish	muscle	3.65	0.32	**22.9**	0.88	1970s	Zitko et al. 1972	Buhler et al. 1969
Sea raven	Fish	viscera	-	0.30	**21.4**	**-**	1970s	Zitko et al. 1972	-
White hake	Fish	muscle	3.65	0.03	**2.1**	0.08	1970s	Zitko et al. 1972	Buhler et al. 1969
White shark	Fish	liver	-	441	**31500**	**-**	1970s	Zitko et al. 1972	-
White shark	Fish	muscle	3.65	0.48	**34.3**	**1.32**	1970s	Zitko et al. 1972	Buhler et al. 1969

Table 9: Average DDT concentrations in invertebrates and their risk quotients (RQ) for the Bay of Fundy

Common name	Group	Tissue type	ΣDDT µg/g-wet	$RQ_{TRG,DDT}$	Year (s)	Reference
American lobster	Invert	heptopancreas	1.59	**113**	1981	Zitko 1981
Clam	Invert	whole	0.00	0.0	1969	Sprague et al. 1969
Mussel	Invert	whole	0.09	**6.4**	1969	Sprague et al. 1969
Scallop	Invert	whole	0.03	**2.1**	1969	Sprague et al. 1969

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