Mercury concentrations in Double-crested Cormorant chicks across Canada

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Abstract

Mercury (Hg) biomagnifies in aquatic food chains and can reach high concentrations in fish-eating birds. Spatial patterns of Hg have been found in freshwater ecosystems across Canada for many taxa including fish and birds. However, it is often challenging to sample a representative population size of adult birds to monitor concentrations of contaminants over a large spatial scale. Moreover, adult birds can migrate and can show a contaminant profile that may not be representative of local resources. The aims of this study were (1) to determine if there was a spatial pattern of Hg in piscivorous birds, (2) to develop a model to estimate Hg concentrations in breeding adults using chicks as proxy, and (3) to develop predictive equations among non-lethal samples that representative of local resources in adults (blood and growing feathers). Double-crested Cormorant (*Phalacrocorax auritus*) chick growing feathers were sampled at 19 sites across Canada ($n = 106$). Adult tissues (freshly grown feathers; $n = 8-16$ per feather type and blood; $n = 160$) were sampled at five of those locations to establish correlations between age classes and between adult tissues. We found an increase in Hg concentrations with latitude up to 50°N followed by a decrease. There was a decrease in Hg concentrations from west to east, which contradicts previous studies. We found a good correlation of Hg concentrations between adults and chicks and among adult tissues. Our model showed that it is possible to estimate Hg concentrations in adults across Canada using chicks as proxy. Our study shows that chicks can be a suitable proxy for monitoring local mercury concentrations and that they are representative of adults.

Capsule: Concentrations of mercury in cormorant chicks are influenced by latitude

Keywords: geospatial patterns, biomonitor, monitoring, age class, aquatic birds

Introduction

Mercury (Hg) biomagnifies in aquatic food chains (Lavoie et al. 2013) and can reach toxic concentrations in upper-trophic-level organisms such as fish-eating birds (Depew et al. 2012, Wolfe et al. 1998). Hg is ubiquitous, but concentrations in wildlife vary spatially depending on the physico-chemistry of the waterbody (e.g., pH) and Hg input in the system (e.g., atmospheric deposition and point sources (Clayden et al. 2013, Kidd et al. 2012)). The trophic structure of a system will also define Hg concentrations in apex predators due to spatial variations in trophic levels (e.g., food chain length (Kidd et al. 2012, Lavoie et al. 2013, Vander Zanden and Fetzer 2007)). Spatial patterns of Hg have been found in freshwater ecosystems across North America for many taxa including fish and bird species and concentrations typically increase towards southeastern Canada and northeastern United States (Depew et al. 2013, Evers et al. 1998, Evers et al. 2003, Scheuhammer et al. 2001, Wente 2004). These patterns are best explained by high deposition of Hg and sulfate, low pH, and surrounding landscapes dominated by forests and/or wetlands (Depew et al. 2013, Driscoll et al. 2007, Evers et al. 2007).

Selecting a representative species to monitor concentrations of Hg over a large spatial scale represents a challenge. Characteristics for good aquatic monitoring species require that they are widespread, abundant, easy to access, easy to capture (and recapture where repeated measurements are necessary), have high territorial fidelity, are high in the food chain (e.g., piscivorous), and have a long lifespan (Evers et al. 1998). Low cost is also an asset in the choice of species used by monitoring agencies. The Double-crested Cormorant (*Phalacrocorax auritus*), an upper-trophic-level piscivorous bird, meet these requirements when nesting on the ground (Dorr et al. 2014, Lavoie et al. 2014, Lavoie et al. 2015). Double-crested cormorants

were chosen because chicks are nidicolous (altricial), which means that they require parental care after hatching, such as nourishment, incubation, and protection (Dorr et al. 2014). Cormorant chicks are therefore less prone to escaping when sampled and are available for a long period of time (3–4 weeks for ground nesters and 6–8 weeks for tree or cliff nesters (Dorr et al. 2014)). In addition, cormorant populations are widespread in North America and colonies generally contain many nests (Dorr et al. 2014), making it easier to sample birds without disturbing population sizes. Cormorants show high concentrations compared to other bird species and almost a third of the individuals are at moderate to severe risk to MeHg contamination with many individuals exceeding toxicity benchmarks (Ackerman et al. 2016b). Finally, cormorants embryos are categorized as showing low sensitivity to Hg contamination (Heinz et al. 2009). Yet, field studies in the Great Lakes showed toxic effects of mercury (oxidative stress (Gibson et al. 2014)) and PAHs (DNA mutations (King et al. 2014)) in cormorants. Monitoring resilient species means that if toxicity is found for those conservative species, more sensitive species are likely to show symptoms of toxicity. Considering the reasons enumerated above, we believe that cormorants are a good model species for monitoring and for establishing spatial patterns of contaminants and potentially other chemical tracers such as stable isotopes.

A potential major issue for monitoring studies using birds is that adults of many species migrate and several tissues can show a contaminant profile that may not be entirely representative of local resources of the monitored system (Lavoie et al. 2014). For instance, Hg that was assimilated from food sources at the overwintering area is transferred to the feathers grown during the overwintering period and sequestered as soon as the blood supply to the feathers atrophies. Concentrations measured in such feathers will therefore be representative of the overwintering area even if they were sampled elsewhere (e.g., at the breeding area (Lavoie et al.

2014)). More dynamic tissues (or tissues with fast turnover rate) such as blood and growing feathers will represent a more recent exposure to Hg. Chick tissues can be a good sampling unit because uptake of Hg originates mainly from local food resources (i.e., on breeding grounds) and tissues synthesized locally (e.g., blood or growing feathers) are not influenced by resources from distant ecosystems (Hughes et al. 1997, Stewart et al. 1997). In fish-eating birds, maternal transfer of mercury to eggs or to hatching chicks' internal tissues have been shown (Ackerman et al. 2011, Ackerman et al. 2016a) and this can bias Hg concentrations because it can come from a mixture of local sources (exogenous reserves) or distant source (endogenous reserves (Becker et al. 1993, Bond and Diamond 2010, Hughes et al. 1997)). However, the effect of maternal transfer is reduced after a few days as chicks grow (biomass dilution and depuration) and concentrations are subsequently defined by local dietary resources (Ackerman et al. 2011). Hg from chick blood is then transferred to growing feathers, which represent short-term local intake of Hg (Bearhop et al. 2000, Hughes et al. 1997). Another reason to use chicks for large-scale monitoring studies is that they are easier to catch than adults, which can drastically reduce time and cost related to sample size requirements.

