Plasma concentrations of organohalogenated contaminants in white-tailed eagle nestlings – The role of age and diet

Mari Engvig Løseth[a]*, Nathalie Briels[a], Igor Eulaers[b], Torgeir Nygård[c], Govindan Malarvannan[d], Giulia Poma[d], Adrian Covaci[d], Dorte Herzke[e], Jan Ove Bustnes[f], Gilles Lepoint[g], Bjørn Munro Jenssen[a,b], Veerle L. B. Jaspers[a]

[a]Department of Biology, Norwegian University of Science and Technology (NTNU), Høgskoleringen 5, 7491 Trondheim, Norway

[b]Department of Bioscience, Aarhus University, Fredriksborgvej 399, 4000 Roskilde, Denmark

[c]Norwegian Institute for Nature Research (NINA), Høgskoleringen 9, 7034 Trondheim, Norway

[d]Toxicological Centre, University of Antwerp, Universiteitsplein 1, 2610 Wilrijk, Belgium

[e]Norwegian Institute for Air Research (NILU), FRAM - High North Research Centre on Climate and the Environment, 9007 Tromsø, Norway

[f]Norwegian Institute for Nature Research (NINA), FRAM - High North Research Centre on Climate and the Environment, 9007 Tromsø, Norway

[g]Laboratoire d´Océanologie, University of Liège, 4000 Liège (Sart-Tilman), Belgium

* Corresponding author: mari.loseth@ntnu.no

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

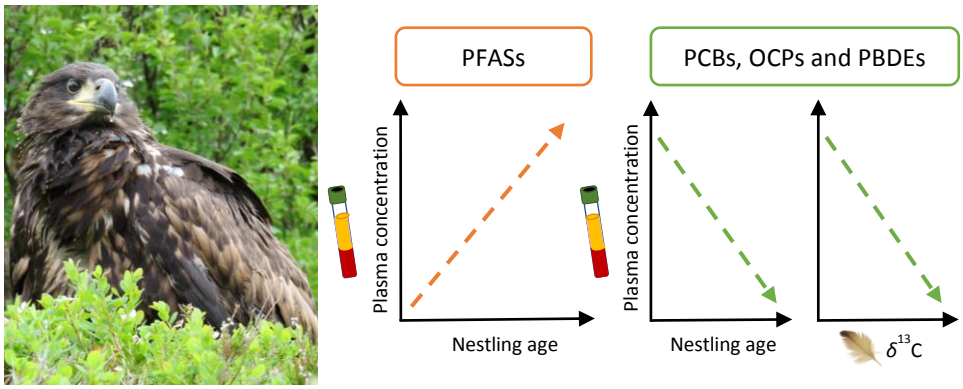
Highlights:

- Significant temporal and spatial variations were found for all compound groups
- Age was the most important predictor for contaminant variation in nestling plasma
- Concentrations of legacy PCBs, OCPs and PBDEs decreased with age
- Concentrations of PFASs increased with age
- $\delta^{13}C$ significantly predicted the variation of legacy PCBs, OCPs and PBDEs

Keywords:

Temporal; spatial; growth dilution; stable isotopes; *Haliaeetus albicilla*; pollution

Graphical abstract



Abstract

Concentrations of organohalogenated contaminants (OHCs) can show significant temporal and spatial variation in the environment and wildlife. Most of the variation is due to changes in use and production, but environmental and biological factors may also contribute to the variation. Nestlings of top predators are exposed to maternally transferred OHCs in the egg and through their dietary intake after hatching. The present study investigated spatial and temporal variation of OHCs and the role of age and diet on these variations in plasma from Norwegian white-tailed eagle (*Haliaeetus albicilla*) nestlings. The nestlings were sampled at two locations, Smøla and Steigen, in 2015 and 2016. The age of the nestlings was recorded (range: 44 – 87 days old) and stable carbon and nitrogen isotopes (δ^{13}C and δ^{15}N) were applied as dietary proxies for carbon source and trophic position, respectively. In total, 14 polychlorinated biphenyls (PCBs, range: 0.82 – 59.05 ng/mL), 7 organochlorinated pesticides (OCPs, range: 0.89 – 52.19 ng/mL), 5 polybrominated diphenyl ethers (PBDEs, range: 0.03 – 2.64 ng/mL) and 8 perfluoroalkyl substances (PFASs, range: 4.58 – 52.94 ng/mL) were quantified in plasma samples from each location and year. The OHC concentrations, age and dietary proxies displayed temporal and spatial variations. The age of the nestlings was indicated as the most important predictor for OHC variation as the models displayed significantly decreasing plasma concentrations of PCBs, OCPs, and PBDEs with increasing age, while concentrations of PFASs were significantly increasing with age. Together with age, the variations in PCB, OCP and PBDE concentrations were also explained by δ^{13}C and indicated decreasing concentrations with a more marine diet. Our findings emphasise age and diet as important factors to consider when investigating variations in plasma OHC concentrations in nestlings.

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

1. Introduction

Organohalogenated contaminants (OHCs) are a diverse group of chemicals that have been used in lubricants, pesticides, flame retardants and surface treatments (Mackay et al., 2006). OHCs include legacy compounds such as polychlorinated biphenyls (PCBs), as well as emerging compounds such as per- and polyfluoroalkyl substances (PFASs). By being resistant to chemical and biological degradation, OHCs persist in the environment (Muir and de Wit, 2010; UNEP, 2009). While most legacy OHCs are lipophilic, the emerging PFASs are amphipathic due to a different chemical structure with a hydrophilic functional group (Lau et al., 2007) . Even so, their physicochemical properties and persistency result in high potential for bioaccumulation and biomagnification through food chains (Borgå et al., 2004, 2012). The concentrations of OHCs can show significant temporal and spatial variations both in the environment and wildlife (Faxneld et al., 2016; Helgason et al., 2008; Hung et al., 2016; Wierda et al., 2016). Most of these variations are due to changes in production and use of the compounds (Hung et al., 2016; Wang et al., 2014). However, environmental and biological factors can also contribute significantly to the observed variations (Bourgeon et al., 2013; Bustnes et al., 2015; Leat et al., 2011).

The white-tailed eagle (*Haliaeetus albicilla*) occupies a high trophic level and can accumulate a wide range of OHCs, even at an early age (Bustnes et al., 2013; Eulaers et al., 2014; Løseth et al., 2019; Sletten et al., 2016). Nestlings are exposed to maternally transferred OHCs during development in the egg (Faxneld et al., 2016; Nordlöf et al., 2010; Nygård and Polder, 2012) and the exposure continues after hatching through their dietary intake (Bourgeon et al., 2013). Adult white-tailed eagles are mostly resident within their breeding areas (Willgohs, 1984), thus the contaminant burdens of their eggs and nestlings reflect contaminant levels in local prey. This makes white-tailed eagle nestlings good sentinels of local environmental pollution (Helander et al., 2008; Olsson et al., 2000).

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

The diet of the white-tailed eagle consists mainly of marine fish and seabirds (Koivusaari et al., 1976; Willgohs, 1984), which may have accumulated high concentrations of OHCs. As the diet is a major source of OHC exposure following hatching, stable isotopes of nitrogen (δ^{15}N) and carbon (δ^{13}C) are often applied as dietary proxies to investigate the nestlings' trophic position and dietary carbon source, respectively (Fry, 2006; Inger and Bearhop, 2008). The ratio of ^{15}N to ^{14}N increases by about 2-5 ‰ per trophic level as the lighter nitrogen isotopes are excreted through nitrogenous waste products. The ratio of ^{13}C to ^{12}C can also increase with increasing trophic level, though it is mostly used to distinguish between marine and terrestrial dietary carbon sources. Terrestrial primary producers have lower δ^{13}C values compared to marine ones. This is reflected in the tissues of their consumers and persists at higher trophic levels within the food chain (Fry, 2006; Inger and Bearhop, 2008; Kelly, 2000). Keratinized matrices, such as feathers, are metabolically inert after their growth and can preserve the stable isotopes deposited into the matrix during its growth (Inger and Bearhop, 2008). A homogenate of nestling feathers can therefore provide information about their diet during the growth period of the feathers (Bearhop et al., 2002).

As many OHCs have been shown to interfere with physiological processes linked to development and growth (Cassone et al., 2012; Jenssen et al., 2010; Nøst et al., 2012), there is special concern about levels and effects of these compounds in young developing birds. As nestlings develop and grow, their maternally transferred contaminants are significantly diluted by their growth (Bourgeon et al., 2013; Bustnes et al., 2013). However, nestlings are also exposed to OHCs through their diet and plasma concentrations of compounds with high ability for bioaccumulation may increase as the nestlings reach their adult body size at fledging (Borgå et al., 2004; Bustnes et al., 2013). Previously, only few studies have accounted for age and growth when investigating OHCs in nestlings (Bourgeon et al., 2013; Bustnes et al., 2013; Dauwe et al., 2006; Olsson et al., 2000). In the present study, we aimed

106 to investigate variations of OHC concentrations in plasma from white-tailed eagle nestlings
107 sampled from two locations in two consecutive years. Secondly, we aimed to explore if
108 variation in dietary proxies (δ^{13}C and δ^{15}N) and biological variables (such as age of the
109 nestlings) could account for parts of the spatial and temporal variation of these OHCs. As the
110 diet is the major source of OHCs, we expected to find a strong influence of the dietary
111 proxies presenting increased plasma OHCs with increasing δ^{15}N (higher trophic position) and
112 increasing δ^{13}C (more marine prey). Thus, we also expected to find small differences in
113 OHCs in nestlings from the two locations as habitat differences may also influence the
114 diversity of prey species at the two locations. No differences were expected between the two
115 sampling years, as to the authors knowledge there are no local sources of OHCs at the two
116 locations. We also expected to find higher concentrations in plasma of older and/or larger
117 nestlings as OHCs have a high potential for bioaccumulation.

118 ## 2. Materials and methods

119 The plasma OHC concentrations of the individual OHCs have been published previously
120 (Løseth et al., 2019, supplementary information), in a study where three non-invasive
121 matrices (plasma, feathers and preen oil) from white-tailed eagle nestlings were compared for
122 legacy and emerging contaminants. In the current study, however, we present unpublished
123 data on stable isotopes and age to explain variation in the plasma concentrations of ΣPCBs,
124 ΣOCPs, ΣPBDEs and ΣPFASs.

125 ### 2.1. Field sampling

126 The study was conducted on 70 white-tailed eagle nestlings from two archipelagos in
127 Norway, Smøla (63.3-63.5°N; 7.8-8.2°E) and Steigen (67.7-67.9°N; 14.6-14.8°E), during the
128 breeding seasons of 2015 and 2016 (Figure 1). We sampled 35 nestlings both from Smøla
129 (2015: n = 13, 2016: n = 22) and Steigen (2015: n = 14, 2016: n = 21) during June-July of

these two years (see supplementary information (SI), Table S1 for details). Sex determination was based upon morphometric measurements (Helander et al., 2007), while the age was estimated from the tail feather length. The tail feather emerges at day 30 and grows with (mean ± SE) 4.95 ± 0.02 mm per day (Pers. comm. Torgeir Nygård). Wing length has previously been used to estimate age in Swedish white-tailed eagle nestlings (Helander et al., 2007) and in our study wing and tail feather length were strongly correlated (r_{70} = 0.94, p < 0.01). All nestlings were sampled for body feathers and blood as described in Løseth et al. (2019). Body feathers were gently pulled from the dorsal region and stored in polyethylene zipper bags (VWR, USA) at -20°C. A blood sample of 8 mL was collected in heparinised vacutainers through brachial venepuncture. The blood samples were centrifuged at 860 *g* and plasma was transferred into cryogenic tubes (Nalgene®, USA) and stored at -20 °C. The sampling was approved by the Norwegian Food Safety Authority (Mattilsynet; 2015/6432 and 2016/8709) and the handling of the birds were in accordance with the regulations of the Norwegian Animal Welfare Act.

