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Individual variability in contaminants and physiological status in a resident Arctic seabird species

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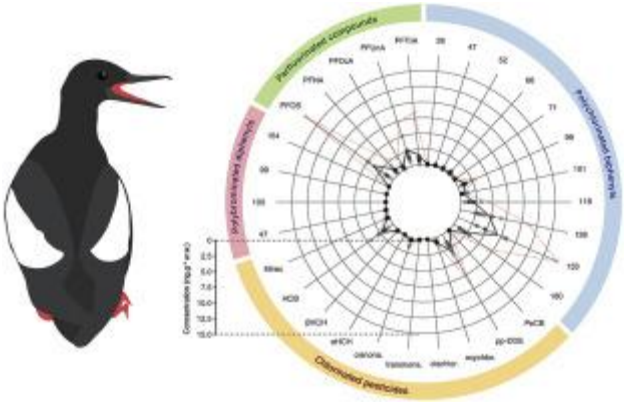
Abstract :

While migratory seabirds dominate ecotoxicological studies within the Arctic, there is limited knowledge about exposure and potential effects from circulating legacy and emerging contaminants in species who reside in the high-Arctic all year round. Here, we focus on the case of the Mandt's Black guillemot (Cepphus grylle mandtii) breeding at Kongsfjorden, Svalbard (79.00°N, 11.66°E) and investigate exposure to legacy and emerging contaminants in relation to individual physiological status, i.e. body condition, oxidative stress and relative telomere length. Despite its benthic-inshore foraging strategy, the Black guillemot displayed overall similar contaminant concentrations in blood during incubation (∑PCB11 (15.7 ng/g w.w.) > ∑PFAS5 (9.9 ng/g w.w.) > ∑Pesticides9 (6.7 ng/g w.w.) > ∑PBDE4 (2.7 ng/g w.w.), and Hg (0.3 µg/g d.w.) compared to an Arctic migratory seabird in which several contaminant-related stress responses have been observed. Black guillemots in poorer condition tended to display higher levels of contaminants, higher levels of reactive oxygen metabolites, lower plasmatic antioxidant capacity, and shorter telomere lengths; however the low sample size restrict any strong conclusions. Nevertheless, our data suggests that nonlinear relationships with a threshold may exist between accumulated contaminant concentrations and physiological status of the birds. These findings were used to build a hypothesis to be applied in future modelling for describing how chronic exposure to contaminants may be linked to telomere dynamics.

Graphical abstract



Highlights

► The Arctic resident Black guillemot showed overall similar exposure to contaminants during incubation compared to an Arctic migratory seabird in which several contaminant-related stress responses have been observed. ► Black guillemots in poorer condition tended to display higher levels of contaminants and shorter telomere lengths. ► Our data suggests that nonlinear relationships with a threshold may exist between accumulated contaminant concentrations and physiological status of the birds.

Keywords : black guillemot, pollutants, Polar Regions, oxidative stress, seabirds, telomeres

1. Introduction

A seabird's contaminant burden at a given time can be viewed as the product of cumulative processes driven by small or large differences in foraging strategies, physiology and life history. As a result, individuals foraging high in the food web, on lipid-rich prey - or in areas with higher inputs of contaminants often present the highest concentrations of contaminants (Muir *et al.,* 1992; de March *et al.,* 1998; Dietz *et al.*, 2000). The growing body of research on Arctic seabirds demonstrates that species with higher concentrations of contaminants experience sub-lethal effects, such as lower breeding success, reduced return rate and higher stress responses (e.g. Erikstad *et al.,* 2013; Tartu *et al.,* 2014a; Bustnes *et al.,* 2015). Yet, the fact that the toxicological effects do not depend solely on contaminant exposure, but are also influenced by life history, physiology, and other environmental stressors (Bourgeon *et al.*, 2012; Goutte *et al.*, 2014) often obscure the relationships between contaminant exposure and health impacts. Consequently, two populations - or species - may exhibit different negative effects of contaminants, despite having similar contaminant body burdens (Bustnes *et al.,* 2015). Most seabirds nesting in the Arctic migrate to lower latitudes after breeding and experience different contaminant exposure at their overwintering sites compared to their Arctic breeding grounds (Baert *et al.,* 2013; Fort *et al.,* 2014); however species that reside in the high Arctic all year round remain understudied and so does their response to the current multi-stress scenarios in Polar Regions.

The Mandt's Black guillemot (*Cepphus grylle mandtii*) is known to be one of the few seabird species that overwinter in the Arctic, as it's movements are limited to open water near the sea ice margins close to its breeding site (Brown, 1985; Divoky *et al.,* 2016). While this seabird's life history exposes it to various contaminants that have deposited in the Arctic via long-range transport/deposition mechanisms (Vorkamp *et al.,* 2004 & 2015; Braune *et al.,* 2015 & 2016; Peck *et al.,* 2016), the Black guillemot is considered to be sensitive to current and future climate alterations due to its dependency on the sea ice marginal zone in both the breeding (Divoky *et al.,* 2015) and non-breeding seasons (Divoky *et al.*, 2016). Typically, the Black guillemot will alternate between the use of nearshore waters in the breeding season and offshore waters during non-breeding season foraging mostly on Arctic cod (Hobson, 1993), which could differ in contamination and quality of prey (i.e. lipid content and antioxidants). During the breeding season, it rarely feeds more than ten kilometres away from the nest site; while chicks

are fed almost exclusively on benthic fish, adults have a mixed diet of both benthic fish and invertebrates (Cairns, 1987a & b; Barrett *et al.*, 2002).

The few exposure studies on black guillemots have used invasive and/or lethal sampling matrices or tissues that reflect time-interval of contaminants exposure outside the breeding seasons (e.g. liver, muscles, eggs, feathers) (e.g. Koistinen *et al.*, 1995; Appelquist *et al.*, 1983; Borgå *et al.*, 2001 & 2005 & 2007; Olafsdottir *et al.*, 2005; Vorkamp *et al.*, 2015). Thus, the impact of exposure from blood circulating contaminants on health parameters during sensitive life history events (e.g. breeding season) in this resident, nearshore Arctic species remains unknown.

Exposure to contaminants has the potential to cause an imbalance in the anti-oxidant defence system through increased levels of oxidative stress (Constantini *et al.*, 2014; Wielsøe *et al.*, 2015). Here contaminants may play two different roles: a) as an actual threat through energy competition resulting from biotransformation and maintenance (e.g. Durant *et al.*, 2007), and/or b) as a "blurring" factor by triggering the hypothalamic-pituitary-adrenalin axis (HPA) (Tartu *et al.*, 2014a), *i.e.* due to their endocrine disruptive properties, or by interfering with the antioxidant defence system itself. Both mechanisms may result in increased oxidative stress and ultimately reduced health status over time.

A new and promising cellular biomarker in ecotoxicology is telomere dynamics. Telomeres are repetitive, non-coding DNA sequences located at the end of each chromosome, whose length is reduced during cell division (i.e. through aging) - and through oxidative stress (see Blackburn, 2000; Von Zglinicki, 2002; Epel *et al.*, 2004; Monaghan & Haussmann, 2006, Reichert & Stier, 2017). Thus, telomere length depends on hereditary factors, early-life conditions and life-history trade-offs (Beaulieu *et al.*, 2011; Schull *et al.*, 2016) and should be informative as to integrated stress during an individual's lifetime (Horn *et al.*, 2010). In addition, telomere length is a strong predictor of longevity (Barrett *et al.*, 2013). Long-lived species, such as seabirds, are able to decrease the rate of telomere reduction through enzymatic activity (Blackburn, 2005; Haussmann *et al.*, 2007), and telomere length is proposed to be a predictor of intrinsic individual quality (Bauch *et al.*, 2013; Young *et al.*, 2017). Recently, Blévin *et al.* (2016

112 & 2017a) found associations between contaminants (i.e. perfluorinated compounds, *oxy*-
113 chlordane) and relative telomere length in breeding black-legged kittiwakes (*Rissa tridactyla*).

114 The main objective of the present study is to investigate how exposure to both legacy and
115 emerging contaminants may influence physiological status in an understudied resident Arctic
116 seabird species. This is attempted by relating circulating concentrations of both legacy and
117 emerging contaminants in blood to individual physiological status defined by body condition,
118 stress levels and relative telomere length in black guillemots.

2. Materials and methods

2.1 Sampling

Our study was carried out in Kongsfjorden, Svalbard (79.00°N, 11.66°E), on three different black guillemot nest sites (Juttaholmen, Midtholmen, Observasjonsholmen) during the breeding season of 2015 (June-July), with the necessary permissions from the Governor of Svalbard (RIS ID 10290) and Norwegian Animal Welfare authorities (FOTS ID 7666). We used snares to catch black guillemots (n = 15) resting on rocks outside of their nest during incubation phase. Of these, 11 birds were successfully sampled for blood (i.e. minimum 2 mL, maximum < 10 % of body mass) from their brachial wing vein, using heparinized syringes. Blood samples were kept cold (< 4°C) and in the dark, before being centrifuged and stored at -20°C. All captured birds were biometrically measured (i.e. body mass, wing length, bill and skull size) and inspected for brood patch.

2.2 Contaminant analyses

We measured mercury (Hg) in red blood cells (~ 0.5 mL) of 11 black guillemots. Hg was determined using in-house methods: NILU-U-117 “Procedure for sample digestion by use of high performance microwave reactor (UltraClave)” and NILU-U-65 “Procedure for determination of Hg in biological samples using Could Vapor Atomic Fluorescence Spectrophotometry (CV-AFS)”. Aliquots of 0.20-0.35 g of red blood cells were added 5 mL concentrated supra pure HNO_3 and 3 mL deionized water (MilliQ) and digested at 250^{o}C for 15 minutes, using UltraClave (Milestone, Italy). Blank samples and certified reference material (CRM) were digested in the same run as the blood samples. After digestion, samples, blank samples and CRM were diluted to 50 mL. From the extracts, subsamples of 25 mL were further diluted to 50 mL before 5 ml BrCl were added. Determination of Hg was performed using CV-AFS from Tekran, Canada according to method US-EPA-1631.