The objectives of this study was were (1) to determine if there was a spatial pattern of Hg in piscivorous birds from selected breeding sites distributed across Canada, (2) to develop a model to estimate Hg concentrations in breeding adults using chicks as proxy, and (3) to develop predictive equations among non-lethal samples (blood and growing feathers) in adults.

Materials and methods

Sample Collection

Chicks

Double-crested Cormorant chick feathers were sampled from breeding sites across Canada through associations with other agencies to determine spatial patterns of mercury. The primary feather #1 (hereafter called P1) of chicks was sampled from 19 sites across Canada with four to ten chicks per site for a total sample size of 106 (Figure 1; Table 1) ranging from latitude 42.83 to 54.69°N and from longitude -111.79 to -61.93°W. Chicks were sampled from different nests at each site to keep independence between individuals. Cormorant chicks were caught from accessible nests (e.g., nest on the ground or in low trees). The tip of the growing P1 of the right wing was cut with a pair of scissor. The tip of the feather (which grows first) was sampled to ensure that the same developmental period was sampled among chicks and among colonies. Chick feathers were sampled because uptake of Hg originates almost exclusively from local food sources (i.e., on breeding grounds (Hughes et al. 1997)). In contrast, fully grown feathers in adult birds may contain Hg that was accumulated from resources of distant ecosystems (Lavoie et al. 2014, Lavoie et al. 2015). All feathers were put in paper envelopes until sample preparation. The maximum foraging radius of adult birds during the breeding period is approximately 40 km (Custer and Christine 1992) and a minimum distance of 80 km was therefore kept between colonies to ensure that food sources given by parents to their chicks were distinct among colonies.

Adults

Adults were caught at five locations in the Great Lakes area as described in Gibson et al. (2014). Blood ($n = 160$) and growing breast feathers ($n = 14$) were sampled as described in Lavoie et al. (2014, 2015). Briefly, approximately 3 ml of blood was sampled and stored in acid-washed polyethylene vials. Blood samples were stored on icepaks in the field for a maximum of 12 h, centrifuged to keep the cellular fraction (red blood cells) and frozen until analysis. Growing

breast feathers were sampled as a result of an artificial molt induced by sampling those same feathers 3–6 weeks earlier. Growing flight feathers (P1: $n = 8$; primaries: $n = 16$; secondaries: $n = 12$; Table 2) were sampled opportunistically throughout the summer by cutting the tip of growing feathers without cutting the blood supply in the sheath. Blood and regrown feathers represent short term exposure of Hg through diet (Bearhop et al. 2000, Monteiro and Furness 2001a, b, Nisbet et al. 2002), although carryover from overwintering locations is possible for birds exposed to high concentrations (Lavoie et al. 2014).

Sample preparation and mercury analysis

The rachis was cut from all feather samples to avoid any influence of blood supply in the rachis of regrowing feathers (Burger et al. 1992). Each vane of each feather was washed with acetone prior to Hg analysis (Burger and Gochfeld 2000). Red blood cells were freeze-dried prior to analysis (Gibson et al. 2014).

Mercury was analysed in red blood cells and growing feathers as described in Gibson et al. (2014) and in Lavoie et al. (2014). Briefly, samples were digested in a microwave oven with nitric acid, hydrogen peroxide, and deionized water. Total Hg analysis was done by cold vapour atomic fluorescence spectrometry following U.S. EPA method 1631 (U.S. EPA 2002). Certified reference materials from the Canadian National Research Council, Canada (DOLT–4 and DORM–3) and from the Institute for Reference Materials and Measurements, Geel, Belgium (human hair: BCR–397) were analyzed. Mean ($\pm$ SD) recoveries were $100.6 \pm 11.9\%$ ($n = 33$) for DOLT–4, $113.7 \pm 14.2\%$ ($n = 37$) for DORM–3 and $96.1 \pm 11.2\%$ ($n = 6$) for BCR–397 in adult tissues and $101.2 \pm 10.0\%$ ($n = 12$) for DOLT–4, and $110.8 \pm 13.2\%$ ($n = 7$) for DORM–3 in chick feathers. Samples THg concentrations are expressed in microgram per gram dry weight

($\mu\text{g/g d.w.}$) and are assumed to approximate MeHg concentrations in feathers and red blood cells of fish-eating birds (Bond and Diamond 2009, Kim et al. 1996, Lavoie et al. 2010).

Data analysis

A multiple linear regression (MLR) with chick Hg concentration as the dependent variable and latitude and longitude as independent variables was done. Model selection was done by Akaike Information Criterion corrected for small sample sizes (AICc) to determine the most parsimonious model using the Multi-Model Inference package *MuMIn* (Barton 2013) based on a set of candidate models of main factors and their interaction. When the difference in AICc (ΔAICc) between the best model and the second best model was high (>4) and when Akaike weight (w ; model probability) of the top model was high, the model output for the model with the lowest AIC was considered to be the most likely model for the dataset. To determine the empirical support of a model over another one, we calculated evidence ratios, which is the Akaike weight of top model over the weight of another model such as the 2nd best model or the null model (i.e. intercept only, (Burnham et al. 2011)).

We developed a model to predict adult Hg concentrations across Canada using concentrations from chicks. Adults ($n = 160$) and chicks ($n = 30$) were caught at five common locations, but could not be paired together because they had no parental link. We therefore generated all possible combinations of regressions between adult blood and chick P1 feathers within each of the five sites (e.g., an adult from site A could only be paired to a chick from site A). Sample sizes used for the model were 5–10 for chicks and 7–114 for adults. We ran a loop of 10,000 iterations of a reduced major axis (RMA) model including all five locations at each iteration. The RMA model was done with the package *lmodel2* (Legendre 2013) with 99 permutations with chick P1

feather concentration as the independent variable and adult blood concentration as the dependent variable. Growing feathers from adults could not be sampled at all five sites and could therefore not be used for the predictive loop model. RMA (Model II) was used as opposed to a simple linear regression (Model I) because both variables were random and were subjected to error. The average slope, intercept, and 2.5–97.5% confidence intervals (CI) of the slope and intercept were used to predict adult blood concentrations. To apply this model to other adult tissues, we ran RMA models between predicted blood concentrations as the independent variable and P1 ($n = 8$), primaries ($n = 16$), secondaries ($n = 12$), breast ($n = 14$), or any growing ($n = 39$) feathers as dependent variables. Hg concentrations were log-transformed for statistical tests to approximate normal distributions.