2.2. Stable isotope analyses

We analysed stable isotopes in the body feathers, which were still growing at the time of sampling and thus connected to the blood circulation at the calami. The analysis for bulk feather stable carbon (^{12}C and ^{13}C) and nitrogen isotopes (^{14}N and ^{15}N) was performed at the MARE Centre of the University of Liège, Belgium. Clean stainless steel and glass tools were used to remove the calami and for washing and cutting of the feathers. The tools were thoroughly rinsed with acetone between individuals. Feathers were washed in Milli-Q water as previously described in Løseth et al. (2019) to remove dust and particles from feathers prior to analysis. A subsample of homogenised cleaned feather material (mean ± SD: 1.55 ± 0.37 mg) was wrapped into a tin combustion cup and analysed for its elemental and isotopic composition using a vario MICRO cube elemental analyser (Elementar Analysen systeme

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

GmBH, Hanau, Germany) coupled to an IsoPrime100 mass spectrometer (Isoprime, Cheadle, United Kingdom). The reported stable carbon and nitrogen isotope values are expressed as δ (‰) relative to the international reference standards Vienna PeeDee Belemnite and atmospheric nitrogen, respectively. An internal reference material (i.e., glycine) was measured for every tenth sample and revealed an imprecision (±1 SD) of 0.23 and 0.16 ‰ for $\delta^{13}C$ and $\delta^{15}N$, respectively.

2.3. Chemical analyses

The targeted compounds for the analyses were polychlorinated biphenyls (PCB; IUPAC congeners 28, 49, 52, 74, 95, 99, 101, 105, 110, 118, 138, 149, 153, 156, 170, 171, 177, 180, 183, 187, 194, 206 and 209) and organochlorinated pesticides (OCPs; dichlorodiphenyltrichloroethane (*p,p'*-DDT), *p,p'*-dichlorodiphenyldichloroethylene (*p,p'*-DDE), three isomers of hexachlorocyclohexane (α-, β-, and γ-HCH), chlordanes (*oxy*-chlordane (OxC), *cis*-nonachlor (CN) and *trans*-nonachlor (TN)) and hexachlorobenzene (HCB)). The targeted legacy flame retardants were polybrominated diphenyl ether (PBDE) congeners; BDE 28, 47, 99, 100, 153, 154 and 183. The targeted perfluoroalkyl substances (PFASs) were perfluorobutanoic acid (PFBA), perfluoropentanoic acid (PFPeA), perfluorohexanoic acid (PFHxA), perfluoroheptanoic acid (PFHpA), perfluorooctanoic acid (PFOA), perfluorononanoic acid (PFNA), perfluorodecanoanoic acid (PFDcA), perfluoroundecanoic acid (PFUnA), perfluorododecanoic acid (PFDoA), perfluorotridecanoic acid (PFTrA), perfluorotetradecanoic acid (PFTeA), perfluorooctanesulfonamide (PFOSA), perfluorobutane sulfonate (PFBA), perfluoropentane sulfonate (PFPS), perfluorohexane sulfonate (PFHxS), perfluoroheptane sulfonate (PFHpS), linear and branched perfluorooctane sulfonate (Lin-PFOS and Br-PFOS) and perfluorononane sulfonate (PFNS).

Procedures used for the extraction and quantification have been described in detail by Løseth et al. (2019). In brief, PCBs, OCPs and PBDEs were extracted using *n*-hexane:dichloromethane (DCM, 1:1, v:v) and fractionation was performed on Supelclean^{TM} ENVI Florisil cartridges (500 mg, 3 mL, Supelco® Analytical). The compounds were eluted with *n*-hexane:DCM and quantified according to Eulaers et al. (2011a). PFASs were extracted with methanol using the Powley method (Powley et al., 2005) and quantified according to Herzke et al. (2009). Internal standards and their recoveries are listed in SI (Table S2 and S3) and ranged from 30 – 118 % for PCBs, 41 – 90 % for OCPs, 74 – 97 % for PBDEs, and 59 – 101 % for PFASs. For every tenth plasma sample, a procedural blank was analysed to control for background contamination. To control the performance of the analytical method of the PCB, OCP and PBDE extraction, a human plasma sample from the Arctic Monitoring and Assessment Programme interlaboratory exercise was analysed for every 20th sample. For PFAS extractions, a commercially available human plasma sample (NIST SRM 1957, USA) was analysed for every tenth sample. No background contamination was encountered in the blanks for any of the analysed PFASs. For legacy POPs not detectable in the blanks, the limits of quantification (LOQs) were set to ten times the signal-to-noise ratio of sample runs or were calculated as three times the standard deviation of the procedural blanks for each compound. For PFASs, the LOQs were calculated as three times the signal-to-noise ratio of the procedural blanks for each compound. The LOQs for all compounds are available in the SI (Tables S4-S6).

2.4. Statistical analyses

The statistical analyses were performed using R (v. 3.4.2, R Development Core Team, 2008). The compounds that could be quantified in more than 50 % of the samples within each year and location were 14 PCB congeners (CB 99, 101, 105, 118, 138, 153, 156, 170, 171, 177, 180, 183, 187 and 194), seven OCPs (OxC, TN, CN, *p,p'*-DDE, *p,p'*-DDT, HCB and *β*-

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

HCH), five PBDE congeners (BDE 47, 99, 100, 153 and 154) and eight PFASs (Br-PFOS, Lin-PFOS, PFOA, PFNA, PFDcA, PFUnA, PFDoA and PFTriA) (Table 1 and Table S7). Data below the limit of quantification (LOQ) were substituted with LOQ * detection frequency (Voorspoels et al., 2002) for each compound. Profiles of the compounds included in the statistical analyses are available in Figure S1. Due to the structure of the data, with two to three chicks in some nests, only statistical tests from the *nlme*: Linear and nonlinear mixed effect models package (Pinheiro et al., 2018) were applied and nest identity was always included as a random variable to avoid pseudoreplication of nestlings within nests. Statistical significance was assumed at $\alpha = 0.05$.

Due to collinearity between compounds within each contaminant group (Table S8 and S9), compounds were summed (Σ) per group (Σ_{14}PCBs, Σ_{7}OCPs, Σ_{5}PBDEs and Σ_{8}PFASs) for statistical modelling. All variables were investigated for influential outliers, normality and homoscedasticity (Zuur et al., 2010). Variables that were not normally distributed were $\log_e$ transformed to meet criteria of parametric statistics. To ensure normality of the residuals of the model, two outliers were removed from the OCP modelling. These outliers were two young individuals sampled in Steigen in 2015 (47.2 and 52.4 days old) which also had the highest plasma concentrations of OCPs (46.3 and 52.2 ng/mL, respectively).

Age was included as an explanatory variable, instead of body mass or body condition due to multicollinearity. It is important to note that each nestling was only sampled once and to investigate the true variation with increasing age it is preferred to sample the same individuals repeatedly. A detailed description of the calculation of body condition and correlations between age, body mass and body condition can be found in the SI. Body mass, size and age are all correlated when the nestlings are growing, but body mass may show large variations between sexes and on an individual level due to different climates, habitats, diets

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

and parental experience. Age presents a more stable variable as it, on an individual level, can only increase, regardless of sex and diet.

Correlations between $\log_e \Sigma$ contaminant groups, age, $\delta^{13}C$ and $\delta^{15}N$ were investigated using Pearson correlation coefficient test. A strong correlation was detected between $\delta^{15}N$ and $\delta^{13}C$ ($r_{70} = 0.76$, $p < 0.01$, Figure S3), but both variables were included in the first model selection as they represent trophic position and dietary source, respectively. To investigate temporal and spatial variation of Σ_{14}PCBs, Σ_7OCPs, Σ_5PBDEs, Σ_8PFASs, age, $\delta^{13}C$ and $\delta^{15}N$, linear mixed effect analyses of variance (Lme-Anovas) were applied with location, year and the interaction between location and year as explanatory variables (Table S10). Tukey's honestly significant difference (HSD) post hoc test was applied to investigate differences in age between locations and years.

To investigate how age and the dietary proxies may contribute to the observed temporal and spatial variation, we performed linear mixed effect models for each compound group. The initial full model included location, year, the interaction between location and year, age, $\delta^{15}N$ and $\delta^{13}C$. The most parsimonious models were selected using Akaikes Information Criterion for small sample sizes (AICc). Each model was analysed for variance inflation factors (VIF) with a threshold of VIF < 3 to identify problems with collinearity among explanatory variables (Zuur et al., 2009, 2010). The model selection showed that the effect of $\delta^{15}N$ was only significant with the presence of $\delta^{13}C$ in the model, and VIF values for $\delta^{15}N$ were over 3 for some of the models. This may be due to the significant correlation detected between the two stable isotopes. For the final model selection, we therefore chose to include only $\delta^{13}C$, age, location, year and the interaction between location and year. Model selection was performed on models fitted with maximum likelihood (ML), while parameters were estimated using restricted maximum likelihood (REML). Models with ΔAICc < 2 are discussed below.

In addition to AICc, marginal pseudo-R^2 (R^2_m; explaining the variation of the fixed factors) and conditional pseudo-R^2 (R^2_c; explaining the variation of both fixed and random factors) were extracted according to Nakagawa and Schielzeth (2013).

3. Results and discussion

3.1. Organohalogenated contaminants

The compound groups found with the highest median wet weight concentrations in plasma were PFASs > PCBs > OCPs > PBDEs. Within each compound group, the compounds with the highest concentrations were linear PFOS (3.86 – 31.85 ng/mL), CB 153 (0.21 – 26.27 ng/mL), *p,p′*-DDE (0.48 – 47.61 ng/mL) and BDE 47 (0.01 – 1.82 ng/mL), respectively (Table S7). The concentrations of Σ_{14}PCBs, Σ_7OCPs, Σ_5PBDEs and Σ_8PFASs (Table 1, Figure S2A) were lower than or within the same range of those previously reported in plasma from white-tailed eagle nestlings from Norway (Bustnes et al., 2013; Eulaers et al., 2011a, 2011b, 2013, 2014; Gómez-Ramírez et al., 2017).

3.2. Nestling age and dietary proxies

The age span of the nestlings varied significantly between locations and years, although the nestlings were sampled within the same two calendar weeks each year (Table 1, Figure S2B). In 2015, the nestlings from Smøla were on average 79 days old, which was 15 days older than those from Steigen ($z = 3.5$, $p < 0.01$). The Smøla nestlings sampled in 2015 were also 13 days older than those sampled at Smøla and Steigen in 2016 ($z = 3.2 – 3.4$, $p < 0.01$, Table S10). In 2016, there were no significant age differences between the nestlings sampled at Smøla and Steigen. We also found significantly higher δ^{15}N and δ^{13}C, as well as narrower dietary niches, in nestlings from 2015 than in nestlings from 2016 ($F_{(1,44)} = 8.8$ and 4.9, $p < 0.01$, respectively, Figure S3, Table 1). The results also showed that the nestlings from

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

Steigen fed on a diet more enriched in ^{15}N than those from Smøla ($F_{(1,44)} = 15.7$, $p < 0.01$, Figure S3), indicating that the Steigen nestlings may have been feeding on a higher trophic position. The temporal variation found for both stable isotopes may indicate a slight change in prey species between the two years at both locations. Within both years, some birds from Smøla and Steigen had $\delta^{13}C$ values lower than -20 ‰ which indicates the influence of more terrestrial prey in their diet (Fry, 2006). This was coherent with the observed prey remains around their nests, which, besides from fish and seabirds, consisted of terrestrial species such as greylag goose (*Anser anser*), hare (*Lepus timidus*) and hedgehogs (*Erinaceus europaeus*). The interannual dietary changes reported here are not uncommon for opportunistic feeders such as white-tailed eagles (Inger and Bearhop, 2008), as it can correspond to variations in availability of prey species.

3.3. Model selection to best explain OHC variation

The results from the model selection confirmed age and diet as important predictors for the temporal and spatial variation of legacy OHCs observed in the initial analyses (Table S10) as they were included in all the most parsimonious models for PCBs, OCPs and PBDEs (Table 2, see Table S11 - S13 for all competing models). For PFASs on the other hand, only age was selected as an important predictor for the observed temporal and spatial variation (Table S10) as it was included in all the most parsimonious models for PFASs variation (Table 2, see Table S14 for all competing models). It is important to note that these results are statistical models which are estimating the OHC variation and in order to investigate the true OHCs variation with increasing age, repeated sampling is necessary.