In the statistical analysis, we also included published contaminant data from reported in Haarr *et al.* (2017) on the same black guillemots individual as in present study. These contaminant data include concentrations of polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethane (DDTs) and its metabolites, isomers of hexachlorocyclohexane

147 (HCH), hexachlorobenzene (HCB), chlordanes, polybrominated diphenyl ethers (PBDEs),
148 perfluorinated compounds (PFAS) that was extracted from plasma (1 mL) using protocols
149 described in detail in Blévin *et al.* (2017 a & b) and Haarr *et al.* (2017).

150 **2.3 Isotopic signatures in black guillemots**

151 Signatures of stable isotopes are used to estimate individual foraging behaviour in two-
152 dimensions: pelagic-benthic and low-high trophic level feeding. The use of carbon and nitrogen
153 stable isotope ratios (∂^{13}C and ∂^{15}N in ‰) is an established practice for assessing carbon source
154 (e.g. pelagic vs. benthic) and relative positions in the food web (i.e. high or low trophic levels)
155 (Hobson *et al.*, 1995). Red blood cells (~200 µL) from black guillemots (n = 11) were freeze-
156 dried and analysed for heavy and light isotopes of nitrogen and carbon at the University of
157 Windsor, Canada; as described in Fisk *et al.* (2001).

158 **2.4 Oxidative stress, telomere length and body condition in black guillemots**

159 We estimated oxidative stress levels in plasma of black guillemots using well-established
160 methods on derivatives of reactive oxygen metabolites (d-ROM) in plasma and total plasma
161 antioxidant capacity (Oxy), described in detail by Schull *et al.* (2016). In short, we measured the
162 amount of hyperoxides (d-ROM test, Diacron International) in 8 µl of plasma for each bird,
163 (expressed as mg H_2O_2 equivalent dl^{-1}) with duplicates for each sample. We used the Oxy
164 adsorbent test (Diacron International, Grosseto, Italy) to quantify the ability to buffer massive
165 oxidation (i.e. hydroperoxide acid) in 5 µl of 1:100 diluted plasma, for each bird with duplicates
166 for each sample. The variation between individuals was 5.5% and 5.5% for d-ROM and Oxy,
167 respectively (only one plate was run, thus no inter-plate repeatability was assessed).

168 We extracted DNA using Nucleospin Blood QuickPure Kit (Macherey-Nagel, Düren
169 Germany) and estimated the relative length of telomeres by qPCR using a well-established
170 protocol for King penguin (*Aptenodytes patagonicus*) (Stier *et al.*, 2014; Le Vaillant *et al.*, 2015;
171 Reichert *et al.*, 2015). Chromosomic sex of individuals was determined from red blood cells
172 extracted DNA following Sambrook *et al.* (1989) protocol.

173 In order to understand how successfully a bird interacts with its environment, we used a
174 body condition measure to estimate its supply of energy (i.e. fat and protein content) (Labocha &

Hayes, 2012). Since larger birds are obviously heavier than smaller birds, the variation in mass generally correlates with size, while departures from this pattern are explained by inter-individual differences in body condition. Therefore, we calculated a Scale Mass Index - SMI, using the biometric measure with the highest correlation coefficient with body mass in our data (i.e. bill length) as length measure to scale the mass accordingly (see Peig & Green 2009):

$$SMI_{individual} = \text{body mass}_{individual} \times (bill_{mean}/bill_{individual}) \wedge coef_{\text{body mass}\sim\text{beak}}$$

2.5 Statistical analyses

Statistical analysis was conducted in R 3.4.2 (R Development Core Team, 2016). One black guillemot individual was excluded due to issues during chemical analyses, thus leaving us with a final dataset of ten birds.

The Limit of Detection for contaminants (LOD) was set as the mean of the blanks added by three times the standard deviation of the blanks (Armbruster & Pry, 2008). Values below LOD – or non-detects – were treated using distributional methods and imputation by maximum likelihood (Baccarelli *et al.*, 2005). For each contaminant, we assumed a log-normal distribution. Non-detects were treated using three different steps: 1) the most likely distribution was fitted based on both censored and non-censored data using a simple Expectation-Maximisation algorithm; 2) the minimum number of non-censored points (i.e. cut-off) necessary for a robust inference of the log-normal distribution parameters was estimated by simulations, and filtered accordingly; 3) for contaminants with a sufficiently low number of non-detects, multiple imputation by maximum likelihood was performed using a specified interval (i.e. zero to LOD).

Contaminant profiles are mixtures of many contaminants so to extract robust indicators that reliably compare individuals in regression analyses, we need to reduce variation to a lower number of dimensions. We thus first collapsed contaminants into groups depending on their chemical structure, in order to account for differences in weights (i.e. functional relationships with response variables) between different chemical groups. Secondly, since contaminants within the same chemical group may have different dose-dependency (i.e. toxic potential), we need to scale their relative contribution to focus on the shape and not the size of the data (Greenacre, 2017). Hence, we preferred to use an equally weighted index that rank individuals based on their

exposure to different chemical groups and total contaminants load. This approach does not account for differences in toxic potency and functional relationships within chemical groupings; however we consider it to address more correctly the underlying theoretical assumptions, and the approach itself produces an index that provide a more transparent interpretation when used as a predictor, compared to principle components for example. Thus, we created a score index for each chemical group (i.e. iPCB, iPesticides, iPBDE, iPFAS and iHg) and one for total contaminant load (iCON), where each individual is given a summed score relative to the mean of the sample population for each contaminant:

$$\text{Index Contaminant group X} = \sum(x_i / \overline{x})$$

where x_i are the values of each of the individual chemical groups in individual *i*, $\overline{x}$ represents the mean concentration of the individual chemical over the entire sample population within the given chemical group (X).

We applied different generalized linear models (GLMs) to test potential relationships between the contaminant groups and the response variables as well as residual plot analysis to assess the validity of the model. If the residuals showed non-random patterns, we tried different families of transformations, g(y), until the residuals plots of the regression g(y) = beta_0 + ∑ beta X were consistent with independent Gaussian error.

3. Results

3.1 Contaminants and isotopic signatures in black guillemots

Concentrations of Hg in red blood cells of black guillemots was measured to 0.32 ± 0.8 ug/g d.w. Treatment of LOD censored values resulted in detection of all 11 targeted PCB congeners, but only 10 of the 17 organochlorine pesticides, 5 of 20 perfluorinated compounds, and 4 of 10 PBDEs, in plasma of black guillemots (see Supplementary Table 1). ∑PCBs was the dominating compound group (dominated by PCB153 > PCB138 > PCB180), followed by ∑PFAS (mainly PFOS > PFUnA) > ∑Pesticides (mainly HCB > p,p'-DDE) > ∑PBDEs (mainly PBDE47) (see Fig. 1).

The black guillemots show a high dispersion in trophic positions (∂15N ranging from 12.0 – 13.5‰), while carbon isotopic signatures show a clear benthic signal and overall low dispersion - except one extreme individual ($\partial^{13}C$ = -21.5‰). This female presented also higher lipid content and was one the two individuals with highest contaminant concentrations, highest stress levels and shortest telomeres. As we did not have the sufficient information (unknown breeding status) to exclude her from the statistical analysis, we kept this individual in further analysis (see Discussion).

3.2 Relationships among contaminants, isotopic signatures and physiological status in black guillemots

We created four score indices based on the chemical groupings where each bird was given a score relative to the mean of the population per chemical group (i.e. iPCB, iPesticides, iPBDE, iPFAS and iHg) and one for total contaminant load (i.e. iCON). Correlation tests (Spearman's test) between the indices showed that birds with high scores on PCB also had high scores on the other contaminants. However, we found no significant correlations among iPBDE, iPFAS and iHg, or between iHg and iPesticides (see Supplementary Fig 2.). All score indices, except iHg, correlated significantly with iCON.

No significant relationship among isotopic signatures, lipid content and physiological status measures (for p-values see results from Spearman's correlation test in Supplementary Fig. 1) was detected. We found a significant positive correlation between body condition (SMI) and telomere length (RTL) and a significant negative correlation among iPesticides and SMI and RTL (see p-values from Spearman's correlation test in Supplementary Fig 1).

Our low sample size prevented us from conducting a thorough evaluation of the relationship between contaminants and physiological status in black guillemots. However, the data provided insight into these relationships and showed a consistent tendency with several variables driven by the two most contaminated individuals (see Fig 2). It should be noted that one individual (female) obscured this tendency; the individual with the shortest telomeres was in the modal of lower contaminant concentrations.

3.3 From contaminants to telomere length

Initial evaluation of our data through the residual plot analysis revealed a nonlinear relationship between iCON, SMI, Oxy, d-ROM and RTL (see Fig 2). The observed patterns imply that the average effect of the predictor on the response variable increases for moderate contaminant concentrations, but after a certain value is reached, the response approaches a saturation level (K) that needs to be determined. This suggests that an S-shaped function would be appropriate and we opted for a regression model of the form:

$$Y_K = K\frac{e^{(\beta_0+\beta_1 X)}}{1+e^{(\beta_0+\beta_1 X)}}$$

This function describes all nonlinear effects found in the data better than other simple nonlinear functions such as quadratic or exponential. This parametric family of functions describes all nonlinear effects found in our variables and is flexible enough to provide a good fit in all situations in our dataset. In this equation, Y represents the response (i.e. SMI, d-ROM, Oxy, RTL), and X represents each of the contaminant score indices (i.e. iCON, iPCB, iPesticides, iPFAS, iHg) and physiological status variables (i.e. SMI, d-ROM, Oxy) (see Table 1 for parameter estimates and p-values).