R software version 3.3.2 (R Core Team 2016) was used for geospatial data treatment, mapping, and statistical analyses.

Results

Spatial trends in chicks

Hg concentrations in chick P1 varied from 0.79 ± 0.43 (mean $\pm$ SD) $\mu\text{g/g}$ at Lake Pelletier (Quebec; 48.2128°N , -79.0519°W) to 8.3 ± 2.6 $\mu\text{g/g}$ at Lake Malartic (Quebec; 48.2560°N , -78.1042°W ; Table 1; Figure 1) and were related to latitude and longitude. A significant polynomial relation with latitude was observed with an increase in Hg concentrations up to approximately 50°N followed by a decrease ($R^2 = 0.28$, $F_{(1,103)} = 19.5$, $p < 0.001$; Figure 1). We found a decrease in Hg concentrations from west to east, but the model only explained 5% of the variation in Hg ($R^2 = 0.05$, $F_{(1,104)} = 4.9$, $p = 0.028$). When latitude (polynomial) and longitude

were taken together, the best model based on lowest AIC explained 31% of the variation in Hg (multiple linear regression: $F_{(3,102)} = 15.3$, $p < 0.001$):

$$\text{Log}_{10}(\text{Hg})_{\text{chick P1}} = 1.2(\text{latitude}) - 0.01(\text{latitude}^2) - 0.008(\text{longitude}) - 29.9; R^2 = 0.31 \quad (1)$$

The Akaike weight (i.e., model probability) of the best model was 0.83. The evidence ratio showed that the best model was 4.9 times more likely than the second best model and 15,000 times more likely than the null model.

Relationship between chicks and adults

The equation of the RMA loop model to predict adult blood Hg concentrations using chick P1 Hg concentrations in the form of $y = b$ (2.5–97.5% CI) $\times x + a$ (2.5–97.5% CI; Figure 2), where 2.5–97.5% CI are the lower (2.5%) and upper (97.5%) confidence intervals (CI) was:

$$\log_{10}(\text{Hg})_{\text{adult blood}} = 0.637 (0.10\text{--}2.08) \times \log_{10}(\text{Hg})_{\text{chick P1}} + 0.396 (-0.13\text{--}0.59); R^2 = 0.40 \quad (2)$$

The CIs showed that the slope was different than 0, but not different than the 1:1 relationship, whereas the intercept was not different than 0.

Relationships between adult tissues

Hg concentrations in regrown feathers were predicted across Canada using the predicted Hg concentration in adult blood (see equations in Table 3). Predicted adult blood varied from $2.1 \pm 0.7 \mu\text{g/g}$ at Lake Pelletier to $9.5 \pm 1.8 \mu\text{g/g}$ at Lake Malartic (Table 4; Figure 3a). For feathers, concentrations were highest in P1 regrown feathers (3.1 ± 2.2 to $70.8 \pm 29.4 \mu\text{g/g}$; Figure 3b) and lowest in breast regrown feathers (5.6 ± 1.6 to $19.0 \pm 3.0 \mu\text{g/g}$) at Lake Pelletier and Lake Malartic, respectively.

Discussion

Spatial trends

A spatial pattern mainly explained by latitude (polynomial relationship: increase in Hg concentrations up to 50°N followed by a decrease) and longitude (decrease from west to east) was observed. Our work contradicts other studies that found an increase in Hg concentrations from west to east (Depew et al. 2013, Evers et al. 1998, Evers et al. 2003, Scheuhammer et al. 2001, Wente 2004). Several reasons could explain the spatial pattern in this study and the discrepancy between our longitudinal gradient and other studies. First, concentrations of Hg in aquatic organisms are known to vary spatially with Hg inputs. Concentrations of Hg in mosquitoes (Diptera: Culicidae), Largemouth Bass (*Micropterus salmoides*) and Common Loon (*Gavia immer*) were found to be positively related to atmospheric Hg deposition across the U.S. (Hammerschmidt and Fitzgerald 2005, 2006). Hg accumulation in aquatic organisms are therefore greatly influenced by Hg atmospheric loadings and consequently from anthropogenic emission sources (Hammerschmidt and Fitzgerald 2006). Second, Hg concentrations in aquatic biota are greatly influenced by the physico–chemistry of the waterbody. For instance, low pH and high concentrations of ligands, sulfate, and nutrients are central chemical characteristics for promoting Hg methylation in freshwater systems and are associated with increased Hg bioavailability at the water–primary production interface (Dittman and Driscoll 2009, Kidd et al. 2012, Ullrich et al. 2001, Watras et al. 1998). Low water pH has been linked with high Hg concentrations in invertebrates (Chetelat et al. 2011, Clayden et al. 2014) and apex predators such as fish and fish–eating birds (Burgess and Meyer 2008). Methylmercury also binds strongly to dissolved organic carbon (DOC), which can act as a vector of MeHg transport from the catchment area to lakes (Braaten et al. 2014) and can increase concentrations in biota in some

cases. Third, the west to east gradient observed in other studies reaches northeastern U.S or southeastern Canada where pH, sulfate and forest type are optimal for Hg bioaccumulation and where Hg hotspots are observed (Depew et al. 2013, Evers et al. 2007, Evers et al. 1998, Evers et al. 2003, Scheuhammer et al. 2001, Wentz 2004). Our study is lacking sites in these regions (except for a marine site in Magdalen Islands; 47.2603°N, -61.9311°W with low Hg concentration $2.0 \pm 1.2 \mu\text{g/g}$) and it is possible that a wider spatial range would lead to a different conclusion. Finally, the trophic structure of a system can also define Hg concentrations in apex predators due to spatial variations in food chain lengths (Kidd et al. 2012, Lavoie et al. 2013, Vander Zanden and Fetzer 2007). We observed that adults in Lake Ontario fed low trophic-level species to their chicks such as Alewife (*Alosa pseudoharengus*; trophic level (TL; mean $\pm$ standard error from Fishbase (Froese and Pauly 2016)) = 3.4 ± 0.3) and Round Goby (*Neogobius melanostomus*; TL = 3.3 ± 0.1), while adults in Lake Nipissing fed high trophic level species to their chicks such as Walleye (*Sander vitreus*; TL = 4.5 ± 0.0) and Muskellunge (*Esox masquinongy*; TL = 4.5 ± 0.3). Consequently, chicks in Lake Ontario ($1.9 \pm 0.6 \mu\text{g/g}$; $n = 25$) had lower feather Hg concentrations than in Lake Nipissing ($5.7 \pm 1.4 \mu\text{g/g}$; $n = 5$; $t = -5.8$; $p = 0.003$). We believe that spatial patterns are defined by a combination of the factors mentioned above although we cannot identify which ones are the most important ones in this case.