3.3.1 Legacy OHC variation

Contrary to our hypothesis, the models for Σ_{14}PCBs, Σ_{7}OCPs and Σ_{5}PBDEs indicated significantly lower concentrations of legacy OHCs in older nestlings and in nestlings with a

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

diet more enriched in ^{13}C (i.e. more marine prey; Figure 2). Some of these models also included location, year and the interaction between location and year, which contributed to a better fit of the model. The results of the lme-Anova showed significant temporal and spatial variation in PCB, OCP and PBDE levels (Table S10), however when we accounted for age and diet in the model selection, the temporal and spatial variations for PCBs and PBDEs were not significant anymore (Table 2). It was only for Σ_7OCPs that the estimates indicated significantly higher concentrations in nestlings from Steigen than those from Smøla (p = 0.01), as well as significantly higher concentration in nestlings from Steigen in 2015 than in 2016 (p = 0.03). In contrast to what was observed for Σ_{14}PCBs and Σ_5PBDEs, the effect of age was not statistically significant for Σ_7OCPs (β_1 = 0.012, p = 0.07). However, it is important to mention that for these models two of the youngest and most contaminated individuals were excluded from the analyses to ensure normality of the residuals, and that the inclusion of these outliers resulted in a significant effect of age on Σ_7OCPs (β_1 = 0.018, p = 0.03). This should therefore be considered in the interpretation of the estimates of the Σ_7OCP models.

3.3.1.1 Influence of age

The inverse relationship between plasma legacy OHC concentrations and age found in the present study was in accordance with previous reports for CB 153 and *p,p'*-DDE in plasma of white-tailed eagle nestlings (Bustnes et al., 2013), plasma levels of PCBs and PBDEs in great tit (*Parus major*) nestlings (Dauwe et al., 2006) and liver concentrations of PCBs, *p,p'*-DDE and HCB in European shag (*Phalacrocorax aristotelis*) nestlings (Jenssen et al., 2010; Murvoll et al., 2006). In contrast, a previous study on white-tailed eagle nestlings did not find decreased PCB or *p,p'*-DDE concentrations in plasma of older nestlings (Olsson et al., 2000), neither did a study of PBDEs in plasma of bald eagle nestlings (Guo et al., 2018). The nestlings from the present study were on average 69 days old (range: 44 – 87 days old), while

most of the nestlings from Olsson et al. (2000) were less than 57 days old (range: < 36 – 57 days old) and from Guo et al. (2018) were on average 46 days old (range: 28 – 56 days old). Our significant effect of age may be due to the greater age span, larger sample size and homogenous age classes in the present study, thus allowing more time for growth dilution or changes in metabolic capability/excretion in older nestlings and a higher statistical probability to detect such changes.

Even though nestlings are continuously exposed to OHCs through their diet, a study on experimental feeding of great skua chicks (*Stercorarius skua*) found that their contaminant load was more influenced by maternal than trophic transfer regardless of diet (Bourgeon et al., 2013). A study of paired egg and plasma samples of bald eagled from the Great Lakes between 2000 and 2012 found that egg concentrations of PBDEs were over 30 times higher than the plasma concentrations of nestlings from the same nests (Guo et al., 2018). Nygård and Polder (2012) also found very high concentrations of PCBs (mean: 2839 ng/g fresh weight (fw)) and *p,p'*-DDE (mean: 950 ng/g fw) in white-tailed eagle eggs sampled in Norway between 2005 and 2010. Although egg and plasma concentrations cannot be directly compared, these reported concentrations were several folds higher than the plasma concentrations found in the present study. As concentrations in plasma reflect internal concentrations in the nestling, we propose that the decreasing legacy OHC concentrations with increasing age may be due to growth dilution of maternally derived compounds deposited with high concentrations in the eggs.

3.3.1.2 Influence of diet

Our results also indicated decreasing Σ_{14}PCBs, Σ_{7}OCPs and Σ_{5}PBDEs concentrations with increasing δ^{13}C, which corresponds with previous reports of decreases in CB 153, *p,p'*-DDE and HCB in white-tailed eagle nestlings with diets more enriched in ^{13}C (Bustnes et al.,

2013). Bustnes et al. (2013) explained this relationship by the depleted ^{13}C levels found in lipids compared to proteins (Post et al., 2007) and suggested that the diet of the more contaminated nestlings may have contained more lipid-rich prey, such as gulls (*Laridae*), which may also have contained higher concentrations of biomagnifying OHCs (Bustnes et al., 2013). Surprisingly, the more contaminated nestlings from Smøla were feeding on a lower trophic position (depleted in ^{15}N) and terrestrial prey remains were surrounding their nest which were located more inland on the island. The contaminant concentrations in these nestlings may therefore have been highly influenced by maternally derived OHCs (Bourgeon et al., 2013). White-tailed eagles have been reported to change their diet in the winter according to the availability of prey species (Willgohs, 1984). It is therefore possible that the mothers of these nestlings have fed on a diet more enriched in lipids, containing higher concentrations of OHCs, during the winter months and before egg laying. Such seasonal dietary changes of the mothers may influence the concentrations of legacy OHCs in their eggs and subsequently in their nestlings (Bourgeon et al., 2013). In contrast, stable isotopes deposited in the keratin in nestling feathers originate mostly from their diet and not from maternal transfer (Bearhop et al., 2002). Although we cannot be certain whether such a dietary change has taken place, one should always keep in mind that the stable isotopes analysed in feathers only reflect the diet in the period during which they were grown (Bearhop et al., 2002).

A study on bald eagle nestlings also found that $\delta^{13}C$ was generally a better predictor of legacy OHC concentrations than $\delta^{15}N$ in eagles from marine environments, even when the two stable isotope ratios were correlated (Elliott et al., 2015). This was confirmed by the results in the current study as the final model selection did not include $\delta^{15}N$ and no significant correlations were found between $\delta^{15}N$ and the OHC groups. However, significant positive correlations between $\delta^{15}N$ or trophic level and several legacy POPs have been found in previous studies

372 on both white-tailed eagle (Bustnes et al., 2013; Eulaers et al., 2013, 2014) and bald eagle
373 nestlings (*Haliaeetus leucocephalus*; Elliott et al., 2015).

374 3.3.2. PFAS variation

375 Contrary to the legacy OHCs models, the models for PFASs indicated no significant effect of
376 $\delta^{13}C$ on PFAS concentrations in plasma and the most parsimonious model included age,
377 location and year (Table 2, Figure 3). These results were not unexpected as PFASs have
378 different physicochemical properties than legacy OHCs and may therefore have different
379 exposure routes and toxicokinetics (Lau et al., 2007).

380 *3.3.2.1 Influence of age*

381 Interestingly, we found opposite age-related effects for PFASs than for PCBs, OCPs and
382 PBDEs, confirming our initial hypothesis of increasing plasma concentrations with increasing
383 age. A similar increase with age has also been reported earlier for PFOS in white-tailed eagle
384 nestlings (Bustnes et al., 2013) and for PFNA and PFUnA in bald eagle nestlings (Route et
385 al., 2014). The PFAS concentrations in the current study were also similar to the
386 concentrations found in white-tailed eagle eggs from Norway in 2005 – 2010 (mean: 55.3
387 ng/g fw; Nygård and Polder, 2012), which suggest that maternal transfer may be of less
388 importance for PFAS exposure than for the legacy OHCs. The increasing PFAS
389 concentrations with age are therefore more likely originating from dietary sources, than from
390 maternal transfer, as maternally deposited concentrations are diluted by growth regardless of
391 the physicochemical properties of the compounds (Bustnes et al., 2013).

392 *3.3.2.2 Spatial variation*

393 The model estimates also indicated significantly higher PFAS concentrations in nestlings
394 from Steigen than in those from Smøla (Table 2, $p < 0.01$). At the same time, significantly

higher $\delta^{15}N$ were detected in nestlings from Steigen than nestlings from Smøla as well as significant correlations between PFAS concentrations and $\delta^{13}C$ (r_{70} = 0.25, p = 0.03) and $\delta^{15}N$ (r_{70} = 0.44, p < 0.01). Thus, we cannot exclude trophic position as an important factor influencing this PFAS variation. Nevertheless, the absence of stable isotopes in the most parsimonious PFAS models corresponds with previous reports in plasma from Norwegian white-tailed eagle nestlings (Bustnes et al., 2013; Gómez-Ramírez et al., 2017) and several seabirds (Gebbink et al., 2011; Haukås et al., 2007; Leat et al., 2013; Miller et al., 2015; Vicente et al., 2015).

3.3.2.3 Temporal variation

The model also indicated significantly higher PFAS concentrations in nestlings sampled in 2015 than in 2016, at both locations (Table 2, p < 0.01). This interannual variation corresponds with a previous study on white-tailed eagle nestlings from Troms and Vesterålen, Norway in 2011 and 2012 (Sletten et al., 2016). The authors of that study suggested dietary differences as the main reason for that variation (Sletten et al., 2016), which corresponds with the present study as we also detected significant differences in stable isotopes between years. Interestingly, the difference between 2015 and 2016 in PFAS plasma concentrations in the present study also corresponds with reports on PFASs in air, where higher concentrations of several PFASs were found at three monitoring stations in Norway in 2015 compared to 2016 (Bohlin-Nizzetto et al., 2017; Bohlin-Nizzetto and Aas, 2016). Thus, yearly differences in long range transport of PFASs and its precursors may play a role, as they can be subsequently taken up into the food web (Houde et al., 2011) and their top predators (Bustnes et al., 2015). To our knowledge, there are no significant PFAS sources at the two locations that may influence PFASs concentrations in the white-tailed eagle nestlings. However, due to the significantly higher stable isotope values in nestlings from 2015 and correlation between $\delta^{15}N$ values and PFAS concentrations, we suggest a combination of

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

PFAS exposure from long range transport and dietary sources as important factors explaining this temporal variation.

4. Conclusions

In the present study, we report age as one of the most important predictors for spatial and temporal variation of OHCs in plasma from white-tailed eagle nestlings from Smøla and Steigen, Norway. It is important to note that the nestlings in the present study were only sampled once, and that the models were based on results from nestlings ranging from 44 to 87 days old. Our results indicated lower plasma concentrations of PCBs, PBDEs and OCPs in older nestlings, while the concentrations of PFASs were higher in the older nestlings. The variations in PCBs, OCPs and PBDEs were also significantly explained by the dietary carbon source (δ^{13}C), indicating that nestlings feeding on a diet with more marine prey had lower plasma concentrations of these compounds. The stable isotope ratio of nitrogen (δ^{15}N) was of less importance in the present study, however it indicated that nestlings from Steigen were feeding at a higher trophic position than those from Smøla. We also found higher stable isotope ratios in nestlings sampled in 2015 compared to 2016 which may suggest dietary differences. Overall, our results indicate a need to take age into consideration when investigating OHC concentrations in bird of prey nestlings, regardless of the sample matrix (as strong correlations were found between concentrations of PCBs, OCPs and PBDEs in feathers, plasma and preen oil; see Løseth et al., 2019). Our results also indicate that diet may contribute to variations in plasma OHC concentrations, especially for PCBs, OCPs and PBDEs in opportunistic birds such as the white-tailed eagle.

5. Acknowledgements

The authors acknowledge the Norwegian Research Council and NTNU for funding Mari E. Løseth, Nathalie Briels, Veerle L.B. Jaspers and the NewRaptor project (# 230465). Giulia

444 Poma and Govindan Malarvannan acknowledge funding from the University of Antwerp for
445 their post-doc fellowships. Trond V. Johnsen, Paula Marcinekova, Jørgen Flo, Courtney
446 Waugh, Espen L. Dahl, Johannes Schrøder and Aasmund Gylseth are acknowledged for their
447 assistance during sample collection. We thank Grethe S. Eggen, Lene N. Torgersen from
448 NTNU and Linda Hanssen from NILU for assisting with the chemical analyses. Additional
449 funding for fieldwork in Steigen was provided by the Hazardous Substances Flagship (the
450 Raptor project) at the Fram Center in Tromsø.