K is a constant determined by minimising the mean squared error of the regression fit, that is:

$$K = \arg\min \sum (Y_K - Y_K^*)^2$$

where Y_K is the value of the function using the measured data, and Y_K^* is the predicted value under a logistic model fitted with value K as scaling factor, that is:

$$\text{logit}\left(\frac{Y^*}{K}\right) = \beta_0 + \beta_1 X$$

We found a significant relationship between RTL and iPesticides (p-value = 0.022) RTL and iCON (p-value = 0.047), RTL and SMI (p-value = 0.002), and between SMI and iPesticides (p-value = 0.039). There were no significant relationships between oxidative stress and any of the other variables.

4. Discussion

4.1 Contaminant concentrations and profiles in black guillemots during the breeding season

To our knowledge, the present study is the first to report concentrations of circulating contaminants in blood, a less-invasive matrix, in breeding Mandt's Black guillemot. Blood reflect contaminant concentrations from recently consumed prey at a relatively short time-frame (< 10 hours, Drouillard & Norstrom, 2000) and thus our study is restricted to reflect the contamination situation for black guillemots during incubation period. Our results shed light on different aspect of the black guillemots interactions with contaminants: i) black guillemots seem to experience different contaminant exposure depending on the Arctic region and season/foraging areas, and ii) black guillemots during breeding season appear to experience similar exposure to circulating contaminants compared to another seabird species in which several studies have reported increased stress-response, decreased physiological condition and adult survival related to contaminants (e.g. Tartu *et al.* 2013; Goutte *et al.*, 2015; Blevin *et al.*, 2017 a & b).

Black guillemots are diving, subsurface feeders that switch from offshore Arctic cod (*Boreogadus saida*) to nearshore and near-nest-site benthic prey such as sculpin (*Myoxocephalus scorpioides*) or different invertebrates during their breeding season (Cairns, 1987a & b; Mehlum & Gabrielsen, 1993; Byers *et al.*, 2010). While non-breeding guillemots and guillemots from other Arctic regions share similar contaminant patterns, black guillemots present during the breeding season in Kongsfjorden different patterns of POPs in the plasma (∑DDTs >∑PCBs > HCB > ∑CHL > Mirex> ∑HCHs) compared to what has been previously reported for liver and muscle of black guillemots from the Barents Sea and in eggs from the Canadian Arctic (∑PCBs > ∑DDTs >HCB > ∑CHL > ∑HCHs >Mirex) (Borgå *et al.*, 2001; Braune *et al.*, 2018). We also found a higher number of congeners of PFASs than earlier studies investigating liver and eggs of black guillemots in the West Arctic (Martin *et al.*, 2004; Peck *et al.*, 2016).

The black guillemots in our study show similar mean $\partial^{13}C$ signal as earlier studies on non-breeding black guillemots from Greenland which indicate that black guillemots breeding in Kongsfjorden may forage less benthic at least during incubation (Linnebjerg *et al.*, 2016). One

female, however, has a more pelagic $\partial^{13}C$ signal similar to juvenile black guillemots during non-breeding season in the central Barents Sea (Borgå *et al.*, 2005). This specific female also displays higher contaminant concentrations, especially HCB, *trans*-Nonachlor, PBDE-47, Hg and certain PFASs which support a different exposure scenario depending on either breeding state or foraging grounds. Unfortunately, although there was no apparent brood patch, we do not have sufficient information to determine the breeding status (e.g. juvenile or inexperienced/late/failed breeder). Implications of differences in breeding status on physiological status is further discussed in chap. 4.2. $\partial^{15}N$ signals range from 11.9 to 13.5‰, which is similar to the isotopic range of juveniles reported by Borgå *et al.* (2005). At best, our results suggest that black guillemots experience different contaminant situation at different foraging grounds (offshore-nearshore/benthic-pelagic). Whether foraging offshore or pelagic lead to higher exposure remains unclear as the most contaminated bird (a male with especially high level of PFASs) displayed a $\partial^{13}C$ signal similar to the rest of the sample population.

Compared to other seabird species breeding in the same fjord, black guillemots i) have lower Hg concentrations (0.32 ug/g d.w.) than previously reported for black-legged kittiwakes (*Rissa tridactyla*; 1.8 ug/g d.w. red blood cells, Tartu *et al.,* 2013) and ii) show similar wet weight plasma of ∑PCBs, ∑Pesticides, ∑PBDEs and ∑PFAS compared to black-legged kittiwake sampled the same year (Haarr *et al.*, 2017). However, plasma concentrations are often used as proxy for body burden, expressed as lipid weight concentrations to control for the varying lipid content in blood within and between species (Bustnes *et al.,* 2001; Hebert & Keenleyside, 1995). The two species differed in lipid weight contaminant concentrations, thus, we cannot exclude that the two species may still differ in total body burden, i.e. stored contaminants in other organs and tissues accumulated outside breeding season.

Seabirds can go through drastic changes in body mass during breeding season, resulting in lipid-mobilisation and subsequently re-mobilise of contaminants into the bloodstream (Henriksen *et al.*, 1998; Bustnes *et al.,* 2012 & 2017; Fenstad *et al.,* 2016). This seems unrealistic as black guillemots appear rather to increase their body mass during incubation (Aevar Petersen, *pers.com.*) and we find no relationship between body condition and blood lipid content. It is more plausible that black guillemots have a higher lipid content than other seabird species as reported in Harr et al. (2017) due to frequent inputs from more lipid-rich prey during incubation.

Black-legged kittiwakes are subsurface feeders foraging on fish, invertebrates and zooplankton in the pelagic part of the ecosystem, with an offshore foraging distribution during winter. During the breeding season, the black-legged kittiwakes of Kongsfjorden forage within the same fjord system as the black guillemots in this study (nearshore) but in different parts (benthic vs. pelagic) (Vihtakari *et al.,* 2018). Our results also support that auks have a lower biotransformation capacity compared to other gulls (Borgå *et al.,* 2005); as plasma concentrations mirror the same high proportion of cis-Chlordane in ∑CHL as found in internal-tissue samples of non-breeders and black guillemot eggs (Borgå *et al.,* 2001; Braune *et al.,* 2018). Thus, it may seem that even though these two seabirds species have different foraging strategies and biotransformation capacities, they end up with the same exposure to circulating contaminants (of the ones we targeted in this study, except Hg). Furthermore, even though blood lipid content and isotopic signatures fall short to explain contaminants concentrations in the Black guillemot of our study, foraging in less contaminated areas but on prey with higher lipid could lead to the same exposure as the opposite. This aspect may only be understood through integrative methods combined with individual tracking data, as demonstrated by Fort *et al.* (2014).

4.2 The relationships between contaminants and physiological status

Telomere length is a time-integrated measure of life history processes, including age and experienced stress (Young *et al.*, 2013). It has also been proposed as a predictor for *intrinsic* quality (see discussions on different definitions and use of quality in Wilson & Nussey (2010) and Bergeron *et al.* (2011)). In this case, lower quality individuals are expected to present a lower physiological status (i.e. poorer condition) experience more stress and thus greater telomere attrition. Our results show that birds in poor condition had shorter telomeres than birds in better condition, which is consistent with findings in other seabird species showing that relative telomere length and body condition are positively related (Le Vaillant *et al.*, 2015). In addition, birds with higher contaminant scores also tended to be in poorer condition and have shorter telomeres (only significant for total contaminant load and pesticides). This is consistent

366 with what has been observed for oxychlordane and telomere length in the chick-rearing black-
367 legged kittiwakes by Blévin *et al*. (2016), despite that concentrations were lower in our study.

368 The two most contaminated individuals (one female and one male) separating the dataset
369 into two groups dictated the significance in the correlations observed in our data. Although
370 observed at the nest site together with other breeding individuals, we were not able to determine
371 breeding status for the female (i.e. non-breeder or failed or late breeder). Moreover, we lack
372 information about the age of these birds, thus it is difficult to conclude whether or not this female
373 was a juvenile visitor that did not try to breed at all or an unexperienced breeder attempting and
374 failing to reproduce, returning to off-shore foraging ground to eat and coming back to the colony.
375 This bird could also have been a late breeder that recently arrived from off-shore foraging
376 grounds: in the closely related Pigeon guillemot (*Cepphus columba*), experienced breeders arrive
377 earlier than unexperienced, and during egg-formation, females have shown reduced nest
378 attendance compared to males (Nelson, 1987). This female may also have failed because of a
379 poor condition as indicated, by her low body condition, high stress levels and short telomeres.
380 Only in the first situation (juvenile), would it be statistically sound to exclude the individual from
381 the analysis as an outlier; however immature individuals tend to have smaller body size measures
382 (i.e. wing length, bill depth) and this was not the case for this specific individual compared to the
383 rest of the sample population. Thus, with noted caution, we consider it more plausible that the
384 female reflect natural variation of contaminant concentrations and physiological condition
385 related to individual foraging tactics within breeding black guillemots.

386 Another aspect is the sample size: an increase of sample size could well have distinguished
387 these two individuals as clear outliers; on the other hand, one should also take caution being-
388 blinded by the mean-response of the population (Bennett, 1987) as toxic effects can also
389 influence the variance of the physiological response (Forbes & Depledge, 1999). The distribution
390 in our contaminant data may also be the product of cumulative processes: some birds end up as
391 extremes due to individual strategies over time (e.g. individual foraging specialisation as
392 observed for diving seabirds by Bearhop *et al.* (2006)). Such extreme values induce non-linearity
393 in the data (i.e. S-shaped function) with 1) lower contamination score with little or no effect
394 observed on response variables measured, and 2) high contamination score and higher effect on
395 response variables measured. This is also what we see as the main tendency in our data. Yet iHg

396 did not correlate with the other score indices but showed a more linear functional relationship
397 with the physiological status responses.