Relationship between chicks and adults

We found that cormorant chick feathers are a good monitoring tool for large-scale Hg assessments. Chicks of other species have been successfully used for local indicators of Hg pollution or for spatial analyses (Ackerman and Eagles-Smith 2009, Becker et al. 1993, Hughes et al. 1997, Stewart et al. 1997). Cormorants are particularly relevant because they are widespread and abundant across North America (Dorr et al. 2014) and chicks are relatively easy

to capture (pers. obs.). In addition, there was a low cost related to the collection of chick samples across Canada because collaborations were made for the sampling (see *Materials and Methods*). Chicks Hg concentrations were found to be representative of those of adults as observed by the linear relationship (Figure 2; Equation 2). Our predictive chick–adult model only explained 40% of the variation, but this is considered as a large effect size according to Cohen (1992). Discrepancies in concentration between chicks and adults can be mainly explained by the sampling design. Chicks were not taken from the same nests as adults and were paired through random iterations. The pairing is therefore at the scale of the colony rather than the nest (i.e., chick–female pairing). In addition, discrepancies could be due to age class differences in bioenergetics and toxicokinetics (e.g., growth dilution (Ackerman et al. 2011, Monteiro and Furness 2001a, b)), types of prey consumed (i.e., adults can bring food to their chicks that may differ than what they eat (Barrett et al. 2007)), and sources of contaminants (i.e., adults may carry contaminants from their migratory habitat (Lavoie et al. 2014); see below). While the match in concentrations between adults and chicks is imperfect, chick values can still be used for spatial patterns assessment of bird colonies because the relative differences among sites should remain regardless of the age class or tissue used. Monitoring cormorant chick feathers could therefore allow the determination of spatial trends and hotspots across North America. In addition, feather sampling of chicks is non–lethal contrary to internal tissues or eggs sampling, which is particularly relevant for endangered species.

Chick feathers is not the optimal choice for monitoring toxicological risks compared to adult blood or eggs (Ackerman et al. 2016b). However, we believe that they can be very useful for the identification of spatial patterns and hotspots for the following reasons. First, chick feathers are relatively easy to obtain and large–scale sampling and large sample sizes can be achieved

through collaborations. Second, chicks have limited mobility and are flightless for several weeks and are consequently easier to catch than adults. Third, feather sampling requires little training contrary to blood sampling and collaborations are therefore easier. Finally, shipping feathers across the country or between countries does not require any sample preparation or preservation contrary to blood, eggs or other internal tissues that require proper packing, freezing or drying. This is especially important if samples are retained at the customs for inspection during shipping. Chick feathers are therefore a useful monitoring tool for assessment of large-scale spatial patterns and hotspots.

A potential problem related to the use of chicks is the maternal transfer of Hg. Strong positive relationships have been found between female tissues (e.g., blood) and their eggs or chicks (Ackerman et al. 2016a, Burgess and Meyer 2008, Heinz et al. 2010, Kenow et al. 2015). Because adults are migratory, tissues Hg concentrations may represent a mixture of local resources (exogenous reserves; e.g., breeding area) or distant resources (endogenous reserves; e.g., wintering area (Becker et al. 1993, Hughes et al. 1997, Lavoie et al. 2014)). For species where chicks are covered with down at hatching (e.g., gulls), those feathers are indicative of the mercury in eggs and therefore of the mother (i.e., maternal transfer (Ackerman et al. 2016b)). Females transfer Hg from this mixture of different areas to the eggs, which are then transferred to chicks down. However, cormorants are born naked (no down or feathers) and down growth is only completed after 1–2 weeks (Dorr et al. 2014). The impact of maternal transfer decreases after a few days as chicks grow (biomass dilution and depuration) and local dietary resources subsequently determine concentrations (Ackerman et al. 2011). Hg from blood is then transferred to chicks' growing feathers, which represent short-term local intake of Hg (Hughes et al. 1997). Cormorant chick primary feathers start to grow by 16–19 days old and they are approximately

2.5 cm long by 21–23 days old (Dorr et al. 2014). Maternal transfer to cormorant chick primary feathers (P1) in our study is therefore unlikely and is believed to represent local dietary and contamination sources.

Relationships between adult tissues

We found good relationships between blood and different types of growing feathers in adults ($R^2 = 0.69\text{--}0.88$), which enables inter-tissues predictions of non-lethal samples. Eagles-Smith et al (2008) found strong relationships among internal tissues (e.g., blood, muscle, liver), but weaker relationships between internal tissues and feathers (head and breast). However, feathers sampled in Eagles-Smith et al (2008) were fully grown feathers that may indicate a mixture of resources provenance. Feathers in our study were regrown feathers, which are good indicators of short-term Hg intake (Bearhop et al. 2000, Monteiro and Furness 2001a, b, Nisbet et al. 2002).

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References

- Ackerman, J.T., and Eagles-Smith, C.A. 2009. Integrating Toxicity Risk in Bird Eggs and Chicks: Using Chick Down Feathers To Estimate Mercury Concentrations in Eggs. *Environ. Sci. Technol.* **43**(6): 2166-2172.
- Ackerman, J.T., Eagles-Smith, C.A., and Herzog, M.P. 2011. Bird mercury concentrations change rapidly as chicks age: toxicological risk is highest at hatching and fledging. *Environ. Sci. Technol.* **45**(12): 5418-5425.
- Ackerman, J.T., Eagles-Smith, C.A., Herzog, M.P., and Hartman, C.A. 2016a. Maternal transfer of contaminants in birds: mercury and selenium concentrations in parents and their eggs. *Environ. Pollut.* **210**: 145-154.
- Ackerman, J.T., Eagles-Smith, C.A., Herzog, M.P., Hartman, C.A., Peterson, S.H., Evers, D.C., Jackson, A.K., Elliott, J.E., Vander Pol, S.S., and Bryan, C.E. 2016b. Avian mercury exposure and toxicological risk across western North America: A synthesis. *Sci. Total Environ.* **568**: 749-769.
- Barrett, R.T., Camphuysen, K., Anker-Nilssen, T., Chardine, J.W., Furness, R.W., Garthe, S., Huppopp, O., Leopold, M.F., Montevecchi, W.A., and Veit, R.R. 2007. Diet studies of seabirds: A review and recommendations. *ICES J. Mar. Sci.* **64**(9): 1675-1691.
- Barton, K. 2013. MuMIn: Multi-model inference. R package version 1.9.13. URL: <http://CRAN.R-project.org/package=MuMIn> (accessed April 13, 2014).