451 6. References

452 Bearhop, S., Waldron, S., Votier, S.C., Furness, R.W., 2002. Factors That Influence
453 Assimilation Rates and Fractionation of Nitrogen and Carbon Stable Isotopes in Avian
454 Blood and Feathers. Physiol. Biochem. Zool. 75, 451–458. doi:10.1086/342800

455 Bohlin-Nizzetto, P., Aas, W., 2016. Monitoring of environmental contaminants in air and
456 precipitation, annual report 2015. Norwegian Environmental Agency. ISBN: 978-82-
457 425-2841-4

458 Bohlin-Nizzetto, P., Aas, W., Warner, N., 2017. Monitoring of environmental contaminants
459 in air and precipitation, annual report 2016. Norwegian Environmental Agency. ISBN:
460 978-82-425-2888-9

461 Borgå, K., Fisk, A.T., Hoekstra, P.E., Muir, D.C.G., 2004. Biological and chemical factors of
462 importance in the bioaccumulation and trophic transfer of persistent organochlorine
463 contaminants in Arctic marine food webs. Environ. Toxicol. Chem. 23, 2367–2385.
464 doi:10.1897/03-518

465 Borgå, K., Kidd, K.A., Muir, D.C.G., Berglund, O., Conder, J.M., Gobas, F.A.P.C.,
466 Kucklick, J., Malm, O., Powell, D.E., 2012. Trophic magnification factors:

Considerations of ecology, ecosystems, and study design. Integr. Environ. Assess. Manag. 8, 64–84. doi:10.1002/ieam.244

Bourgeon, S., Leat, E.K.H., Furness, R.W., Borgå, K., Hanssen, S.A., Bustnes, J.O., 2013. Dietary versus maternal sources of organochlorines in top predator seabird chicks: An experimental approach. Environ. Sci. Technol. 47, 5963–5970. doi:10.1021/es400442q

Bustnes, J.O., Bårdsen, B.J., Herzke, D., Johnsen, T. V, Eulaers, I., Ballesteros, M., Hanssen, S. a, Covaci, A., Jaspers, V.L.B., Eens, M., Sonne, C., Halley, D., Moum, T., Nøst, T.H., Erikstad, K.E., Ims, R.A., 2013. Plasma concentrations of organohalogenated pollutants in predatory bird nestlings: associations to growth rate and dietary tracers. Environ. Toxicol. Chem. 32, 2520–2527. doi:10.1002/etc.2329

Bustnes, J.O., Bangjord, G., Ahrens, L., Herzke, D., Yoccoz, N.G., 2015. Perfluoroalkyl substance (PFAS) concentrations in a terrestrial raptor: Relationships to environmental conditions and individual traits. Environ. Toxicol. Chem. 34, 184–191. doi:10.1002/etc.2782

Cassone, C.G., Vongphachan, V., Chiu, S., Williams, K.L., Letcher, R.J., Pelletier, E., Crump, D., Kennedy, S.W., 2012. In ovo effects of perfluorohexane sulfonate and perfluorohexanoate on pipping success, development, mRNA expression, and thyroid hormone levels in chicken embryos. Toxicol. Sci. 127, 216–224. doi:10.1093/toxsci/kfs072

Dauwe, T., Jaspers, V.L.B., Covaci, A., Eens, M., 2006. Accumulation of organochlorines and brominated flame retardants in the eggs and nestlings of great tits, *Parus major*. Environ. Sci. Technol. 40, 5297–5303. doi:10.1021/es060747a

Elliott, J.E., Brogan, J., Lee, S.L., Drouillard, K.G., Elliott, K.H., 2015. PBDEs and other

490 POPs in urban birds of prey partly explained by trophic level and carbon source. Sci.
491 Total Environ. 524–525, 157–165. doi:10.1016/j.scitotenv.2015.04.008

492 Eulaers, I., Covaci, A., Hofman, J., Nygård, T., Halley, D.J., Pinxten, R., Eens, M., Jaspers,
493 V.L.B., 2011a. A comparison of non-destructive sampling strategies to assess the
494 exposure of white-tailed eagle nestlings (*Haliaeetus albicilla*) to persistent organic
495 pollutants. Sci. Total Environ. 410–411, 258–265. doi:10.1016/j.scitotenv.2011.09.070

496 Eulaers, I., Covaci, A., Herzke, D., Eens, M., Sonne, C., Moum, T., Schnug, L., Hanssen,
497 S.A., Johnsen, T.V., Bustnes, J.O., Jaspers, V.L.B., 2011b. A first evaluation of the
498 usefulness of feathers of nestling predatory birds for non-destructive biomonitoring of
499 persistent organic pollutants. Environ. Int. 37, 622–30. doi:10.1016/j.envint.2010.12.007

500 Eulaers, I., Jaspers, V.L.B., Bustnes, J.O., Covaci, A., Johnsen, T. V., Halley, D.J., Moum,
501 T., Ims, R.A., Hanssen, S.A., Erikstad, K.E., Herzke, D., Sonne, C., Ballesteros, M.,
502 Pinxten, R., Eens, M., 2013. Ecological and spatial factors drive intra- and interspecific
503 variation in exposure of subarctic predatory bird nestlings to persistent organic
504 pollutants. Environ. Int. 57–58, 25–33. doi:10.1016/j.envint.2013.03.009

505 Eulaers, I., Jaspers, V.L.B., Halley, D.J., Lepoint, G., Nygård, T., Pinxten, R., Covaci, A.,
506 Eens, M., 2014. Brominated and phosphorus flame retardants in White-tailed Eagle
507 *Haliaeetus albicilla* nestlings: bioaccumulation and associations with dietary proxies
508 (δ^{13}C, δ^{15}N and δ^{34}S). Sci. Total Environ. 478, 48–57.
509 doi:10.1016/j.scitotenv.2014.01.051

510 Faxneld, S., Berger, U., Helander, B., Danielsson, S., Miller, A., Nyberg, E., Persson, J.-O.,
511 Bignert, A., 2016. Temporal Trends and Geographical Differences of Perfluoroalkyl
512 Acids in Baltic Sea Herring and White-Tailed Sea Eagle Eggs in Sweden. Environ. Sci.

513 Technol. 50, 13070–13079. doi:10.1021/acs.est.6b03230

514 Fry, B., 2006. Stable isotope ecology, Stable Isotope Ecology. Springer Science, USA.
515 doi:10.1007-0-387-33745-8

516 Gebbink, W.A., Letcher, R.J., Hebert, C.E., Chip Weseloh, D. V., 2011. Twenty years of
517 temporal change in perfluoroalkyl sulfonate and carboxylate contaminants in herring
518 gull eggs from the Laurentian Great Lakes. J. Environ. Monit. 13, 3365–3372.
519 doi:10.1039/c1em10663e

520 Gómez-Ramírez, P., Bustnes, J.O., Eulaers, I., Herzke, D., Johnsen, T.V., Lepoint, G., Pérez-
521 García, J.M., García-Fernández, A.J., Jaspers, V.L.B., 2017. Per- and polyfluoroalkyl
522 substances in plasma and feathers of nestling birds of prey from northern Norway.
523 Environ. Res. 158, 277–285. doi:10.1016/j.envres.2017.06.019

524 Guo, J., Simon, K., Romanak, K., Bowerman, W., Venier, M., 2018. Accumulation of flame
525 retardants in paired eggs and plasma of bald eagles. Environ. Pollut. 237, 499–507.
526 doi:10.1016/j.envpol.2018.02.056

527 Haukås, M., Berger, U., Hop, H., Gulliksen, B., Gabrielsen, G.W., 2007. Bioaccumulation of
528 per- and polyfluorinated alkyl substances (PFAS) in selected species from the Barents
529 Sea food web. Environ. Pollut. 148, 360–371. doi:10.1016/j.envpol.2006.09.021

530 Helander, B., Bignert, A., Asplund, L., 2008. Using raptors as environmental sentinels:
531 monitoring the white-tailed sea eagle *Haliaeetus albicilla* in Sweden. Ambio A J. Hum.
532 Environ. 37, 425–431. doi:10.1579/0044-7447(2008)37

533 Helander, B., Hailer, F., Vilà, C., 2007. Morphological and genetic sex identification of
534 white-tailed eagle *Haliaeetus albicilla* nestlings. J. Ornithol. 148, 435–442.
535 doi:10.1007/s10336-007-0156-y

536 Helgason, L.B., Barrett, R., Lie, E., Polder, A., Skaare, J.U., Gabrielsen, G.W., 2008. Levels
537 and temporal trends (1983-2003) of persistent organic pollutants (POPs) and mercury
538 (Hg) in seabird eggs from Northern Norway. Environ. Pollut. 155, 190–198.
539 doi:10.1016/j.envpol.2007.10.022

540 Herzke, D., Nygård, T., Berger, U., Huber, S., Røv, N., 2009. Perfluorinated and other
541 persistent halogenated organic compounds in European shag (*Phalacrocorax aristotelis*)
542 and common eider (*Somateria mollissima*) from Norway: a suburban to remote pollutant
543 gradient. Sci. Total Environ. 408, 340–348. doi:10.1016/j.scitotenv.2009.08.048

544 Houde, M., De Silva, A.O., Muir, D.C.G., Letcher, R.J., Canada, E., Saint-laurent, C., Street,
545 M., 2011. Monitoring of Perfluorinated Compounds in Aquatic Biota : An Updated
546 Review. Environ. Sci. Technol. 45, 7962–7973. doi: 10.1021/es104326w

547 Hung, H., Katsoyiannis, A.A., Brorström-Lundén, E., Olafsdottir, K., Aas, W., Breivik, K.,
548 Bohlin-Nizzetto, P., Sigurdsson, A., Hakola, H., Bossi, R., Skov, H., Sverko, E., Barresi,
549 E., Fellin, P., Wilson, S., 2016. Temporal trends of Persistent Organic Pollutants (POPs)
550 in arctic air: 20 years of monitoring under the Arctic Monitoring and Assessment
551 Programme (AMAP). Environ. Pollut. 217, 52–61. doi:10.1016/j.envpol.2016.01.079

552 Inger, R., Bearhop, S., 2008. Applications of stable isotope analyses to avian ecology. Ibis
553 (Lond. 1859). 447–461. doi:10.1111/j.1474-919X.2008.00839.x

554 Jenssen, B.M., Aarnes, J.B., Murvoll, K.M., Herzke, D., Nygård, T., 2010. Fluctuating wing
555 asymmetry and hepatic concentrations of persistent organic pollutants are associated in
556 European shag (*Phalacrocorax aristotelis*) chicks. Sci. Total Environ. 408, 578–585.
557 doi:10.1016/j.scitotenv.2009.10.036

558 Kelly, J.F., 2000. Stable isotopes of carbon and nitrogen in the study of avian and mammalian

559 trophic ecology. Can. J. Zool. 78, 1–27. doi:10.1139/z99-165

560 Koivusaari, J., Nuuja, I., Palokangas, R., Hattula, M.-L., 1976. Chlorinated Hydrocarbons
561 and Total Mercury in the Prey of the White-tailed Eagle (*Haliaeetus albicilla L* .) in the
562 Quarken Straits of the Gulf of Bothnia , Finland. Bull. Environ. Contam. Toxicol. 15,
563 235–241.