398 The same tendency of relationship with contaminants was also observed in respect to
399 stress: although insignificant, the reactive oxygen metabolites (d-ROM) levels seemed to
400 increase with increasing contaminant score, while for the variance, observations indicate a
401 decrease. The individuals with higher d-ROM had also lower global plasmatic antioxidant
402 defence (Oxy). This is consistent with previous findings in birds (Hegseth *et al.,* 2011; Bourgeon
403 *et al.,* 2012; Sletten *et al.,* 2016; Abbasi *et al.,* 2017) that support the hypothesis that contaminant
404 exposure may contribute to increased oxidative stress (Isaksson *et al.,* 2010; Constantini *et al.,*
405 2014). The lack of relationship between d-ROM and telomeres may be due to a mismatch of
406 tissue (Constantini *et al.,* 2014) or time scales. Telomere length is commonly understood as an
407 integrative measure of accumulated oxidative stress, while d-ROM is an indicator of
408 instantaneous oxidative stress (Urvik *et al.,* 2016).

409 **4.3 Theoretical response pathway for contaminant exposure and telomere length**

410 The low sample size restricted us from evaluating our proposed response pathway in regards to
411 contaminant loading and telomere length. However, the tendency in the data allowed us to build
412 a hypothetical scenario for future investigation in which oxidative stress and telomere length will
413 be dictated by accumulated contaminant burden and body condition above and below certain
414 thresholds.

415 The threshold idea is based on inter-individual physiological variability (Depledge, 1990)
416 resulting in different abilities to perform across different environmental conditions (e.g., low and
417 high food availability, low and high contamination), without increasing the cost (i.e. stress,
418 condition) – or *intrinsic quality* (Bergeron *et al.,* 2011). Furthermore, at some point, contaminant
419 exposure may start influencing these abilities. A central question is then if birds with higher
420 contaminant concentrations are a result of lower body condition, as observed in Wayland *et al.*
421 *(*2002) – or vice versa – if higher concentrations of contaminants have led to lower body
422 condition. In the former, body condition may determine physiological status while the correlating
423 contaminants play a co-founding role; the effect of weight loss is more important for the overall
424 stress situation compared to the contribution from contaminants (Fenstad *et al.,* 2014).

However, black guillemots do not experience significant weight loss during incubation due to shared parental effort and strategies of short-distance trips (both partners forage every day) (Aevar Petersen & George Divoky, *pers.com.*). This is supported by our findings, where no relationship between lipid content, body condition and contaminants was observed. An alternative explanation is that the most contaminated birds may be in poorer condition due to the presence of high contaminant concentrations over time (i.e. strategy driven cumulative processes). In this case, we may expect a shift between physiological states where an individual with contaminant concentrations above a certain threshold value is bound to have a lower physiological status over time.

5. Conclusions

In conclusion, our results demonstrate that, during the breeding season, black guillemots experience the same circulating contaminant concentrations as a pelagic migratory seabirds, the Black-legged kittiwake, for which several clear associations between contaminants, stress response and breeding success have been reported. Our study also emphasize the importance of individual tracking datasets and information on breeding and foraging status in wildlife ecotoxicology. Despite non-significant relationships, our data shows a consistent tendency that black guillemots in poorer condition had shorter telomeres, higher concentrations of contaminants and higher levels of oxidative stress. Our threshold hypothesis should be tested with a larger dataset, coupled with life history data, in the future: if a threshold for effects exists, it will likely both depend on breeding behaviour (e.g. fasting during incubation shift) and environmental stressors (e.g. food availability, habitat quality acting on body condition), and a species' behavioural and physiological traits (e.g. life history, biotransformation). Thus, we should expect the same non-linear relationship between telomere length and contaminants and body condition, with species-specific thresholds, in other seabirds as well.

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Data availability

Data is publicly available on FigShare (DOI: 10.6084/m9.figshare.6225071). [During the review process these data can be accessed using this temporary URL: https://figshare.com/s/8d10a917a3274aba1d6b].

462 **References**

463 Muir, D.C.G. *et al.*, (1992) 'Arctic marine ecosystem contamination', *The Science of The Total*
464 *Envionment*, 122, pp. 75-134.

465 de March, B.G.E. *et al.*, (1998) Persistent organic pollutants, chap 6. In: AMAP Assessment
466 Report: Arctic Pollution Issues. Oslo, Norway: Arctic Monitoring and Assessment Programme,
467 pp. 183 – 372.

468 Dietz, R. *et. al., (2000)* 'Comparison of contaminants from different trophic levels and
469 ecosystems', *Science of The Total Environment*, 245 (1-3), pp.221-231.
470 https://doi.org/10.1016/S0048-9697(99)00447-7

471 Erikstad, K. *et al.*, (2013) 'Persistent organic pollution in a high-Arctic top predator: sex-
472 dependent thresholds in adult survival', *Proceedings of the Royal Society*, 280, p. 20131483. doi:
473 10.1098/rspb.2013.1483.

474 Tartu, S. *et al.*, (2014a) 'The stress of being contaminated? Adrenocortical function and
475 reproduction in relation to persistent organic pollutants in female black legged kittiwakes',
476 *Science of the Total Environment*. Elsevier B.V., 476–477, pp. 553–560. doi:
477 10.1016/j.scitotenv.2014.01.060.

478 Bustnes, J. O. *et al.*, (2015) 'Multiple stressors in a top predator seabird : Potential ecological
479 consequences of environmental contaminants, population health and breeding conditions', *PLoS*
480 *ONE*, 10(7), pp. 1–18. doi: 10.1371/journal.pone.0131769.

481 Bourgeon, S. *et al.*, (2012) 'Individual variation in biomarkers of health: influence of persistent
482 pollutants in Great skuas (Stercorarius skua) breeding at different geographical locations.
483 *Environmental Research*, 118, pp. 31-39.

484 Goutte, A. *et al.*, (2014) 'Demographic consequences of heavy metals and persistent organic
485 pollutants in a vulnerable long-lived bird, the wandering albatross', *Proceedings of the Royal*
486 *Society of London. Series B: Biological Sciences*, 281, pp. 1–8. doi: 10.1098/rspb.2013.3313.

Baert, J. M. *et al.*, (2013) 'Migration and opportunistic feeding increase PCB accumulation in Arctic seabirds', *Environmental Science and Technology*, 47(20), pp. 11793–11801. doi: 10.1021/es402898t.

Fort, J. *et al.*, (2014) 'Spatial Ecotoxicology: Migratory Arctic Seabirds Are Exposed to Mercury Contamination While Overwintering in the Northwest Atlantic'. *Environmental Science Technology*, 48 (19), pp. 11560-7

Brown, R. G. B. (1985) The Atlantic Alcidac at Sea. Chap. 9. pp. 383-426, in: Nettleship, D. N. Birkhead, T. R. (eds.), The Atlantic Alcidae. Academic Press, London.

Divoky G.J. *et al.*, (2016) 'Arctic sea ice a major determinant in Mandt's black guillemot movement and distribution during non-breeding season', Biology Letters 12: 20160275. http://dx.doi.org/10.1098/rsbl.2016.0275

Vorkamp, K. *et al.*, (2004) 'Persistent halogenated compounds in black guillemots (Cepphus grylle) from Greenland - Levels, compound patterns and spatial trends', *Marine Pollution Bulletin*, 48(1–2), pp. 111–121. doi: 10.1016/S0025-326X(03)00369-2.

Vorkamp, K. *et al.*, (2015) 'Novel brominated flame retardants and dechlorane plus in Greenland air and biota', *Environmental Pollution*, 196, pp. 284–291. doi: 10.1016/j.envpol.2014.10.007.

Braune, B. M. *et al.*, (2015) 'Trends of polybrominated diphenyl ethers and hexabromocyclododecane in eggs of Canadian Arctic seabirds reflect changing use patterns', *Environmental Research*, 142, pp. 651–661.

Braune, B. M. *et al.*, (2016) 'Temporal trends of mercury in eggs of five sympatrically breeding seabird species in the Canadian Arctic', *Environmental Pollution*, 214, pp. 124–131. doi: 10.1016/j.envpol.2016.04.006.

Peck, L. E. *et al.*, (2016) 'Persistent organic pollutant and mercury concentrations in eggs of ground-nesting marine birds in the Canadian high Arctic', *Science of the Total Environment*. Elsevier B.V., 556, pp. 80–88. doi: 10.1016/j.scitotenv.2016.02.205.

512 Divoky, G. J. *et al.*, (2015) 'Effects of recent decreases in arctic sea ice on an ice-associated
513 marine bird', Progress in Oceanography. Elsevier Ltd, 136, pp. 151–161. doi:
514 10.1016/j.pocean.2015.05.010.

515 Hobson, K. A. (1993) 'Trophic relationships among high Arctic seabirds: insights from tissue-
516 dependent stable-isotope models', *Marine Ecology Progress Series*, 97, pp. 7-18.

517 Cairns, D. K. (1987a) 'Diet and foraging ecology of black guillemots in northeastern Hudson
518 Bay', *Can. J. Zool*, 65, pp. 1257–1263.

519 Cairns, D.K. (1987b) 'The ecology and energetics of chick provisioning by Black guillemots',
520 *The Condor*, 89, pp. 627-635.

521 Barrett, R. (2002) 'Food consumption by seabirds in Norwegian waters', *ICES Journal of*
522 *Marine Science*, 59(1), pp. 43–57. doi: 10.1006/jmsc.2001.1145.

523 Koistinen, J. *et al.*, (1995) 'PCDEs, PCBs, PCDDs and PCDFs in Black guillemots and White-
524 tailed sea eagles from the Baltic Sea', *Chemosphere*, 30 (9), pp. 1671-1684.

525 Appelquist H. *et al.*, (1987) 'Variation in mercury content of guillemot feathers over 150 years',
526 *Marine Pollution Bulletin*, 16 (6), pp 244-248.