346 Bearhop, S., Ruxton, G.D., and Furness, R.W. 2000. Dynamics of mercury in blood and feathers
347 of great skuas. *Environ. Toxicol. Chem.* **19**(6): 1638-1643.

348 Becker, P.H., Furness, R.W., and Henning, D. 1993. Mercury dynamics in young Common Tern
349 (*Sterna hirundo*) chicks from a polluted environment. *Ecotoxicology* **2**(1): 33-40.

350 Bond, A.L., and Diamond, A.W. 2009. Total and methyl mercury concentrations in seabird
351 feathers and eggs. *Arch. Environ. Contam. Toxicol.* **56**(2): 286-291.

352 Bond, A.L., and Diamond, A.W. 2010. Nutrient allocation for egg production in six Atlantic
353 seabirds. *Canadian Journal of Zoology* **88**(11): 1095-1102.

354 Braaten, H.F.V., de Wit, H.A., Fjeld, E., Rognerud, S., Lydersen, E., and Larssen, T. 2014.
355 Environmental factors influencing mercury speciation in Subarctic and Boreal lakes. *Sci.*
356 *Total Environ.* **476–477**: 336-345.

357 Burger, J., and Gochfeld, M. 2000. Metal levels in feathers of 12 species of seabirds from
358 Midway Atoll in the northern Pacific Ocean. *Sci. Total Environ.* **257**(1): 37-52.

359 Burger, J., Nisbet, I.C.T., and Gochfeld, M. 1992. Metal levels in regrown feathers: assessment
360 of contamination on the wintering and breeding grounds in the same individuals. *J.*
361 *Toxicol. Environ. Health* **37**(3): 363-374.

362 Burgess, N.M., and Meyer, M.W. 2008. Methylmercury exposure associated with reduced
363 productivity in common loons. *Ecotoxicology* **17**(2): 83-91.

364 Burnham, K.P., Anderson, D.R., and Huyvaert, K.P. 2011. AIC model selection and multimodel
365 inference in behavioral ecology: some background, observations, and comparisons.
366 *Behav. Ecol. Sociobiol.* **65**(1): 23-35.

367 Chetelat, J., Amyot, M., and Garcia, E. 2011. Habitat-specific bioaccumulation of
368 methylmercury in invertebrates of small mid-latitude lakes in North America. Environ.
369 Pollut. **159**(1): 10-17.

370 Clayden, M., Kidd, K., Chetelat, J., Hall, B., and Garcia, E. 2014. Environmental, geographic
371 and trophic influences on methylmercury concentrations in macroinvertebrates from
372 lakes and wetlands across Canada. Ecotoxicology **23**(2): 273-284.

373 Clayden, M.G., Kidd, K.A., Wyn, B., Kirk, J.L., Muir, D.C.G., and O'Driscoll, N.J. 2013.
374 Mercury biomagnification through food webs is affected by physical and chemical
375 characteristics of lakes. Environ. Sci. Technol. **47**: 12047–12053.

376 Cohen, J. 1992. A power primer. Psychol. Bull. **112**(1): 155.

377 Custer, T.W., and Christine, B. 1992. Feeding flights of breeding Double-crested Cormorants at
378 two Wisconsin colonies. J. Field Ornithol. **63**(2): 203-211.

379 Depew, D.C., Basu, N., Burgess, N.M., Campbell, L.M., Devlin, E.W., Drevnick, P.E.,
380 Hammerschmidt, C.R., Murphy, C.A., Sandheinrich, M.B., and Wiener, J.G. 2012.
381 Toxicity of dietary methylmercury to fish: Derivation of ecologically meaningful
382 threshold concentrations. Environ. Toxicol. Chem. **31**(7): 1536-1547.

383 Depew, D.C., Burgess, N.M., and Campbell, L.M. 2013. Modelling mercury concentrations in
384 prey fish: Derivation of a national-scale common indicator of dietary mercury exposure
385 for piscivorous fish and wildlife. Environ. Pollut. **176**: 234-243.

386 Dittman, J.A., and Driscoll, C.T. 2009. Factors influencing changes in mercury concentrations in
387 lake water and yellow perch (*Perca flavescens*) in Adirondack lakes. Biogeochemistry
388 **93**(3): 179-196.

- Dorr, B.S., Hatch, J.J., and Weseloh, D.V. 2014. Double-crested Cormorant (*Phalacrocorax auritus*) In The Birds of North America Online, Poole, A. and Gill, F. Eds. Cornell Lab of Ornithology, Ithaca, NY.
- Driscoll, C.T., Han, Y.J., Chen, C.Y., Evers, D.C., Lambert, K.F., Holsen, T.M., Kamman, N.C., and Munson, R.K. 2007. Mercury contamination in forest and freshwater ecosystems in the Northeastern United States. *Bioscience* **57**(1): 17-28.
- Eagles-Smith, C.A., Ackerman, J.T., Adelsbach, T.L., Takekawa, J.Y., Miles, A.K., and Keister, R.A. 2008. Mercury correlations among six tissues for four waterbird species breeding in San Francisco Bay, California, USA. *Environ. Toxicol. Chem.* **27**(10): 2136-2153.
- Evers, D.C., Han, Y.J., Driscoll, C.T., Kamman, N.C., Goodale, M.W., Lambert, K.F., Holsen, T.M., Chen, C.Y., Clair, T.A., and Butler, T. 2007. Biological mercury hotspots in the northeastern United States and southeastern Canada. *Bioscience* **57**(1): 29-43.
- Evers, D.C., Kaplan, J.D., Meyer, M.W., Reaman, P.S., Braselton, W.E., Major, A., Burgess, N., and Scheuhammer, A.M. 1998. Geographic trend in mercury measured in Common Loon feathers and blood. *Environ. Toxicol. Chem.* **17**(2): 173-183.
- Evers, D.C., Taylor, K.M., Major, A., Taylor, R.J., Poppenga, R.H., and Scheuhammer, A.M. 2003. Common Loon Eggs as Indicators of Methylmercury Availability in North America. *Ecotoxicology* **12**(1): 69-81.
- Froese, R., and Pauly, D. 2016. FishBase. World Wide Web electronic publication. www.fishbase.org, version (10/2016).
- Gibson, L.A., Lavoie, R.L., Bissegger, S., Campbell, L.M., and Langlois, V.S. 2014. A positive correlation between mercury and oxidative stress-related gene expression (*GPX3* and

411 *GSTM3*) is measured in female Double-crested Cormorant blood. *Ecotoxicology* **23**:
412 1004-1014.