564 Lau, C., Anitole, K., Hodes, C., Lai, D., Pfahles-Hutchens, A., Seed, J., 2007. Perfluoroalkyl
565 acids: a review of monitoring and toxicological findings. Toxicol. Sci. 99, 366–94.
566 doi:10.1093/toxsci/kfm128

567 Leat, E.H.K., Bourgeon, S., Borgå, K., Strøm, H., Hanssen, S.A., Gabrielsen, G.W., Petersen,
568 Æ., Olafsdottir, K., Magnusdottir, E., Fisk, A.T., Ellis, S., Bustnes, J.O., Furness, R.W.,
569 2011. Effects of environmental exposure and diet on levels of persistent organic
570 pollutants (POPs) in eggs of a top predator in the North Atlantic in 1980 and 2008.
571 Environ. Pollut. 159, 1222–1228. doi:10.1016/j.envpol.2011.01.036

572 Leat, E.H.K., Bourgeon, S., Eze, J.I., Muir, D.C.G., Williamson, M., Bustnes, J.O., Furness,
573 R.W., Borgå, K., 2013. Perfluoroalkyl substances in eggs and plasma of an avian top
574 predator, great skua (*Stercorarius skua*), in the north Atlantic. Environ. Toxicol. Chem.
575 32, 569–576. doi:10.1002/etc.2101

576 Løseth, M.E., Briels, N., Flo, J., Malarvannan, G., Poma, G., Covaci, A., Herzke, D., Nygård,
577 T., Bustnes, J.O., Jenssen, B.M., Jaspers, V.L.B., 2019. White-tailed eagle (*Haliaeetus*
578 *albicilla*) feathers from Norway are suitable for monitoring of legacy, but not emerging
579 contaminants. Sci. Total Environ. 647, 525–533. doi:10.1016/j.scitotenv.2018.07.333

580 Mackay, D., Shiu, W.Y., Ma, K., Lee, S.C., 2006. Halogenated Hydrocarbons, Second edi.
581 ed, Handbook of Physical-Chemical Properties and Environmental Fate for Organic

582 Chemicals, Second Edition. CRC Press, Taylor & Francis Group.

583 Miller, A., Elliott, J.E., Elliott, K.H., Lee, S., Cyr, F., 2015. Temporal trends of
584 perfluoroalkyl substances (PFAS) in eggs of coastal and offshore birds: Increasing
585 PFAS levels associated with offshore bird species breeding on the Pacific coast of
586 Canada and wintering near Asia. Environ. Toxicol. Chem. 34, 1799–1808.
587 doi:10.1002/etc.2992

588 Muir, D.C.G., de Wit, C.A., 2010. Trends of legacy and new persistent organic pollutants in
589 the circumpolar arctic: Overview, conclusions, and recommendations. Sci. Total
590 Environ. 408, 3044–3051. doi:10.1016/j.scitotenv.2009.11.032

591 Murvoll, K.M., Skaare, J.U., Anderssen, E., Jenssen, B.M., 2006. Exposure and effects of
592 persistent organic pollutants in European shag (*Phalacrocorax aristotelis*) hatchlings
593 from the coast of Norway. Environ. Toxicol. Chem. 25, 190–198. doi:10.1897/04-
594 333R.1

595 Nakagawa, S., Schielzeth, H., 2013. A general and simple method for obtaining R2 from
596 generalized linear mixed-effects models. Methods Ecol. Evol. 4, 133–142.
597 doi:10.1111/j.2041-210x.2012.00261.x

598 Nordlöf, U., Helander, B., Bignert, A., Asplund, L., 2010. Levels of brominated flame
599 retardants and methoxylated polybrominated diphenyl ethers in eggs of white-tailed sea
600 eagles breeding in different regions of Sweden. Sci. Total Environ. 409, 238–246.
601 doi:10.1016/j.scitotenv.2010.09.042

602 Nøst, T.H., Helgason, L.B., Harju, M., Heimstad, E.S., Gabrielsen, G.W., Jenssen, B.M.,
603 2012. Halogenated organic contaminants and their correlations with circulating thyroid
604 hormones in developing Arctic seabirds. Sci. Total Environ. 414, 248–56.

605 doi:10.1016/j.scitotenv.2011.11.051

606 Nygård, T., Polder, A., 2012. Miljøgifter i rovfuglegg i Norge. Tilstand og tidstrender,
607 Norwegian Institute for Nature research (NINA), report 834. ISBN: 978-82-426-2429-1

608 Olsson, A., Ceder, K., Bergman, Å., Helander, B., 2000. Nestling blood of the white-tailed
609 sea eagle (*Haliaeetus albicilla*) as an indicator of territorial exposure to organohalogen
610 compounds - An evaluation. Environ. Sci. Technol. 34, 2733–2740.
611 doi:10.1021/es991426k

612 Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D., Heisterkamp, S., Van Willigen, B., 2018.
613 nlme: Linear and Nonlinear Mixed Effects Models. R Package version 3.1-131.

614 Post, D.M., Layman, C.A., Arrington, D.A., Takimoto, G., Quattrochi, J., Montaña, C.G.,
615 2007. Getting to the fat of the matter: Models, methods and assumptions for dealing with
616 lipids in stable isotope analyses. Oecologia 152, 179–189. doi:10.1007/s00442-006-
617 0630-x

618 Powley, C.R., George, S.W., Ryan, T.W., Buck, R.C., 2005. Matrix effect-free amalytical
619 methods for determination of perflurinated carboxylic acids in environmental matrixes.
620 Anal. Chem. 77, 6353–6358. doi:10.1021/ac0508090

621 R Development Core Team, 2008. R: A language and environment for statistical computing.

622 Route, W.T., Key, R.L., Russell, R.E., Lindstrom, A.B., Strynar, M.J., 2014. Spatial and
623 temporal patterns in concentrations of perfluorinated compounds in bald eagle nestlings
624 in the upper Midwestern United States. Environ. Sci. Technol. 48, 6653–6660.
625 doi:10.1021/es501055d

626 Sletten, S., Bourgeon, S., Bårdsen, B.-J., Herzke, D., Criscuolo, F., Massemin, S., Zahn, S.,

627 Johnsen, T.V., Bustnes, J.O., 2016. Organohalogenated contaminants in white-tailed
628 eagle (*Haliaeetus albicilla*) nestlings: An assessment of relationships to immunoglobulin
629 levels, telomeres and oxidative stress. Sci. Total Environ. 539, 337–349.
630 doi:10.1016/j.scitotenv.2015.08.123

631 UNEP (United Nations Environmental Programme), 2009. The Stockholm Convention on
632 Persistent Organic Pollutants. United Nations, adoption of amendments to Annexes A, B
633 and C. Stock. Conv. URL:
634 http://chm.pops.int/Convention/ConventionText/tabid/2232/Default.aspx

635 Vicente, J., Sanpera, C., García-Tarrasón, M., Pérez, A., Lacorte, S., 2015. Perfluoroalkyl and
636 polyfluoroalkyl substances in entire clutches of Audouin's gulls from the ebro delta.
637 Chemosphere 119, S62–S68. doi:10.1016/j.chemosphere.2014.04.041

638 Voorspoels, S., Covaci, A., Maervoet, J., Schepens, P., 2002. Relationship between age and
639 levels of organochlorine contaminants in human serum of a Belgian population. Bull.
640 Environ. Contam. Toxicol. 69, 22–29. doi:10.1007/s00128-002-0004-y

641 Wang, Z., Cousins, I.T., Scheringer, M., Buck, R.C., Hungerbühler, K., 2014. Global
642 emission inventories for C4-C14perfluoroalkyl carboxylic acid (PFCA) homologues
643 from 1951 to 2030, Part I: Production and emissions from quantifiable sources. Environ.
644 Int. 70, 62–75. doi:10.1016/j.envint.2014.04.013

645 Wierda, M.R., Leith, K.F., Roe, A.S., Grubb, T.G., Sikarskie, J.G., Best, D.A., Pittman, H.T.,
646 Fuentes, L., Simon, K.L., Bowerman, W., 2016. Using bald eagles to track spatial
647 (1999–2008) and temporal (1987–1992, 1999–2003, and 2004–2008) trends of
648 contaminants in Michigan's aquatic ecosystems. Environ. Toxicol. Chem. 35, 1995–
649 2002. doi:10.1002/etc.3523

650 Willgohs, J.F., 1984. Havørn i Norge: Næring, forplantningsøkologi, konkurrenter og fiender.
651 Direktoratet for vilt og ferskvannsfisk, Trondheim. ISSN:032-5008

652 Zuur, A.F., Ieno, E.N., Elphick, C.S., 2010. A protocol for data exploration to avoid common
653 statistical problems. Methods Ecol. Evol. 1, 3–14. doi:10.1111/j.2041-
654 210X.2009.00001.x

655 Zuur, A.F., Ieno, E.N., Walker, N.J., Saveliev, A.A., Smith, G.M., 2009. Mixed Effects
656 Models and Extensions in Ecology with R, Springer.
657 doi:10.1017/CBO9781107415324.00