527 Borgå, K. *et al.*, (2001) 'Biomagnification of organochlorines along a Barents Sea food chain',
528 *Environmental Pollution*, 113, pp. 187-198.

529 Borgå, K. *et al.*, (2005) 'Bioaccumulation of PCBs in Arctic seabirds: influence of dietary
530 exposure and congener biotransformation', *Environmental Pollution*, 134, pp. 397–409

531 Borgå, K. *et al.*, (2007) 'Selective bioaccumulation of chlorinated pesticides and metabolites in
532 Arctic seabirds', *Environmental Pollution*, 145, pp. 545-553.

533 Olafsdottir, K. *et al.*, (2005) 'Temporal trends of organochlorine contamination in Black
534 Guillemots in Iceland from 1976 to 1996', *Environmental Pollution*, 133, pp 509-515.
535 doi:10.1016/j.envpol.2004.06.016

536 Constantini, D. *et al.*, (2014) 'Oxidative stress in relation to reproduction, contaminants, gender
537 and age in a long-lived seabird' *Oecologia*, 175, pp. 1107–1116.

538 Wielsøe, M. *et al.*, (2014) 'Perfluoroalkylated substances (PFAS) affect oxidative stress
539 biomarkers in vitro', *Chemosphere*, 129, pp. 239-245.
540 http://dx.doi.org/10.1016/j.chemosphere.2014.10.014

541 Durant, S.E. *et al.*, (2007) 'Energy acquisition and allocation in an ectothermic predator exposed
542 to a common environmental stressor', *Comparative Biochemistry and Physiology Part C:
543 Toxicology & Pharmacology*, 145 (3), pp. 442-448. https://doi.org/10.1016/j.cbpc.2007.01.014

544 Von Zglinicki, T. (2002) 'Oxidative stress shortens telomeres', *Trends in Biochemical Sciences*,
545 27(7), pp. 339–344. doi: 10.1016/S0968-0004(02)02110-2.

546 Epel, E.S. *et al.*, (2004) 'Accelerated telomere shortening in response to life stress', *PNAS*, 101
547 (49), pp. 17312-17315. www.pnas.orgcgidoi10.1073pnas.0407162101

548 Blackburn, E. H. (2000) 'Telomere states and cell fates', *Nature*, 408, pp. 53–56. doi:
549 10.1038/35040500.

550 Monaghan, P & Haussmann, M. F. (2006) 'Do telomere dynamics link lifestyle and lifespan?',
551 *Trends in Ecology and Evolution*, 21(1), pp. 47–53. doi: 10.1016/j.tree.2005.11.007.

552 Reichert, S & Stier, A. (2017) 'Does oxidative stress shorten telomeres in vivo? A review.'
553 *Biology Letters 13:* 20170463.http://dx.doi.org/10.1098/rsbl.2017.0463

554 Beaulieu, M. *et al.*, (2011) 'Oxidative status and telomere length in a long-lived bird facing a
555 costly reproductive event', *Functional Ecology*, 25(3), pp. 577–585. doi: 10.1111/j.1365-
556 2435.2010.01825.x.

557 Schull, Q. *et al.*, (2016) 'The oxidative debt of fasting: evidence for short- to medium-term costs
558 of advanced fasting in adult king penguins', *The Journal of Experimental Biology*, 219(20), pp.
559 3284–3293. doi: 10.1242/jeb.145250.

Horn, T. *et al.*, (2010) 'The use of telomere length in ecology and evolutionary biology', *Heredity*, *105*(6), 497–506. https://doi.org/10.1038/hdy.2010.113

Barrett, E. L. B. *et al.*, (2013) 'Telomere length and dynamics predict mortality in a wild longitudinal study', *Molecular Ecology*, *22*(1), 249–259. https://doi.org/10.1111/mec.12110

Blackburn, E. H. (2005) 'Telomeres and telomerase: Their mechanisms of action and the effects of altering their functions', *FEBS Letters*, 579(4 SPEC. ISS.), pp. 859–862. doi: 10.1016/j.febslet.2004.11.036.

Haussmann, M. F. *et al.*, (2007) 'Telomerase activity is maintained throughout the lifespan of long-lived birds', *Experimental Gerontology*, 42(7), pp. 610–618. doi: 10.1016/j.exger.2007.03.004.

Bauch, C. *et al.*, (2013) 'Telomere length reflects phenotypic quality and costs of reproduction in a long-lived seabird', *Proceedings of the Royal Society B*, 280 (1752), 20122540. doi: 10.1098/rspb.2012.2540

Young, R.C. *et al.*, (2017) 'Effects of developmental conditions on growth, stress and telomeres in black-legged kittiwake chicks', *Molecular Ecology*, 26, pp. 3572-3584

Blévin, P. *et al.*, (2016) 'Exposure to oxychlordane is associated with shorter telomeres in arctic breeding kittiwakes', *Science of the Total Environment*. Elsevier B.V., 563–564, pp. 125–130. doi: 10.1016/j.scitotenv.2016.04.096.

Blévin, P. *et al.*, (2017a) 'Perfluorinated substances and telomeres in an Arctic seabird: Cross-sectional and longitudinal approaches', *Environmental Pollution*, 230, pp. 360-367.

Haarr, A. *et al.*, (2017) 'DNA damage in arctic seabirds: baseline, sensitivity to a genotoxic stressor and association to organohalogen contaminants', Environmental Toxicology and Chemistry. doi: 10.1002/etc.4035

Blévin, P. *et al.*, (2017b) 'Contaminants and energy expenditure in an Arctic seabird: Organochlorine pesticides and perfluoroalkyl substances are associated with metabolic rate in a contrasted manner', *Environmental Research*, 157, pp. 118-126.

586 Hobson, K. A. *et al.,* (1995) 'Sources of primary production, benthic-pelagic coupling, and
587 trophic relationships within the Northeast Water Polynya: insights from ∂13C and ∂15N
588 analysis', *Marine Ecology Progress Series*, 128(1–3), pp. 1–10. doi: 10.3354/meps128001.

589 Fisk, A. T. *et al.,* (2001) 'Chlordane components and metabolites in seven species of Arctic
590 seabirds from the Northwater Polynya: Relationships with stable isotopes of nitrogen and
591 enantiomeric fractions of chiral components', *Environmental Pollution*, 113(2), pp. 225–238.
592 doi: 10.1016/S0269-7491(00)00153-6.

593 Stier, A. *et al.,* (2014) 'Starting with a handicap: phenotypic differences between early- and late-
594 born king penguin chicks and their survival correlates', *Functional Ecology,* 28, pp. 601-611.
595 doi: 10.1111/1365-2435.12204

596 Le Vaillant, M. *et al.,* (2015) 'Telomere length reflects individual quality in free-living adult
597 king penguins' Le Vaillant, *Polar Biology,* 38(12), pp. 2059–2067.

598 Reichert, S. *et al.,* (2015) 'Maternal telomere length inheritance in the King penguin', *Heredity,*
599 114, pp. 10-16. doi: 0.1038/hdy.2014.60

600 Sambrook, J. *et al.,* (1989) 'Molecular Cloning: a Laboratory Manual', 2nd edn. Cold Spring
601 Harbor Laboratory Press, Cold Spring Harbor, New York.

602 Labocha, M.K. & Hayes, J.P. (2012) 'Morphometric indices of body condition in birds: a
603 review', *Journal of Ornithology*, 153, pp. 1–22. DOI 10.1007/s10336-011-0706-1

604 Peig, J. & Green, A.J. (2009) 'New perspectives for estimating body condition from mass/length
605 data: the scaled mass index as an alternative method', *Oikos,* 188, pp. 1883-1891. doi:
606 10.1111/j.1600-0706.2009.17643.x

607 R Core Team (2016) 'R: A language and environment for statistical computing', R Foundation
608 for Statistical Computing, Vienna, Austria. URL http://www.R-project.org/.

609 Armbruster & Pry (2008) 'Limit of Blank, Limit of Detection and Limit of Quantitation',
610 *Clinical Biochemistry Rev* (29) Supplementary.

Baccarelli, A. *et al.,* (2005*)* 'Handling of dioxin measurement data in the presence of non-detectable values: Overview of available methods and their application in the Seveso chloracne study', *Chemosphere,* 60, pp. 898–906

Greenacre, M. (2017) ''Size' and 'shape' in the measurement of multivariate proximity', *Methods in Ecology and Evolution*, 8, pp. 1415–1424.

Drouillard, K.G. & Norstrom, R.J. (2000) 'Dietary absorption efficiencies and toxicokinetics of polychlorinated biphenyls in ring doves following exposure to aroclor® mixtures', Environmental Toxicology and Chemistry, 19 (11), pp. 2707-2714. https://doi.org/10.1002/etc.5620191114

Tartu, S. *et al.,* (2014b). Endocrine and fitness correlates of long-chain perfluorinated carboxylates exposure in arctic breeding black-legged kittiwakes. *Environmental Science and Technology*, *48*(22), 13504–13510. https://doi.org/10.1021/es503297n

Goutte, A. *et al.,* (2015) 'Survival rate and breeding outputs in a high Arctic seabird exposed to legacy persistent organic pollutants and mercury', *Environmental Pollution* (1970), Elsevier Science, pp. 1-9.

Mehlum, F. & Gabrielsen, G. W. (1993) 'The diet of high-arctic seabirds in coastal and ice-covered, pelagic areas near the Svalbard archipelago', *Polar research,* 12(1), pp. 1–20.

Byers, T. *et al.,* (2010) 'Diet of black guillemots and northern fulmars breeding beside a High Arctic polynya', *Polar Biology,* 33 (4), pp. 457-467.

Braune, B.M. *et al.,* (2018) 'Temporal trends of legacy organoschlorines in eggs of Canadian Artcic seabirds monitored over four decades', *Science of the Total Environment,* 646, pp.551-563, doi: 10.1016/j.scitotenv.2018.07.291.