413 Hammerschmidt, C.R., and Fitzgerald, W.F. 2005. Methylmercury in mosquitoes related to
414 atmospheric mercury deposition and contamination. *Environ. Sci. Technol.* **39**(9): 3034-
415 3039.

416 Hammerschmidt, C.R., and Fitzgerald, W.F. 2006. Methylmercury in freshwater fish linked to
417 atmospheric mercury deposition. *Environ. Sci. Technol.* **40**(24): 7764-7770.

418 Heinz, G.H., Hoffman, D.J., Klimstra, J.D., and Stebbins, K.R. 2010. Predicting mercury
419 concentrations in mallard eggs from mercury in the diet or blood of adult females and
420 from duckling down feathers. *Environ. Toxicol. Chem.* **29**(2): 389-392.

421 Heinz, G.H., Hoffman, D.J., Klimstra, J.D., Stebbins, K.R., Kondrad, S.L., and Erwin, C.A.
422 2009. Species differences in the sensitivity of avian embryos to methylmercury. *Arch.*
423 *Environ. Contam. Toxicol.* **56**(1): 129-138.

424 Hughes, K.D., Ewins, P.J., and Clark, K.E. 1997. A comparison of mercury levels in feathers and
425 eggs of osprey (*Pandion haliaetus*) in the North American Great Lakes. *Arch Environ*
426 *Contam Toxicol* **33**(4): 441-452.

427 Kenow, K.P., Meyer, M.W., Rossmann, R., Gray, B.R., and Arts, M.T. 2015. Influence of in ovo
428 mercury exposure, lake acidity, and other factors on common loon egg and chick quality
429 in Wisconsin. *Environ. Toxicol. Chem.* **34**(8): 1870-1880.

430 Kidd, K.A., Clayden, M., and Jardine, T.D. 2012. Bioaccumulation and biomagnification of
431 mercury in food webs. *In Environmental Chemistry and Toxicology of Mercury. Edited*
432 *by G. Liu, Y. Cai and N. O'Driscoll. John Wiley & Sons, Inc., Hoboken, NJ. pp. 453-499.*

Kim, E.Y., Murakami, T., Saeki, K., and Tatsukawa, R. 1996. Mercury levels and its chemical form in tissues and organs of seabirds. *Arch. Environ. Contam. Toxicol.* **30**(2): 259-266.

King, L.E., de Solla, S.R., Small, J.M., Sverko, E., and Quinn, J.S. 2014. Microsatellite DNA mutations in double-crested cormorants (*Phalacrocorax auritus*) associated with exposure to PAH-containing industrial air pollution. *Environ Sci Technol* **48**(19): 11637-11645.

Lavoie, R.A., Baird, C.J., King, L.E., Kyser, K., Friesen, V.L., and Campbell, L. 2014. Contamination of mercury during the wintering period influences concentrations at breeding sites in two migratory piscivorous birds. *Environ. Sci. Technol.* **48**: 13694–13702

Lavoie, R.A., Hebert, C.E., Rail, J.F., Braune, B., Yumvihoze, E., Hill, L.G., and Lean, D.R.S. 2010. Trophic structure and mercury distribution in a Gulf of St. Lawrence (Canada) food web using stable isotope analysis. *Sci. Total Environ.* **408**(22): 5529-5539.

Lavoie, R.A., Jardine, T.D., Chumchal, M.M., Kidd, K.A., and Campbell, L. 2013. Biomagnification of mercury in aquatic food webs: a worldwide meta-analysis. *Environ. Sci. Technol.* **47**(23): 13385–13394.

Lavoie, R.A., Kyser, T.K., Friesen, V.L., and Campbell, L.M. 2015. Tracking Overwintering Areas of Fish-Eating Birds to Identify Mercury Exposure. *Environ. Sci. Technol.* **49**(2): 863-872.

Legendre, P. 2013. lmodel2: Mpdel II Regression. R package version 1.7-2. URL: <http://CRAN.R-project.org/package=lmodel2> (accessed September 2, 2016).

Monteiro, L.R., and Furness, R.W. 2001a. Kinetics, dose-response, and excretion of methylmercury in free-living adult Cory's shearwaters. *Environ. Sci. Technol.* **35**(4): 739-746.

Monteiro, L.R., and Furness, R.W. 2001b. Kinetics, dose-response, excretion, and toxicity of methylmercury in free-living Cory's shearwater chicks. *Environ. Toxicol. Chem.* **20**(8): 1816-1823.

Nisbet, I.C.T., Montoya, J.P., Burger, J., and Hatch, J.J. 2002. Use of stable isotopes to investigate individual differences in diets and mercury exposures among common terns *Sterna hirundo* in breeding and wintering grounds. *Mar. Ecol. Prog. Ser.* **242**: 267-274.

R Core Team. 2016. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. [online] URL: <http://www.R-project.org>. Version 3.3.2.

Scheuhammer, A.M., Perrault, J.A., and Bond, D.E. 2001. Mercury, Methylmercury, and Selenium Concentrations in Eggs of Common Loons (*Gavia Immer*) From Canada. *Environ. Monit. Assess.* **72**(1): 79-94.

Stewart, F.M., Phillips, R.A., Catry, P., and Furness, R.W. 1997. Influence of species, age and diet on mercury concentrations in Shetland seabirds. *Mar. Ecol. Prog. Ser.* **151**: 237-244.

U.S. EPA. 2002. Method 1631: Revision E, mercury in water by oxidation, purge and trap, and cold vapor atomic fluorescence spectrometry. *In* U.S. Environmental Protection Agency: Washington, USA. URL: <http://www.epa.gov> (accessed May 13, 2010).