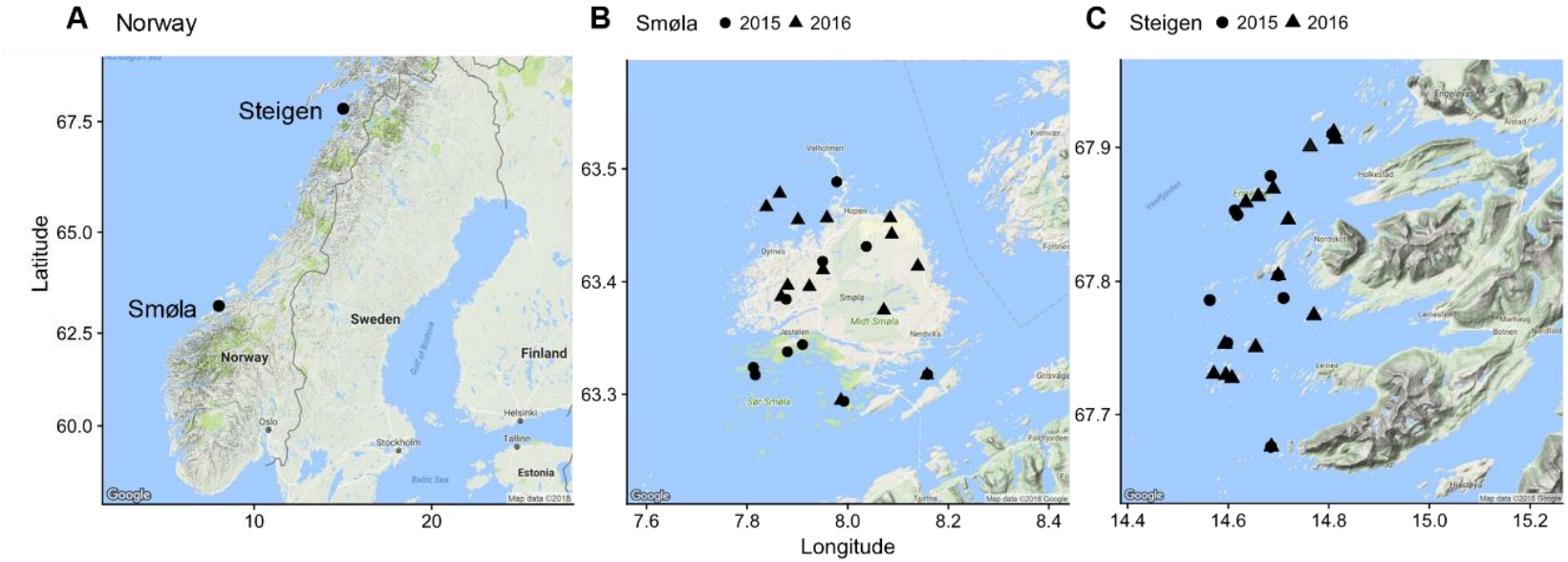
658

List of figures

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

671 **Figure 1**

672



673

Figure 2

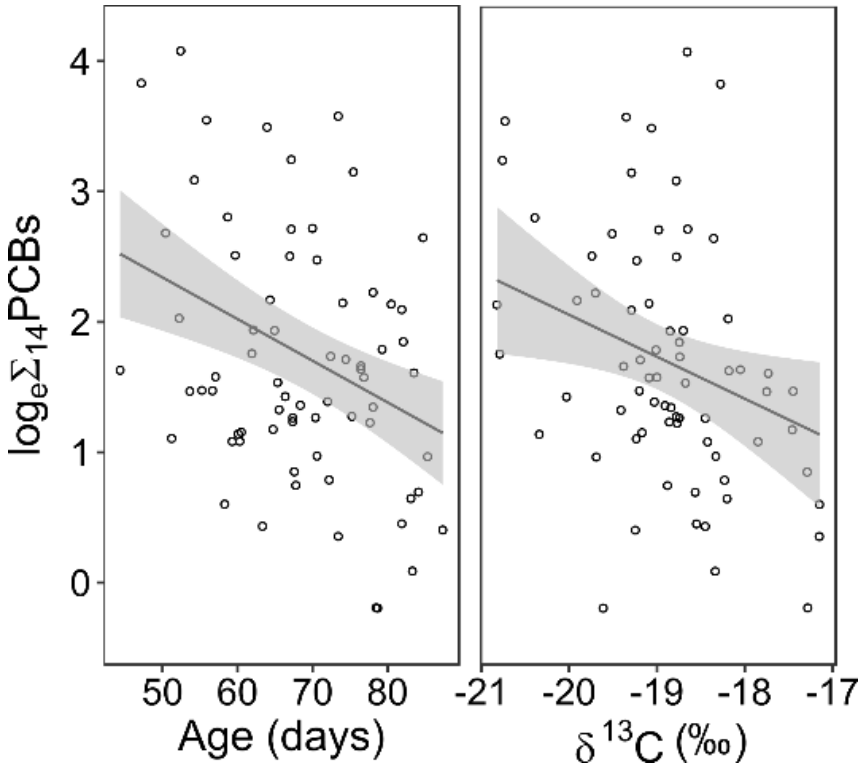


Figure 3

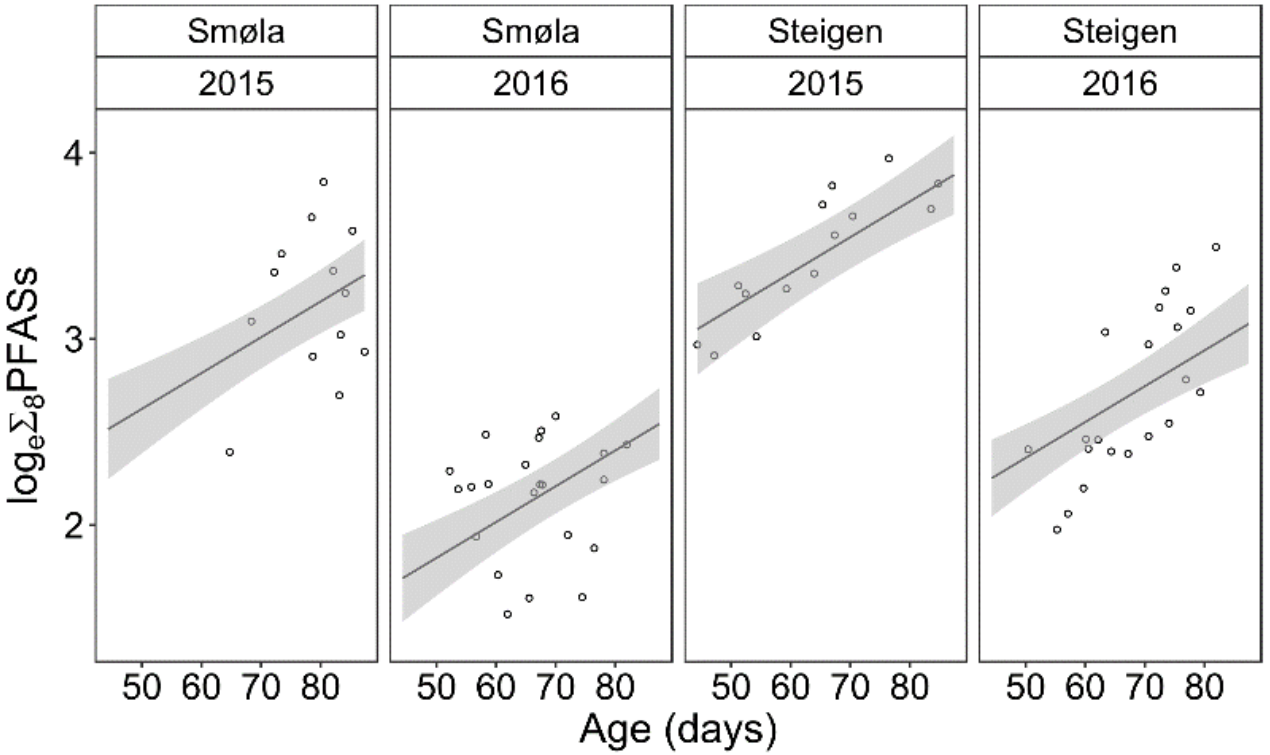


Table 1: Median, min and max values of stable isotopes from body feathers, age and sum of PCBs, OCPs, PBDEs and PFASs detected in plasma of white-tailed eagle nestlings sampled in Smøla and Steigen (Norway) in 2015 and 2016. A full list of concentration data for the individual compounds can be found in Løseth et al. (2019).

		Smøla						**Steigen**					
		2015 *n* = 13			2016 *n* = 22			2015 *n* = 14			2016 *n* = 21		
	unit	median	min	max	median	Min	max	median	min	max	median	min	max
$\delta^{13}C$	‰	-18.56	-20.82	-17.15	-19.02	-20.79	-17.15	-18.66	-19.24	-17.73	-19.11	-20.34	-18.33
$\delta^{15}N$	‰	+13.82	+12.45	+15.07	+13.39	+11.54	+15.28	+14.54	+13.89	+15.17	+13.96	+13.43	+14.73
Age	days	80.51	64.75	87.37	66.77	52.22	81.92	64.65	44.34	84.75	70.61	50.40	81.92
Σ_{14}PCBs[a]	ng/mL	2.00	0.82	8.47	4.86	1.86	34.52	5.12	2.95	59.05	5.79	1.58	35.92
Σ_{7}OCPs[b]	ng/mL	2.01	0.89	6.28	2.75	1.05	15.33	4.75	2.80	52.19	5.79	1.31	12.96
Σ_{5}PBDEs[c]	ng/mL	0.10	0.06	0.46	0.16	0.05	1.51	0.34	0.10	2.64	0.23	0.03	0.73
Σ_{8}PFASs[d]	ng/mL	25.69	10.29	46.65	9.18	4.58	13.26	31.80	18.36	52.94	12.76	7.21	32.90

[a] Σ_{14}PCBs: CB 99, 101, 105, 118, 138, 153, 156, 170, 171, 177, 180, 183, 187 and 194

[b] Σ_{7}OCPs: OxC, TN, CN, *p,p'*-DDE, *p,p'*-DDT, HCB and β-HCH

[c] Σ_{5}PBDEs: BDE 47, 99, 100, 153 and 154

[d] Σ_{8}PFASs: Br-PFOS, Lin-PFOS, PFOA, PFNA, PFDcA, PFUnA, PFDoA and PFTriA

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

Table 2: Model estimates from the most parsimonious models (ΔAICc < 2) explaining the variation of Σ_{14}PCBs, Σ_7OCPs, Σ_5PBDEs and Σ_8PFASs in plasma of white-tailed eagle nestlings (n = 70) from Smøla and Steigen. The table includes the model intercept (β_0), model estimates (β_x), significance values (p), and marginal pseudo-R^2 (R_m^2) and conditional pseudo-R^2 (R_c^2). The year variable (Yr) represents 2016 and location variable (Loc) represents Steigen. Beta estimates follow the order of the factors in the models. Statistical significance (α = 0.05) is marked with *.

Compound group	Explanatory variables	β_0	β_1	β_2	β_3	β_4	β_5	*p-values*	ΔAICc	R_m^2	R_c^2
Σ_{14}PCBs	~ age + δ^{13}C + Loc	-3.07	-0.03	-0.36	0.43			<0.01*; 0.01*; 0.08	0.00	0.28	0.89
	~ age + δ^{13}C	-2.61	-0.03	-0.35				<0.01*; 0.01*	0.81	0.22	0.89
	~ age + δ^{13}C + Loc + Yr + Loc:Yr	-3.66	-0.03	-0.35	1.03	0.57	-0.95	0.01*; 0.02*; 0.01*; 0.12; 0.06	1.03	0.34	0.89
Σ_7OCPs[a]	~ age + δ^{13}C + Loc + Yr + Loc:Yr	-5.00	-0.01	-0.36	0.91	0.13	-0.80	0.07; <0.01*; <0.01*; 0.62: 0.03*	0.00	0.37	0.91
	~ δ^{13}C + Loc + Yr + Loc:Yr	-5.71	-0.35	1.07	0.28	-0.98		<0.01*; <0.01*; 0.23; <0.01*	0.15	0.37	0.88
Σ_5PBDEs	~ age + δ^{13}C	-6.71	-0.03	-0.38				<0.01*; <0.01*	0.00	0.22	0.86
	~ age + δ^{13}C + Loc + Yr + Loc:Yr	-8.39	-0.02	-0.43	0.87	0.14	-0.86	0.02*; <0.01*; 0.03*; 0.70; 0.08	0.46	0.32	0.86
	~ age + δ^{13}C + Loc	-7.07	-0.03	-0.38	0.31			<0.01*; <0.01*; 0.19	0.54	0.25	0.86
	~ age + δ^{13}C + Yr	-7.28	-0.03	-0.43	-0.31			<0.01*; <0.01*; 0.23	0.83	0.23	0.86
	~ age + δ^{13}C + Loc + Yr	-7.65	-0.03	-0.43	0.31	-0.31		<0.01*; <0.01*; 0.19; 0.22	1.34	0.27	0.86
Σ_8PFASs	~ age + Loc + Yr	1.66	0.02	0.54	-0.80			<0.01*; <0.01*; <0.01*	0.00	0.73	0.93

[a] Two outliers were removed from these models, n = 68.

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

Supporting information:

Plasma concentrations of organohalogenated contaminants in white-tailed eagle nestlings – the role of age and diet

Mari Engvig Løseth[a*], Nathalie Briels[a], Igor Eulaers[b], Torgeir Nygård[c], Govindan Malarvannan[d], Giulia Poma[d], Adrian Covaci[d], Dorte Herzke[e], Jan Ove Bustnes[f], Bjørn Munro Jenssen[a,b], Gilles Lepoint[g], Veerle L. B. Jaspers[a]

Affiliations:

[a]Department of Biology, Norwegian University of Science and Technology (NTNU), Høgskoleringen 5, 7491 Trondheim, Norway

[b]Department of Bioscience, Aarhus University, Fredriksborgvej 399, 4000 Roskilde, Denmark

[c]Norwegian Institute for Nature Research (NINA), Høgskoleringen 9, 7034 Trondheim, Norway

[d]Toxicological Centre, University of Antwerp, Universiteitsplein 1, 2610 Wilrijk, Belgium

[e]Norwegian Institute for Air Research (NILU), FRAM - High North Research Centre on Climate and the Environment, 9007 Tromsø, Norway

[f]Norwegian Institute for Nature Research (NINA), FRAM - High North Research Centre on Climate and the Environment, 9007 Tromsø, Norway

[g]Laboratoire d´Océanologie, University of Liège, 4000 Liège (Sart Tilman), Belgium

* corresponding author: mari.loseth@ntnu.no

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

Table S2: Overview of white-tailed eagle nestlings sampled in Smøla and Steigen (Norway), in 2015 and 2016.

	Smøla		Steigen	
	2015	**2016**	**2015**	**2016**
Number of nests	10	14	9	15
Number of nestlings	13	22	14	21
Number of females	7	9	8	10
Number of males	6	13	6	11

Table S3: Compounds and their concentrations in internal standards used for extraction of targeted PCBs, PBDEs, OCPs, NBFRs, PFRs, DPs and PFASs. Obtained from Løseth et al. (2019).

Internal standard	**Concentrations and compounds**
IS1 (POPs)	200 pg/µL PCB 143
	25 pg/µL BDE 77
	25 pg/µL ε-hexachlorocyclohexane (ε-HCH)
IS2 (ECs)	200 pg/µL ^{13}C-bis(2-ethylhexyl)-3,4,5,6-tetrabromophthalate (TBPH)
	50 pg/µL ^{13}C-syn-dechlorane plus (DP)
	50 pg/µL ^{13}C-anti-DP
	1 ng/µL triphenyl phosphate (TPHP-d15)
	1 ng/µL tris(chloroethyl) phosphate (TCEP-d12)
	1 ng/µL tris-(1,3-dichloro-2-propyl) phosphate (TDCIPP-d15)
	1 ng/µL triamyl phosphate (TAP)
	2 ng/µL tri-(2-butoxyethyl) phosphate (TBOEP-d6)
IS3 (DPs)	200 pg/µL ^{13}C-DPs
IS4 (PFASs)	0.1 ng/µL ^{13}C-PFAS mix

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

Table S4: Recoveries of internal standards in plasma samples from white tailed eagles from Smøla and Steigen (Norway), from 2015 and 2016. Compounds not analysed are marked with "na". Obtained from Løseth et al. (2019).

	2015		2016	
Plasma	Mean	sd	Mean	sd
CB 143	69	39	108	8
ε-HCH	53	12	79	11
BDE 77	83	9	87	10
^{13}C-HCB	na	na	102	17
^{13}C-TBPH	18	14	25	12
^{13}C-s-DP	47	13	55	10
^{13}C-a-DP	50	10	52	11
TAP	93	13	79	14
TCEP-d12	89	33	72	22
TBEP-d6	78	35	68	28
TPhP-d15	86	15	80	12
TDCPP-d15	92	15	80	11
^{13}C-PFPA	101	16	87	12
^{13}C-PFHxA	93	22	86	10
^{13}C-PFHpA	90	16	93	12
^{13}C-PFOA	88	15	87	12
^{13}C-PFDcA	81	32	89	11
^{13}C-PFUnA	77	13	84	12
^{13}C-PFDoA	59	15	83	19
^{13}C-PFHxS	90	17	87	12
^{13}C-PFOS	79	17	84	12
^{13}C-PFOSA	83	17	75	9

Table S5: Targeted chlorinated compounds analysed in plasma from white-tailed eagle nestlings sampled at Steigen and Smøla (Norway) in 2015 and 2016. Only samples from 2016 were analysed for *p,p'*-DDD. PCB congeners are numbered by the IUPAC system (International Union of Pure and Applied Chemistry). Limit of quantification (LOQ) for the compounds are the same for 2015 and 2016 and are presented as ng/mL. Obtained from Løseth et al. (2019).