Martin, J. W. *et al.,* (2004) 'Identification of Long-Chain Perfluorinated Acids in Biota from the Canadian Arctic', *Environmental Science and Technology*, 38(2), pp. 373–380. doi: 10.1021/es034727+.

636 Linnebjerg, J.F. (2016) 'Dechipering the structure of the West Greenland marine food web using
637 stable isotopes (∂13C, ∂15N)', Marine Biology, 163:230, DOI 10.1007/s00227-016-3001-0

638 Tartu, S. *et al.*, (2013) 'To breed or not to breed: endocrine response to mercury contamination
639 by an Arctic seabird', *Biology Letters*, 9, 20130317, http://dx.doi.org/10.1098/rsbl.2013.0317

640 Bustnes, J. O. *et al.*, (2001) 'Whole blood concentrations of organochlorines as a dose metric for
641 studies of the glaucous gull (Larus hyperboreus)', *Environmental Toxicology and Chemistry*,
642 20(5), pp. 1046–1052. doi: 10.1002/etc.5620200515.

643 Hebert, C.E. & Keenleyside, K.A. (1995) 'To normalize or not to normalize? Fat is the question',
644 *Environmental Toxicology and Chemistry*, 14 (5), pp. 801-807.

645 Henriksen, E. O. *et al.*, (1998) 'Validation of the use of blood samples to assess tissue
646 concentrations of organochlorines in glaucous gulls, Larus hyperboreus', *Chemosphere*, 37(13),
647 pp. 2627–2643. doi: 10.1016/S0045-6535(98)00162-3.

648 Bustnes, J. O. *et al.*, (2012) 'Temporal Dynamics of Circulating Persistent Organic Pollutants in
649 a Fasting Seabird under Different Environmental Conditions', *Environmental Science and*
650 *Technology*, (August), pp. 10287–10294.

651 Bustnes, J. O. *et al.*, (2017) 'Temporal variation in circulating concentrations of organochlorine
652 pollutants in a pelagic seabird breeding in the High Arctic', *Environmental Toxicology &*
653 *Chemistry* 36:, pp. 442-448.

654 Fenstad, A. A. *et al.*, (2016) 'Antioxidant Responses in Relation to Persistent Organic Pollutants
655 and Metals in a Low- and a High-Exposure Population of Seabirds', *Environmental Science and*
656 *Technology*, 50(9), pp. 4817–4825. doi: 10.1021/acs.est.6b00478.

657 Vihtakari, M. *et al., (2018),* 'Black-legged kittiwakes as messengers of Atlantification in the
658 Arctic', *Scientific Reports*, 8 (1178). doi:10.1038/s41598-017-19118-8

659 Young, R.C. *et al.*, (2013) 'Age, Sex, and Telomere Dynamics in a Long-Lived Seabird with
660 Male-Biased Parental Care', *PLoS ONE*, 8(9): e74931. doi:10.1371/journal.pone.0074931

661 Wilson, A.J. & Nussey, D.H. (2010) 'What is individual quality? An evolutionary perspective',
662 *Trends in Ecology and Evolution*, 25 (4), pp. 207-2014.

663 Bergeron, P. *et al.,* (2011) 'Individual quality: tautology or biological reality?', *Journal of*
664 *Animal Ecology*, 80, pp. 361–364.

665 Nelson, A.D. (1987) 'Factors influencing colony attendance by Pigeon guillemots on Southeast
666 Farallon Island, California', *The Condor,* 89, pp. 340-348.

667 Bennett, A. F. (1987) 'Interindividual variability: an underutilized resource' In New directions in
668 ecological physiology, pp. 147-169. Cambridge, UK: Cambridge University Press.

669 Forbes, V.E. and Depledge M.H. (1996) 'Environmental stress and the dristribution of traits
670 within populations', Chapter 7, ECOtoxicology: Ecological Dimensions, Chapman & Hall,
671 London.

672 Bearhop, S.et al., (2006) 'Stable isotopes indicate sex-specific and long-term individual foraging
673 specialization in diving seabirds', *Marine Biology Progress Series*, 311, pp. 157-164.

674 Hegseth, M.N. *et al.,* (2011) 'Hepatic antioxidant responses related to levels of PCBs and metals
675 in chicks of three Arctic seabird species', *Comparative Biochemistry and Physiology, Part C,*
676 154, pp. 28-35

677 Sletten, S. *et al.,* (2016) 'Organohalogenated contaminants in white-tailed eagle (*Haliaeetus*
678 *albicilla*) nestlings: An assessment of relationships to immunoglobulin levels, telomeres and
679 oxidative stress', *Science of the Total Environment*, 539, pp. 337–349.

680 Abbasi, N.A. *et al.,* (2017) 'Oxidative stress responses in relationship to persistent organic
681 pollutant levels in feathers and blood of two predatory bird species from Pakistan', *Science of the*
682 *Total Environment,* 580, pp. 26–33.

683 Isaksson, C. (2010) 'Pollution and its impact on wild animals: A meta-analysis on oxidative
684 stress', *EcoHealth,* 7(3), pp. 342–350. doi: 10.1007/s10393-010-0345-7.

685 Urvik, J. *et al.*, (2016) 'Variation in the Markers of Nutritional and Oxidative State in a Long-
686 Lived Seabird: Associations with Age and Longevity', *Physiology and Biochemistry Zoology*, 89
687 (5), pp. 417-40.

688 Depledge, M.H. (1990) 'New Approaches in Ecotoxicology: Can Interindividual Physiological
689 Variability Be Used as a Tool to Investigate Pollution Effects?', *Ambio*, 19 (5), pp. 251-252.

690 Wayland, M. *et al.*, (2002) 'Immune function, stress response, and body condition in arctic-
691 breeding common eiders in relation to cadmium, mercury, and selenium concentrations'
692 *Environmental Research*, 90 (1), pp. 47-60.

693 Fenstad, A. A. *et al.*, (2014) 'DNA double-strand breaks in relation to persistent organic
694 pollutants in a fasting seabird', *Ecotoxicology and Environmental Safety*, 106, pp. 68–75.

695 **Figure legends**

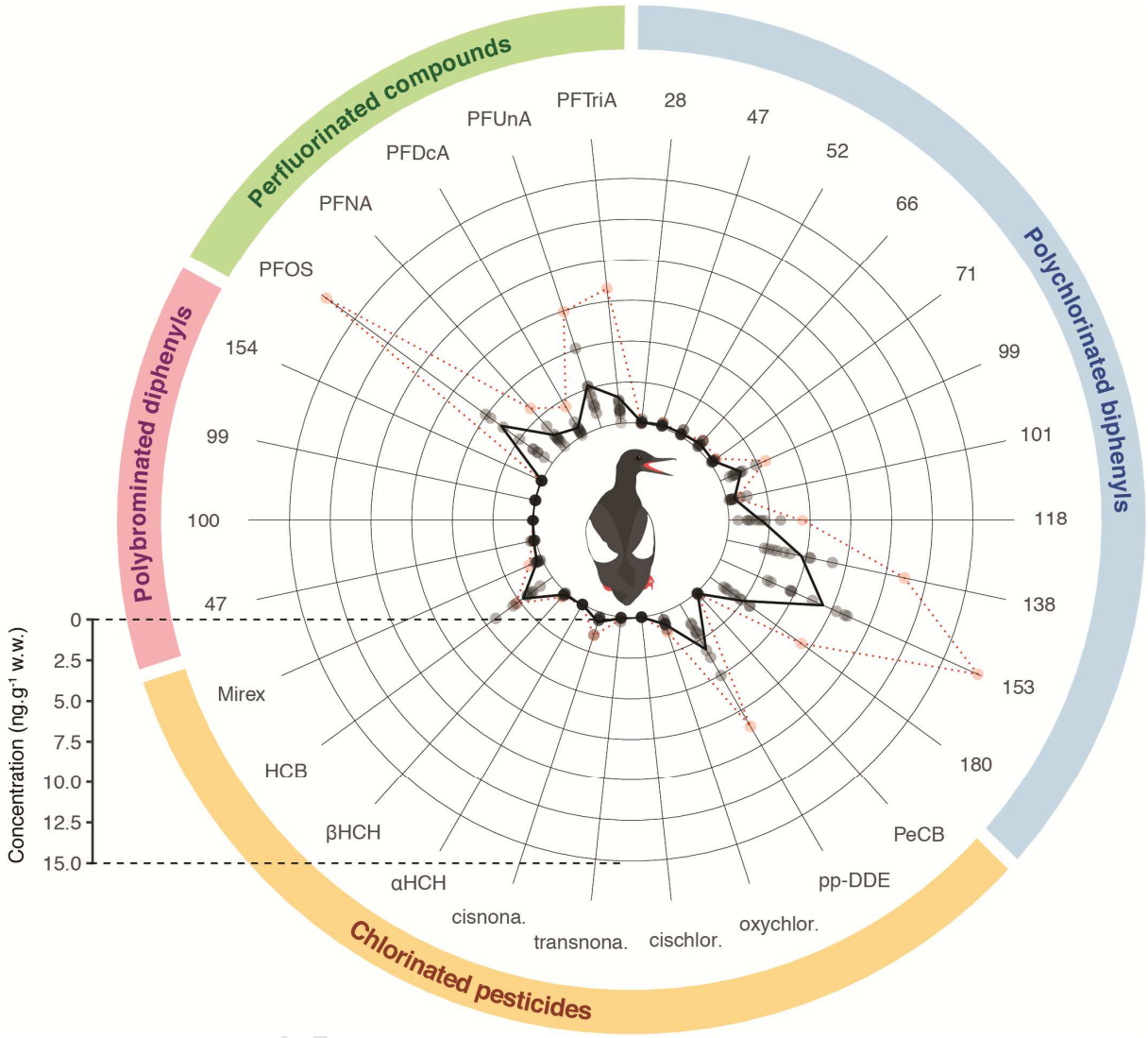
696 **Figure 1. Contaminant profiles in the Black guillemot.** Radar plot showing concentrations
697 (ng/g w.w.) and profile signatures of PCB, pesticides, PBDE and PFAS congeners detected in
698 incubating black guillemots (points, with 'extreme' male marked in red colour) and line (black)
699 indicating the mean.