Ullrich, S.M., Tanton, T.W., and Abdrashitova, S.A. 2001. Mercury in the aquatic environment: a review of factors affecting methylation. *Crit. Rev. Environ. Sci. Technol.* **31**(3): 241-293.

Vander Zanden, M.J., and Fetzer, W.W. 2007. Global patterns of aquatic food chain length. *Oikos* **116**(8): 1378-1388.

478 Watras, C.J., Back, R.C., Halvorsen, S., Hudson, R.J.M., Morrison, K.A., and Wente, S.P. 1998.
 479 Bioaccumulation of mercury in pelagic freshwater food webs. *Sci. Total Environ.* **219**(2-
 480 3): 183-208.

481 Wente, S.P. 2004. A statistical model and national dataset for partitioning fish tissue mercury
 482 concentration between spatio-temporal and sample characteristic effects. United States
 483 Geological Survey Scientific Investigation Report 2004-5199, 15p.

484 Wolfe, M.F., Schwarzbach, S., and Sulaiman, R.A. 1998. Effects of mercury on wildlife: a
 485 comprehensive review. *Environ. Toxicol. Chem.* **17**(2): 146-160.

486

487 Table 1. Mercury (Hg) concentrations (mean $\pm$ standard deviations; SD) in Double-crested Cormorant (*Phalacrocorax auritus*) chick
488 primary 1 feather (P1) from 19 locations across Canada.

Area	Location	Code	Province	P1 ^a	<i>n</i>	Latitude	Longitude	Habitat
Alberta	Beaver Lake	BICH	AB	3.3 $\pm$ 0.5	7	54.691	-111.788	Lake
James Bay	Akimiski Strait	AKI	ON	5.9 $\pm$ 1.7	5	53.049	-82.163	Coastal
Lake Erie	Mohawk Island	MOH	ON	1.8 $\pm$ 1.3	5 ^b	42.834	-79.524	Lake
Lake Huron	Chantry Island	CHAN	ON	3.9 $\pm$ 1.1	10	44.490	-81.403	Lake
	Middle Grant Island	MIDD	ON	3.3 $\pm$ 1.7	5	46.137	-83.324	Lake
	Nottawasaga Island	NOTT	ON	3.1 $\pm$ 1.0	5	44.536	-80.259	Lake
Lake Nipissing	Gull Island	NIP	ON	5.7 $\pm$ 1.4	5 ^b	46.167	-79.594	Lake
Lake Ontario	Hamilton Harbour	HH	ON	1.5 $\pm$ 0.3	10 ^b	43.284	-79.794	Lake
	Scoth Bonnet Island	SBI	ON	1.4 $\pm$ 0.3	5 ^b	43.899	-77.543	Lake
	Snake Island	SNK	ON	2.7 $\pm$ 0.5	5	44.191	-76.543	Lake
	Tommy Thompson Park	TTP	ON	2.1 $\pm$ 0.2	5	43.623	-79.340	Lake
St. Lawrence River	Bergin Island	BERG	ON	3.7 $\pm$ 1.2	5 ^b	45.014	-74.860	River
	Lake St-Pierre	LSP	QC	6.7 $\pm$ 0.4	4	46.198	-72.849	Lake
Abitibi	Lake Castagnier	CAS	QC	5.4 $\pm$ 1.6	5	48.729	-77.764	Lake
	Lake De Montigny	MON	QC	7.2 $\pm$ 1.4	6	48.144	-77.907	Lake
	Lake Malartic	MAL	QC	8.3 $\pm$ 2.6	5	48.256	-78.104	Lake
	Lake Pelletier	PEL	QC	0.79 $\pm$ 0.4	4	48.213	-79.052	Lake
Magdalen Islands	Îlots du Portage	IMAD	QC	2.0 $\pm$ 1.2	5	47.260	-61.931	Coastal
Saskatchewan	Reed Lake	REED	SK	8.1 $\pm$ 1.1	5	50.400	-107.069	Lake

489 ^a Primary 1 (P1) feathers were sampled at all locations except at Reed Lake where P2 were sampled. ^b Chicks used to pair with adults
490 in the predictive loop model.

491 Table 2. Mercury (Hg) concentrations (mean $\pm$ SD) in Double-crested Cormorant (*Phalacrocorax auritus*) adult blood and regrown
 492 feathers at 5 locations.

Location ^a	Blood			<i>n</i> ^b	Breast			<i>n</i>	P1			<i>n</i>	Primaries			<i>n</i>	Secondaries			<i>n</i>	Other			<i>n</i>
BERG	6.1	±	2.1	7																17.4			1	
HH	3.1	±	1.8	114	7.4	±	3.0	14	6.6	±	4.1	6	6.1	±	3.2	12	7.2	±	3.8	9	5.7	±	2.9	25
MOH	3.1	±	1.2	16									17.8			1	8.6			1	8.9	±	6.2	4
NIP	6.2	±	1.5	14																				
SBI	4.7	±	1.2	9					26.9	±	14.1	2	21.3	±	13.8	3	19.7	±	1.3	2	14.6	±	9.5	9

493 ^a Codes are the same as in Table 1. ^b Adult samples used to pair with chicks in the predictive loop model
 494

495 Table 3. Relationship between log₁₀–transformed mercury (Hg) concentrations in Double–crested Cormorant (*Phalacrocorax auritus*)
 496 adult regrown feathers (log₁₀Hg_{regrown feathers}) as dependent variables and adult blood (log₁₀Hg_{blood}) as the independent variable. ^a

Type of regrown feather	<i>n</i>	<i>R</i> ²	<i>b</i>	CI 2.5–97.5%	<i>b</i> = 0	<i>b</i> = 1	<i>a</i>	CI 2.5–97.5%	<i>a</i> = 0	<i>p</i>
P1	8	0.88	2.09	1.50–3.22	N	N	-0.21	-0.81–0.10	Y	<0.001
Primary (any)	16	0.85	1.93	1.54–2.50	N	N	-0.11	-0.40–0.08	Y	<0.001
Secondary (any)	12	0.77	1.29	0.89–1.96	N	Y	0.25	-0.09–0.45	Y	<0.001
Breast	14	0.78	0.81	0.57–1.12	N	Y	0.49	0.36–0.59	N	<0.001
Any ^b	39	0.69	1.65	1.33–2.10	N	N	-0.02	-0.25–0.14	Y	<0.001

497 ^a Equation from a reduced major axis model (model II): log₁₀(Hg)_{regrown feather} = *b* x log₁₀(Hg)_{blood} + *a*. Hypotheses that the slope equals
 498 0 (*b* = 0) or 1 (*b* = 1) or that the intercept equals 0 (*a* = 0) are based on inclusion of the 2.5% to 97.5% confidence intervals (CI).