Organochlorinated compounds

Group	**Abbreviations**	**Compounds**	LOQ plasma
Organo-	OxC	*oxy*-chlordane	0.01
Chlorinated	TN	*trans*-nonachlor	0.01
Pesticides	CN	*cis*-nonachlor	0.01
(OCPs)	HCB	hexachlorobenzene	0.01
	a-HCH	1α,2α,3β,4α,5β,6β-hexachlorocyclohexane	0.01
	b-HCH	1α,2β,3α,4β,5α,6β-hexachlorocyclohexane	0.01
	g-HCH	1α,2α,3β,4α,5α,6β-hexachlorocyclohexane	0.02
	p,p',-DDT	*p,p'*,-dichloro-α,α-diphenyl-β,β,β-trichloroethane	0.02
	p,p',-DDE	*p,p'*,-dichloro-diphenyl-dichloroethylene	0.02
2016	*p,p'*,-DDD	*p,p'*,-dichloro-diphenyl-dichloroethane	0.02
Polychlorinated	CB 28	2,4,4'-trichlorobiphenyl	0.05
Biphenyls	CB 49	2,2',4,5'-tetrachlorobiphenyl	0.05
(PCBs)	CB 52	2,2',5,5'-tetrachlorobiphenyl	0.05
	CB 74	2,4,4',5-tetrachlorobiphenyl	0.05
	CB 95	2,2',3,5',6-pentachlorobiphenyl	0.02
	CB 99	2,2',4,4',5-pentachlorobiphenyl	0.02
	CB 101	2,2',4,5,5'-pentachlorobiphenyl	0.02
	CB 105	2,3,3',4,4'-pentachlorobiphenyl	0.01
	CB 110	2,3,3',4',6-pentachlorobiphenyl	0.01
	CB 118	2,3',4,4',5-pentachlorobiphenyl	0.01
	CB 138	2,2',3,4,4',5'-hexachlorobiphenyl	0.01
	CB 149	2,2',3,4',5',6-hexachlorobiphenyl	0.01
	CB 153	2,2',4,4',5,5'-hexachlorobiphenyl	0.01
	CB 156	2,3,3',4,4',5-hexachlorobiphenyl	0.01
	CB 170	2,2',3,3',4,4',5-heptachlorobiphenyl	0.01
	CB 171	2,2',3,3',4,4',6-heptachlorobiphenyl	0.01
	CB 177	2,2',3,3',4,5',6'-heptachlorobiphenyl	0.01
	CB 180	2,2',3,4,4',5,5'-heptachlorobiphenyl	0.01
	CB 183	2,2',3,4,4',5',6-heptachlorobiphenyl	0.01
	CB 187	2,2',3,4',5,5',6-heptachlorobiphenyl	0.01
	CB 194	2,2',3,3',4,4',5,5'-octachlorobiphenyl	0.01
	CB 206	2,2',3,3',4,4',5,5',6-nonachlorobiphenyl	0.01
	CB 209	Decachlorobiphenyl	0.01

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

Table S6: Targeted flame retardants analysed in plasma from white-tailed eagle nestlings, sampled at Steigen and Smøla (Norway) in 2015 and 2016. PBDE congeners are numbered by the IUPAC system (International Union of Pure and Applied Chemistry). Only samples from 2016 were analyzed for 2'-MeO-BDE 68 and 6'-MeO-BDE 47. Limit of quantification (LOQ) for the compounds are the same for 2015 and 2016 and are presented as ng/mL. Compounds not targeted (analyzed) are marked with "n.a". Obtained from Løseth et al. (2019).

Flame retardants			
Group	**Abbreviations**	**Compounds**	LOQ plasma
Polybrominated diphenyl ethers (PBDEs)	BDE 28	2',4,4'-tribromodiphenyl ether	0.002
	BDE 47	2,2',4,4'-tetrabromodiphenyl ether	0.002
	BDE 99	2,2',4,4',5'-pentabromodiphenyl ether	0.002
	BDE 100	2,2',4,4',6'-pentabromodiphenyl ether	0.002
	BDE 153	2,2',4,4',5,5'-hexabromobiphenyl ether	0.002
	BDE 154	2,2',4,4',5,6'-hexabromobiphenyl ether	0.004
	BDE 183	2,2',3',4,4',5',6'-heptabromodiphenyl ether	0.004
2016	2'-MeO-BDE 68	1,5-Dibromo-3-(2,4-dibromophenoxy)-2-methoxybenzene	0.004
2016	6'-MeO-BDE 47	1,5-Dibromo-2-(2,4-dibromophenoxy)-3-methoxybenzene	0.004

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

Table S7: Targeted compounds for per-and polyfluorinated substance analysed in plasma from white-tailed eagle nestlings, sampled at Steigen and Smøla (Norway) in 2015 and 2016. Limit of quantification (LOQ) is given as ng/mL. Results are obtained from Løseth et al. (2019). Compounds not detected are marked n.d.

Per- and polyfluorinated substances (PFASs)				
	Abbreviations	**Compounds**	LOQ plasma 2015	LOQ plasma 2016
Carboxylic acids	PFBA	Perfluorobutanoic acid	169.11	0.05
	PFPeA	Perfluoropentanoic acid	1.31	0.05
	PFHxA	Perfluorohexanoic acid	1.31	0.10
	PFHpA	Perfluoroheptanoic acid	1.31	0.05
	PFOA	Perfluorooctanoic acid	0.10	0.05
	PFNA	Perfluorononanoic acid	0.10	0.08
	PFDcA	Perfluorodecanoanoic acid	0.10	0.05
	PFUnA	Perfluoroundecanoic acid	0.20	0.08
	PFDoA	Perfluorododecanoic acid	0.20	0.08
	PFTrA	Perfluorotridecanoic acid	0.20	0.10
	PFTeA	Perfluorotetradecanoic acid	0.20	0.10
Sulfonamides	PFOSA	Perfluorooctanesulfonamide	23.35	0.10
Sulfonic acids	PFBS	Perfluorobutane sulfonate	169.11	0.05
	PFPS	Perfluoropentane sulfonate	n.d.	n.d.
	PFHxS	Perfluorohexane sulfonate	0.10	0.05
	PFHpS	Perfluoroheptane sulfonate	0.01	0.08
	Lin-PFOS	Linear perfluorooctane sulfonate	0.20	0.10
	Br-PFOS	Branched perfluorooctane sulfonate	0.20	0.10
	PFNS	Perfluorononane sulfonate	n.d.	0.10

Table S8: Median, min and max concentrations (ng/ml ww) of PCBs, OCPs, PBDEs and PFASs quantified in over 50 % of plasma samples from white tailed eagles from Smøla and Steigen (Norway), from 2015 and 2016. Results are obtained from Løseth et al. (2019).

	Smøla						**Steigen**					
	2015 *n* = 13			2016 *n* = 22			2015 *n* = 14			2016 *n* = 21		
	median	min	max	median	min	max	median	min	max	median	min	max
CB 99	0.16	0.08	0.59	0.18	0.06	1.47	0.5	0.18	4.61	0.23	0.06	0.98
CB 101	0.21	0.09	0.31	0.14	0.01	0.56	0.16	0.07	0.56	0.14	0.02	0.25
CB 105	0.08	0.04	0.3	0.11	0.04	0.79	0.26	0.1	2.59	0.14	0.04	0.66
CB 118	0.23	0.11	0.81	0.41	0.17	2.92	0.7	0.28	7.3	0.50	0.14	2.30
CB 138	0.27	0.11	1.25	1.1	0.4	10.55	0.66	0.29	5.63	1.26	0.28	8.88
CB 153	0.74	0.21	3.06	1.44	0.55	9.48	2.05	1.12	26.27	1.75	0.43	10.16
CB 156	0.02	0.01	0.11	0.06	0.02	0.43	0.07	0.03	0.8	0.07	0.02	0.54
CB 170	0.07	0.02	0.36	0.22	0.07	1.3	0.18	0.06	2.16	0.23	0.07	1.98
CB 171	0.02	0.01	0.07	0.04	0.01	0.23	0.03	0.02	0.37	0.04	0.01	0.26
CB 177	0.02	0.01	0.07	0.04	0.02	0.43	0.03	0.01	0.18	0.05	0.01	0.17
CB 180	0.17	0.04	0.84	0.7	0.2	3.55	0.45	0.13	5.29	0.65	0.19	5.89
CB 183	0.04	0.01	0.19	0.12	0.04	0.76	0.1	0.04	1.15	0.13	0.03	1.03
CB 187	0.11	0.03	0.43	0.32	0.13	2.55	0.22	0.07	1.81	0.36	0.1	1.95
CB 194	0.03	0.02	0.08	0.08	0.02	0.3	0.05	0.02	0.38	0.07	0.02	0.79
Σ_{14}PCBs	**2.00**	**0.82**	**8.47**	**4.86**	**1.86**	**34.52**	**5.12**	**2.95**	**59.05**	**5.79**	**1.58**	**35.92**
OxC	0.04	0.02	0.14	0.08	0.02	0.53	0.24	0.05	2.16	0.13	0.04	0.6
TN	0.19	0.06	0.36	0.26	0.08	0.98	0.22	0.1	1.22	0.29	0.14	0.59
CN	0.07	0.04	0.12	0.13	0.05	0.39	0.07	0.04	0.44	0.14	0.08	0.25
p,p'-DDE	1.21	0.56	5.23	1.2	0.56	9.47	3.95	2.18	47.61	1.45	0.48	8.64
p,p'-DDT	0.2	0.08	0.3	0.15	0.06	0.63	0.13	0.06	0.31	0.27	0.09	0.38
HCB	0.09	0.04	0.21	0.76	0.26	2.96	0.15	0.05	0.8	1.02	0.32	2.46
β-HCH	0.04	0.03	0.06	0.02	0.01	0.08	0.06	0.03	0.32	0.02	0.01	0.07
Σ_{7}OCPs	**2.01**	**0.89**	**6.28**	**2.75**	**1.05**	**15.33**	**4.75**	**2.80**	**52.19**	**5.79**	**1.31**	**12.96**
BDE 47	0.06	0.03	0.28	0.08	0.01	0.81	0.19	0.06	1.82	0.09	0.01	0.36
BDE 99	0.01	0.003	0.03	0.02	0.003	0.16	0.04	0.01	0.23	0.03	0.002	0.08
BDE 100	0.03	0.01	0.11	0.03	0.004	0.35	0.08	0.02	0.5	0.03	0.002	0.14
BDE 153	0.01	0.003	0.02	0.01	0.004	0.08	0.02	0.003	0.07	0.01	0.002	0.08
BDE 154	0.01	0.004	0.03	0.03	0.006	0.17	0.01	0.003	0.04	0.02	0.003	0.10
Σ_{5}PBDEs	**0.10**	**0.06**	**0.46**	**0.16**	**0.05**	**1.51**	**0.34**	**0.1**	**2.64**	**0.23**	**0.03**	**0.73**
Br-PFOS	2.23	0.55	4.20	0.65	0.29	1.49	5.38	1.85	11.70	2.12	0.72	6.83
Lin-PFOS	14.12	6.04	31.85	5.25	2.34	8.47	16.55	9.55	27.07	7.01	3.86	17.5
PFOA	0.35	0.12	0.57	0.12	0.03	0.29	0.53	0.14	1.27	0.40	0.11	0.95
PFNA	1.82	0.57	4.86	0.56	0.35	1.77	3.58	1.56	6.48	1.69	0.63	6.60
PFDcA	1.22	0.66	2.3	0.39	0.25	0.82	1.44	0.91	2.52	0.69	0.36	1.82
PFUnA	3.59	2.43	4.36	1.15	0.68	2.05	3.36	2.30	5.08	1.40	0.94	2.15
PFDoA	0.57	0.32	0.94	0.27	0.09	0.46	0.38	0.20	0.86	0.22	0.15	0.51
Σ_{8}PFASs	**25.69**	**10.29**	**46.65**	**9.18**	**4.58**	**13.26**	**31.80**	**18.36**	**52.94**	**12.76**	**7.21**	**32.90**