700 **Figure 2. Scatterplots of relations between contaminant score indices, body condition,**
701 **oxidative stress, anti-oxidant capacity, and relative telomere length, in females (red) and**
702 **males (blue). (a)** iCON and SMI with RTL indicated by the size of the points. **(b)** iCON and d-
703 ROM with RTL indicated by the size of the points. **(c)** d-ROM and Oxy with iCON indicated by
704 the size of the points. **(d)** iPCB and SMI with RTL indicated by the size of the points. **(e)**
705 iPesticides and SMI with RLT indicated by the size of the points. **(f)** iHg and body SMI with
706 RTL indicated by the size of the points. **(g)** iPBDE and body SMI with RTL indicated by the size
707 of the points. (**h**) iPFAS and SMI with RTL indicated by the size of the points.

708 **Table 1: Parameter estimates for modelling the relationships among contaminants (iCON)**
709 **and physiological status (SMI, d-ROM, RTL) in incubating black guillemots.** Bold
710 characters refer to significant relationships under alpha < 0.05.

711

712



713

714 **FIGURE 1**

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716

717

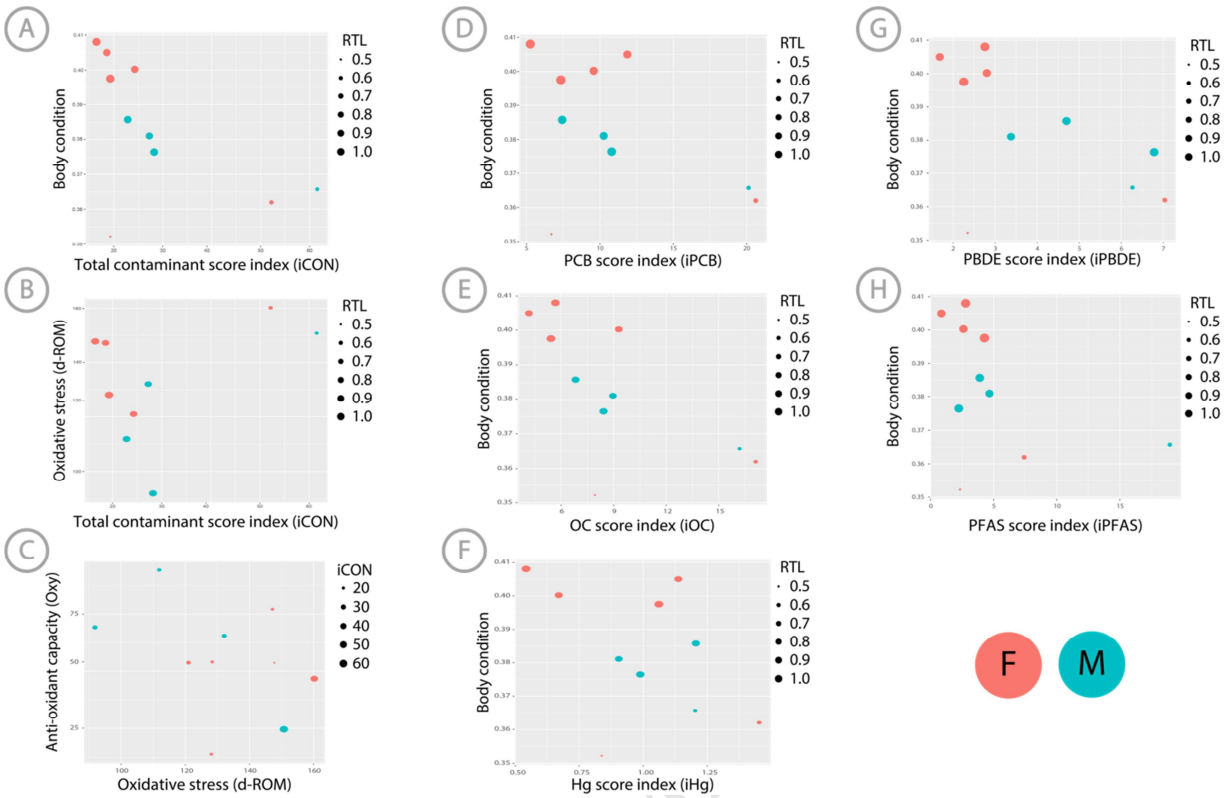


FIGURE 2

721 **Table 1.**

722

Dependent variable	Independent variable	K	Beta0	Beta1	Df	t-value	P-value
RTL	**iCON**	1.5	0.77	-0.02	8	-2.34	**0.047**
	iPCB	1.5	0.77	-0.06	8	-2.12	0.067
	iPesticides	1.5	0.88	-0.08	8	-2.85	**0.022**
	iPBDE	1.5	0.48	-0.09	8	-1.04	0.328
	iPFAS	1.54	0.33	-0.05	8	-1.79	0.112
	iHg	1.5	0.85	-0.73	8	-1.16	0.279
	SMI	1.5	-8.70	23.00	8	4.69	**0.002**
	d-ROM	1.5	1.55	-0.01	8	-1.34	0.218
	Oxy	1.64	-0.31	0.004	8	0.64	0.538
SMI	iCON	2.04	-1.40	-0.002	8	-1.95	0.086
	iPCB	0.61	0.65	-0.012	8	-1.51	0.168
	iPesticides	2.04	-1.38	-0.009	8	-2.47	**0.039**
	iPBDE	0.63	0.57	-0.04	8	-1.90	0.094
	iPFAS	2.04	-1.44	-0.005	8	-1.35	0.216
	iHg	2.04	-1.37	-0.098	8	-1.33	0.219
d-ROM	iCON	800.95	-1.80	0.005	8	1.17	0.277
	SMI	257.19	0.27	-0.557	8	-0.09	0.928
Oxy	iCON	141.27	0.01	-0.018	8	-0.98	0.355
	SMI	141.27	-1.91	3.616	8	0.236	0.819

723

724 **SUPPLEMENTARY MATERIAL**

725 **Supplementary Table 1.** Concentrations of contaminants (pg/g w.w., Hg given as ng/g d.w.),
726 lipid content, dietary signatures (‰), oxidative stress, and relative telomere length in blood from
727 black guillemots. Values are given as detection frequency, mean, standard deviation, and range
728 (min-max).

729 * Due to poor extraction recoveries with two of the three method blanks of the internal standard for
730 oxychlordane, we calculated a conservative detection limit by multiplying the blank response in the blank sample
731 which acceptable extraction recoveries were obtained by a factor of three to avoid the potential of reporting false
732 positives. In addition, a minor isobaric interference coeluted with oxychlordane. As blank substraction was not
733 performed, concentrations approaching detection limits will be heavily influenced by co-eluting interference and
734 should be treated with caution. We decided on a conservative, cautionary approach and excluded oxychlordane from
735 further analyses.

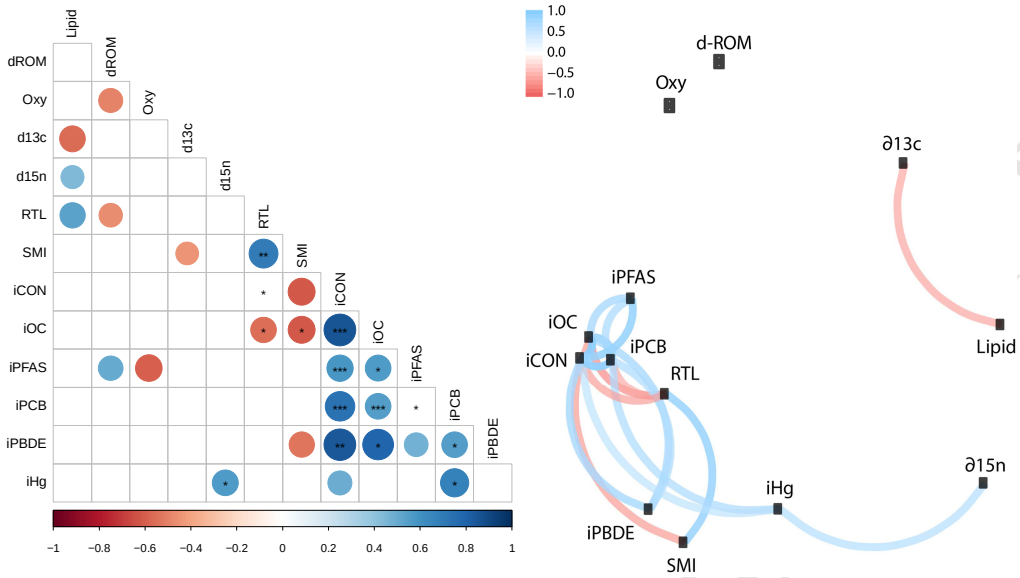
		Detection frequency (%)	Mean (median)	Standard deviation	Min	Max
$\sum Pesticides_{17}$	PeCB	100	109 (92.8)	60.5	45.1	274.5
	aHCH	60	14.8 (15.5)	3.7	9.1	19.2
	bHCH	100	186.2 (180.6)	94.8	59.0	382
	γHCH	10	<LOD			
	HCB	100	2227 (1900)	925	929	4300
	Oxychlordane*	50	900 (778)	248	699	1246
	Trans-chlordane	0	<LOD			
	Cis-chlordane	100	16.1 (16.5)	2.4	12.1	19.5
	Trans-nonachlor	100	85.0 (61.3)	81.6	24.7	297
	Cis-nonachlor	100	605 (452)	456	278	1491
	Mirex	100	352 (327)	214	131	877
	o,p DDE	0	<LOD			
	p,pDDE	100	3188 (2259)	2235	1338	8653
	o,p DDD	0	<LOD			
	p,p DDD	0	<LOD			
	o,p DDT	0	<LOD			
	p,p DDT	0	<LOD			
$\sum PCB_{11}$	PCB28	60	118 (90.6)	69.6	74.2	258
	PCB47	100	173 (147)	83.8	101	328
	PCB52	100	184 (145)	128	86.0	442
	PCB66	100	337 (302)	191	111	612
	PCB71	100	231 (196)	115	82.0	478
	PCB99	100	1390 (1179)	757	625	3033
	PCB101	100	479 (360)	349	147	1277
	PCB118	100	1977 (1779)	1145	577	4529