499 ^b When several types of regrown feathers were sampled from an individual, selection was done (in descending order of priority) to: P1
 500 > primary (any) > secondary (any) > breast.

501

502 Table 4. Predicted Hg in Double-crested Cormorant (*Phalacrocorax auritus*) adult blood and regrown feathers (mean $\pm$ SD)

Location ^a	Blood ^b	Regrown feathers ^c					
		Breast	Primaries	P1	Secondaries	Any	
BICH	5.3 $\pm$ 0.5	11.9 $\pm$ 0.9	19.6 $\pm$ 3.5	20.3 $\pm$ 3.9	15.4 $\pm$ 1.8	15.1 $\pm$ 2.3	
AKI	7.7 $\pm$ 1.4	16.0 $\pm$ 2.3	40.8 $\pm$ 14.1	45.1 $\pm$ 16.9	25.0 $\pm$ 5.8	28.1 $\pm$ 8.3	
MOH	3.5 $\pm$ 1.5	8.4 $\pm$ 2.9	10.0 $\pm$ 9.0	10.0 $\pm$ 9.9	9.3 $\pm$ 5.4	8.2 $\pm$ 6.3	
CHAN	5.9 $\pm$ 1.0	13.0 $\pm$ 1.7	24.6 $\pm$ 8.3	26.0 $\pm$ 9.6	17.8 $\pm$ 3.9	18.2 $\pm$ 5.2	
MIDD	5.2 $\pm$ 1.6	11.6 $\pm$ 3.0	20.1 $\pm$ 12.7	21.1 $\pm$ 14.5	15.2 $\pm$ 6.3	15.1 $\pm$ 8.1	
NOTT	5.1 $\pm$ 1.1	11.5 $\pm$ 2.0	18.7 $\pm$ 7.2	19.3 $\pm$ 8.0	14.7 $\pm$ 4.0	14.4 $\pm$ 4.9	
NIP	7.5 $\pm$ 1.2	15.7 $\pm$ 2.1	38.4 $\pm$ 12.0	42.3 $\pm$ 14.3	24.1 $\pm$ 5.1	26.7 $\pm$ 7.2	
HH	3.2 $\pm$ 0.5	8.0 $\pm$ 1.0	7.6 $\pm$ 2.1	7.3 $\pm$ 2.1	8.2 $\pm$ 1.5	6.7 $\pm$ 1.6	
SBI	3.1 $\pm$ 0.5	7.7 $\pm$ 0.9	7.0 $\pm$ 2.0	6.6 $\pm$ 2.1	7.7 $\pm$ 1.5	6.2 $\pm$ 1.5	
SNK	4.7 $\pm$ 0.5	10.8 $\pm$ 1.0	15.6 $\pm$ 3.5	15.8 $\pm$ 3.8	13.2 $\pm$ 2.0	12.4 $\pm$ 2.4	
TTP	4.0 $\pm$ 0.2	9.4 $\pm$ 0.5	11.1 $\pm$ 1.3	10.9 $\pm$ 1.4	10.5 $\pm$ 0.8	9.3 $\pm$ 0.9	
BERG	5.7 $\pm$ 1.2	12.5 $\pm$ 2.2	22.9 $\pm$ 9.4	24.2 $\pm$ 10.7	16.9 $\pm$ 4.7	17.1 $\pm$ 6.0	
LSP	8.3 $\pm$ 0.3	17.1 $\pm$ 0.5	46.5 $\pm$ 3.3	51.8 $\pm$ 3.9	27.6 $\pm$ 1.3	31.6 $\pm$ 1.9	
CAS	7.2 $\pm$ 1.4	15.2 $\pm$ 2.4	35.9 $\pm$ 13.6	39.3 $\pm$ 16.1	22.9 $\pm$ 5.8	25.2 $\pm$ 8.1	
MON	8.7 $\pm$ 1.1	17.7 $\pm$ 1.8	51.1 $\pm$ 12.4	57.5 $\pm$ 15.2	29.2 $\pm$ 4.7	34.2 $\pm$ 7.1	
MAL	9.5 $\pm$ 1.8	19.0 $\pm$ 3.0	61.8 $\pm$ 23.6	70.8 $\pm$ 29.4	32.9 $\pm$ 8.3	40.0 $\pm$ 13.0	
PEL	2.1 $\pm$ 0.7	5.6 $\pm$ 1.6	3.5 $\pm$ 2.3	3.1 $\pm$ 2.2	4.7 $\pm$ 2.1	3.4 $\pm$ 1.9	
IMAD	3.7 $\pm$ 1.5	8.8 $\pm$ 3.0	10.8 $\pm$ 7.4	10.8 $\pm$ 7.9	9.9 $\pm$ 5.0	8.8 $\pm$ 5.4	
REED	9.4 $\pm$ 0.8	18.9 $\pm$ 1.3	59.5 $\pm$ 9.7	67.7 $\pm$ 12.0	32.4 $\pm$ 3.6	38.9 $\pm$ 5.4	

503 ^a Codes are the same as in Table 1. ^b Predicted from chick P1 feathers using Equation 2. ^c Predicted from adult blood using equations
504 from Table 3.

505

506 Figure captions

507

508 Figure 1. Spatial distribution of Double-crested Cormorant (*Phalacrocorax auritus*) chick mercury (Hg) concentrations across
509 Canada. Black bars represent mean measured Hg concentrations and white bars represent standard deviations (SD).

510

511 Figure 2. All possible combinations of reduced major axis (RMA) between chicks and adults with average equation shown with solid
512 line and 10,000 iterations shown with grey lines. Average with lower and upper confidence intervals (2.5–97.5%) are shown in the
513 equation.

514

515 Figure 3. Spatial distribution of Double-crested Cormorant (*Phalacrocorax auritus*) mercury (Hg) concentrations in A) adult blood
516 predicted from chick P1 (Equation 2) and B) adult P1 predicted from adult blood (see equation in Table 3). Black bars represent mean
517 measured Hg concentrations and white bars represent standard deviations (SD).

518

519 Figure 1.

520

521

522 Figure 2.

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524