	PCB138	100	4689 (3793)	2646	2378	11156
	PCB153	100	6897	4143	3169	17303
	PCB180	100	2474 (2076)	1710	1034	6922
$\sum PBDE_{10}$	PBDE28	20	<LOD			
	PBDE71	0	<LOD			
	PBDE47	100	145 (127)	80.4	72.0	322
	PBDE66	0	<LOD			
	PBDE100	100	48.1 (37.0)	24.2	24.0	88.0
	PBDE99	90	35.9 (32.2)	20.2	12.0	79.0
	PBDE154	90	43.1 (38.0)	19.4	24.0	70.0
	PBDE153	20	<LOD			
	PBDE183	10	<LOD			
	PBDE209	0	<LOD			
$\sum PFAS_{20}$	6:2FTS	0	<LOD			
	8:2FTS	0	<LOD			
	PFOSA	0	<LOD			
	PFBS	0	<LOD			
	PFPS	0	<LOD			
	PFHxS	40	<LOD			
	PFHpS	0	<LOD			
	brPFOS	0	<LOD			
	linPFOS	100	3892 (1984)	4923	630	17162
	PFNS	0	<LOD			
	PFDcS	0	<LOD			

	PFHxA	0	<LOD			
	PFHpA	0	<LOD			
	PFOA	20	<LOD			
	PFNA	100	1067 (807)	876	139	3247
	PFDcA	100	568 (393)	581	44.4	2048
	PFUnA	100	2718 (2086)	2033	926	7450
	PFDoA	20	<LOD			
	PFTriA	90	1781 (872)	2457	577	8286
	PFTeA	20	<LOD			
∑Hg	totalHg	100	323 (330)	83.2	175	469
%Lipid	EOM	100	1.2 (1.34)	0.5	0.5	1.9
Trophic position	∂15N (‰)	100	12.7 (12.9)	0.6	11.9	13.5
Carbon source	∂13C (‰)	100	-19.9 (-19.7)	0.7	-21.5	-19.2
Baseline-corrected trophic level	TL	100	4.5	0.2	5.0	4.3
Derivatives of reactive oxygen metabolites	d-ROM	100	132 (130.2)	20.5	91.8	160
Plasmatic antioxidant barrier	Oxy	100	55.3 (53.9)	23.7	13.9	94.2
Relative telomere length	RTL	100	0.80 (0.87)	0.2	0.5	1.0

736

737



Supplementary Figure 1. Left: Correlations among the contaminant score indices (iCON, iPCB, iPesticides, iHg, iPBDE, iPFAS), dietary descriptors (∂15N, ∂13C, lipid content) and physiological status measures (SMI, d-ROM, Oxy, RTL) in the Black guillemot. Significant relations (Spearman's test) indicated by *** ($p<0.001$), ** ($p<0.05$) and * ($p<0.1$). Right: Correlation network graph between the contaminant score indices, dietary descriptors and physiological status.

Chemical analysis: Details of instrument

Analysis of PCBs

Analysis of PCBs was performed by gas chromatography mass spectrometry (GC/MS) using an Agilent 7890A gas chromatograph (Agilent Technologies, Böblingen, Germany) coupled to a Quattro Micro triple quadrupole mass spectrometer (Waters Coporation, Manchester, United Kingdom). Separation was carried out using a DB-5 MS column (30 m x 250 µm inner diameter (id), 0.25µm film thickness, J&W, Folsom, USA) equipped with a 5 m guard column (Restek

Corporation, Bellefonte, PA, USA) at 1ml/min with Helium as a carrier gas. Samples were injected at 250°C using a split/splitless injector equipped with a 4 mm id packed (wool) glass liner operated in splitless overflow mode with an injection of 5 µL injection volume. Oven temperature program started at an initial temperature of 67°C (1.5 min hold), ramped at 15°C/min to 180°C, followed by a second temperature ramp of 5°C/min to 280°C (hold 3 min). Single ion monitoring was used for the detection of PCB congeners using mass spectrometry in electron impact mode (70 eV) with the ion source temperature of 250°C. Ions monitored for both parent and 13C12 mass-labelled internal standards are outlined in Supplementary Table 2.

Analysis of DDT/DDE

Analysis of DDT and DDE isomers was performed by GC/MS) using an Agilent 7890A gas chromatograph (Agilent Technologies, Böblingen, Germany) coupled to a Quattro Micro triple quadrupole mass spectrometer (Waters Coporation, Manchester, United Kingdom). Separation was performed using a Zebron multi-residue1 column (30 m x 250 µm id, 0.25µm film thickness, Phenomenex, Denmark) equipped with a 5 m guard column (Restek Corporation, Bellefonte, PA, USA) at 1ml/min with Helium as a carrier gas. Similar to PCBs, samples injected using a split/splitless injector equipped with a 4 mm id packed (wool) Sky liner (Restek Corporation, Bellefonte, PA, USA) in splitless overflow mode using a 5 µL injection volume at 220°C. Oven temperature program started at an initial temperature of 85°C (1.0 min hold), ramped at 25°C/min to 280°C (hold 2 min), followed by a second temperature ramp of 40°C/min to 300°C (hold 5 min). Detection of DDT and DDE isomers my mass spectrometry used conditions as described for PCB analysis. Parent and 13C12 mass-labelled internal standards are outlined in Supplementary Table 2.

Analysis of organochlorine pesticides

Analysis of organochlorine pesticides was performed by GC/MS using an Agilent 7890A gas chromatograph equipped with a 5975C inert XL mass spectrometer (Agilent Technologies, Böblingen, Germany). Separation was carried out using a DB-Ultra2 column (25 m x 200 µm id, 0.11µm film thickness, J&W, Folsom, USA) equipped with a 5 m guard column (Restek

788 Corporation, Bellefonte, PA, USA) at 1ml/min with Helium as a carrier gas. Injection of samples
789 was carried out under similar conditions as described for DDT/DDE isomers. Oven temperature
790 program started at an initial temperature of 80°C (2 min hold), ramped at 20°C/min to 100°C
791 (hold 5 min), 20°C/min to 170°C (hold 3 min), 5°C/min to 200°C, and a final ramp of 20°C/min
792 to 300 (hold 2 min). Detection of organochlorine pesticides was performed using chemical
793 ionization with methane as a reagent gas at a flow rate of 0.4 ml/min with a source temperature
794 of 160°C. Parent and 13C12 mass-labelled internal standards are outlined in Supplementary
795 Table 2.
796
797 **Supplementary Table 2.** Quantification and qualifier ions for targeted compounds and their
798 13C-mass labelled internal standards.

Compound	**Quantification/Qualifier ions**
PCBs	
Tri-chlorinated PCB	256/258
13C12- Tri-chlorinated PCB	268
Tetra-chlorinated PCB	290/292
13C12- Tetra-chlorinated PCB	302
Penta-chlorinated PCB	324/326
13C12- Penta-chlorinated PCB	336
Hexa-chlorinated PCB	360/362
13C12- Hexa-chlorinated PCB	370
Hepta-chlorinated PCB	392,394
13C12- Hepta-chlorinated PCB	404
DDT/DDE isomers	
o,p-DDE	246/248
p,p-DDE	246/248
13C12-p,p-DDE	258
o,p-DDD	235/237
p,p-DDD	235/237
o,p-DDT	235/237
p,p-DDT	235/237
13C12 p,p-DDT	247
Organochlorine pesticides	
HCH isomers	253/255

13C6-HCH isomers	260 799
HCB	284/286
13C6 -HCB	290
Oxychlordane	350/352
trans-chlordane	408/410
cis-chlordane	408/410
13C12 –cis-chlordane	418
trans-nonachlor	442/444
cis-nonachlor	442/444
13C12-cis-nonachlor	452
Mirex	441/439
13C12-Mirex	449

800

801 **Quality Assurance**

802

803 To avoid the reporting of false positives, concentrations were only reported for targeted analytes
804 in which both quantification and qualifier ions were detected. In addition, target compounds
805 detected in sample extracts must possess ion ratio between quantification and qualifier within
806 20% to the ion ration observed within the quantification standard. A minimum of three method
807 blanks were extracted with each sample batch to calculate limits of detection per extraction
808 batch. Extraction recoveries for internal standards from sample extracts are listed in
809 Supplementary Table 3.

810

811 **Supplementary Table 3.** Internal standard recoveries (%) and variation within sample extracts.

Compound	Average Recovery (%) and variation
13C12-PCB 28	94 ± 19
13C12-PCB 52	93 ± 16
13C12-PCB 101	90 ± 16
13C12-PCB 118	90 ± 15
13C12-PCB 138	86 ± 16
13C12-PCB 153	88 ± 17
13C12-PCB 180	90 ± 18
13C12-p,p'-DDE	92 ± 14
13C12-p,p'-DDT	59 ± 23

13C6-αHCH	81 ± 17
13C6-βHCH	75 ± 24
13C6-γHCH	85 ± 19
13C6-HCB	80 ± 19
13C12-trans-chlordane	99 ± 22
13C12-cis-chlordane	96 ± 21
13C12-trans-nonachlor	101 ± 24
13C12-cis-nonachlor	84 ± 14
13C12-trans-nonachlor	94 ± 20

812