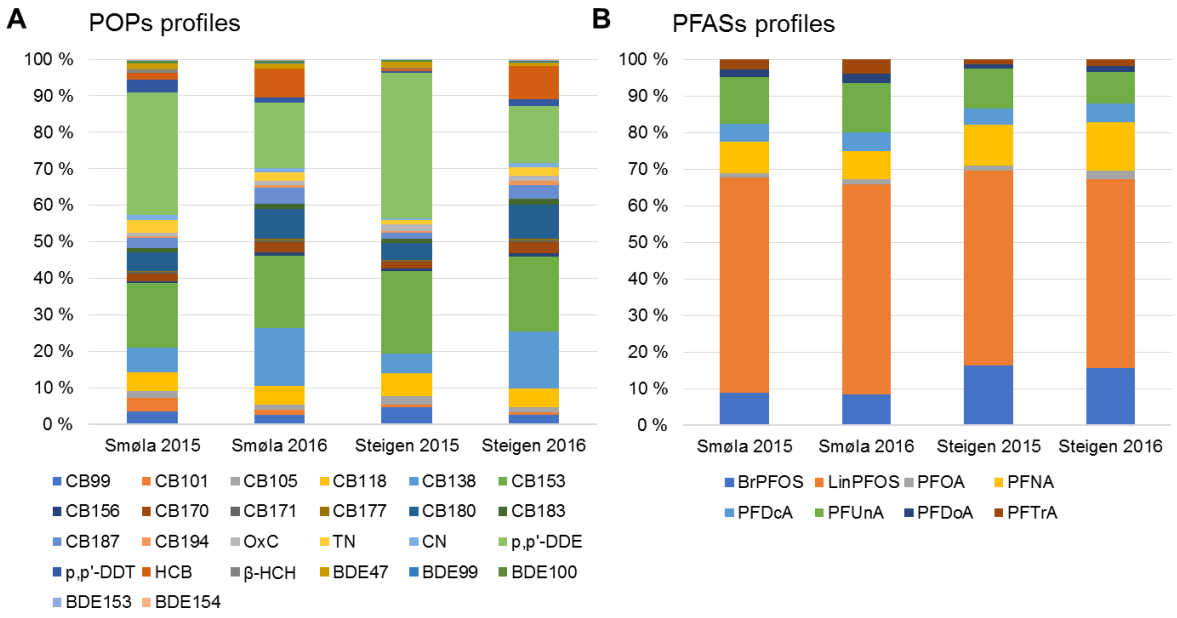


Figure S3: A) PCBs, OCPs and PBDEs and B) PFAS profiles in plasma of white-tailed eagle nestlings sampled in Smøla and Steigen (Norway), in 2015 and 2016.

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

Table S9: Correlations between PCBs, OCPs and PBDEs and sum of each compound group where significant correlations (α = 0.05, Bonferroni corrected) are marked with blue colours for positive and red colours for negative correlations.

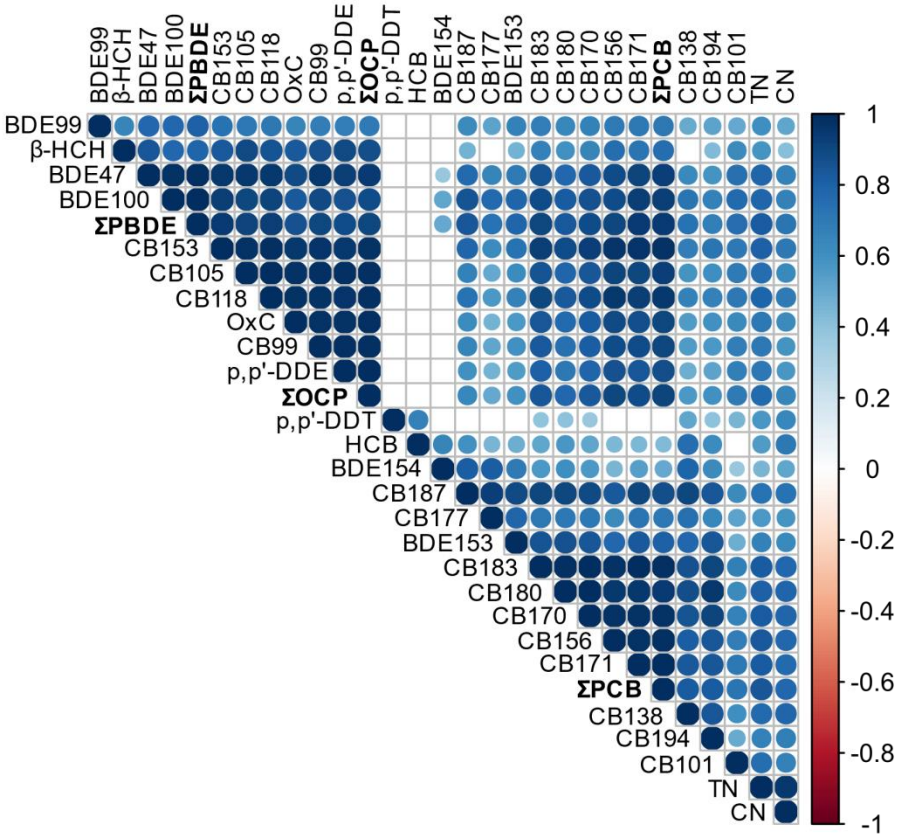
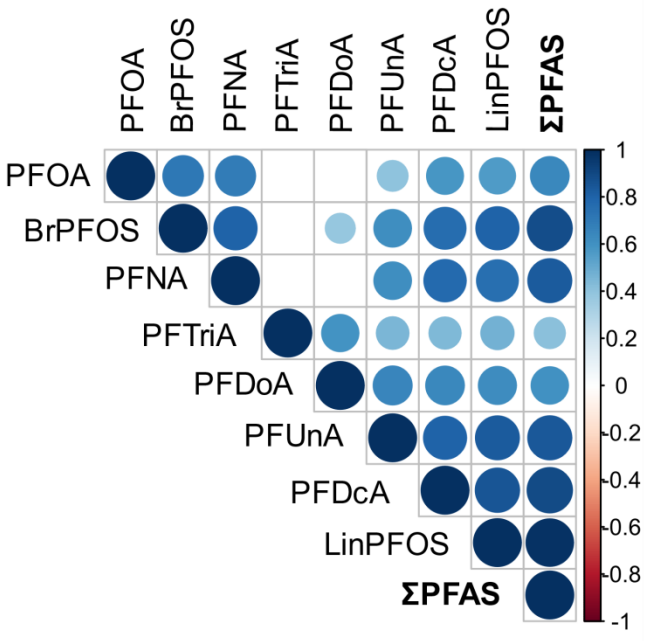


Table S10: Correlations between PFASs where significant correlations (α = 0.05, Bonferroni corrected) are marked with blue colours for positive and red colours for negative correlations.



This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

Body condition index

A body condition index was estimated using a standard major axis (SMA) regression to predict body mass based on wing length (Peig and Green, 2009). This regression was performed on each sex separately due to sexual dimorphism (Helander et al., 2007), and the correlation between body mass and wing length was strong and significant for both sexes (F: r_{35}= 0.70, p <0.01, M: r_{35} = 0.42, p = 0.01). We then obtained an index of body condition for each individual by subtracting the mass predicted by the SMA regression from the observed weight. The difference between the actual weight of the individual and the predicted weight was considered the body condition index (i.e. a positive value indicates a heavier weight than would be expected for a given size of the individual). For nestlings from Smøla the BCI ranged from -0.39 – 1.42 in 2015 and from -1.30 – 0.79 in 2016, while for nestlings from Steigen BCIs ranged from -0.55 – 0.18 in 2015 and -0.33 – 0.71 in 2016.

To avoid problems with multicollinearity, we chose to include age in the models, and not body mass or BCI, as we detected significant correlations between BCI and body mass (r_{70} = 0.64, p < 0.01), between BCI and age (r_{70} = 0.42, p < 0.01) and between age and body mass (r_{70} = 0.32, p < 0.01).

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

Table S11: Results from lme-Anova to investigate spatial and temporal variation of the sums of each OHC group, stable isotopes (δ^{15}N and δ^{13}C) and age. Significance level was set to $\alpha = 0.05$ and significant differences are marked with *.

Σ_{14}PCBs	**$F_{(1,44)}$-value**	***p*-value**	**Σ_{7}OCPs**	**$F_{(1,44)}$-value**	***p*-value**
Intercept	215.08	< 0.01	Intercept	183.90	< 0.01
Location	5.78	0.02*	Location	7.61	0.01*
Year	2.56	0.12	Year	0.41	0.52
Location:Year	7.58	< 0.01*	Location:Year	8.25	< 0.01*
Σ_{5}PBDEs			**Σ_{8}PFASs**		
Intercept	154.54	< 0.01	Intercept	2807.97	< 0.01
Location	3.27	0.08	Location	16.22	< 0.01*
Year	0.01	0.91	Year	67.89	< 0.01*
Location:Year	5.91	0.02*	Location:Year	1.19	0.28
δ^{15}N			**δ^{13}C**		
Intercept	128644.80	< 0.01	Intercept	26144.80	< 0.01
Location	15.71	< 0.01*	Location	0.07	0.80
Year	8.80	< 0.01*	Year	4.89	0.03*
Location:Year	0.06	0.80	Location:Year	0.11	0.74
Age					
Intercept	3030.87	< 0.01			
Location	3.86	0.06			
Year	2.64	0.11			
Location:Year	10.86	< 0.01*			

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

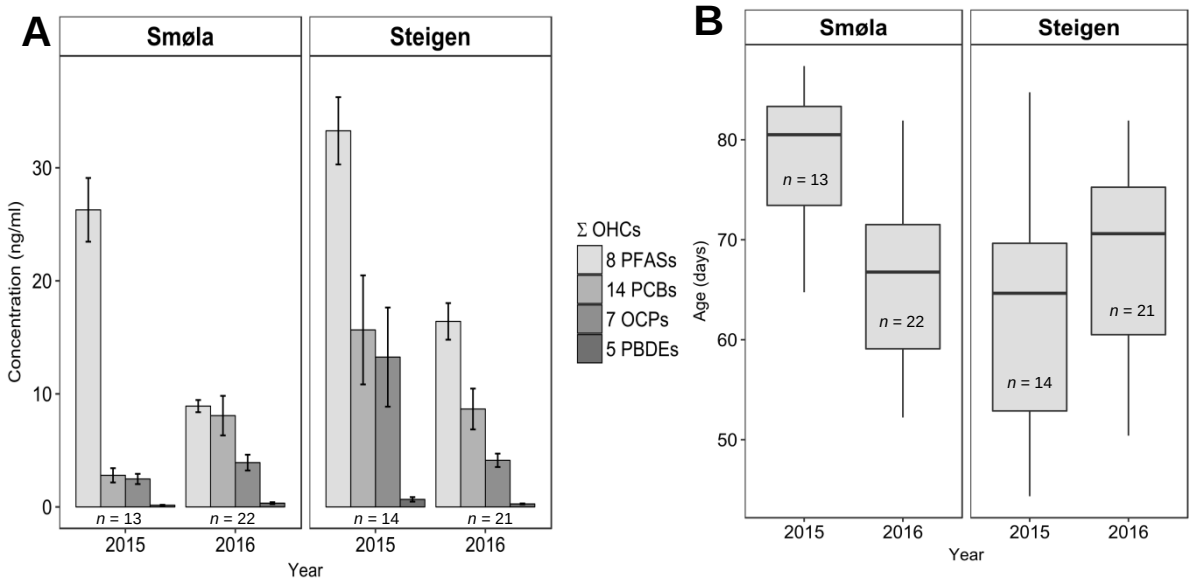


Figure S4:A) OHC concentrations (ng/ml) in plasma from white-tailed eagle nestlings sampled in Smøla and Steigen (Norway), in 2015 and 2016. B) Shows the age (days) distribution of the sampled nestlings.

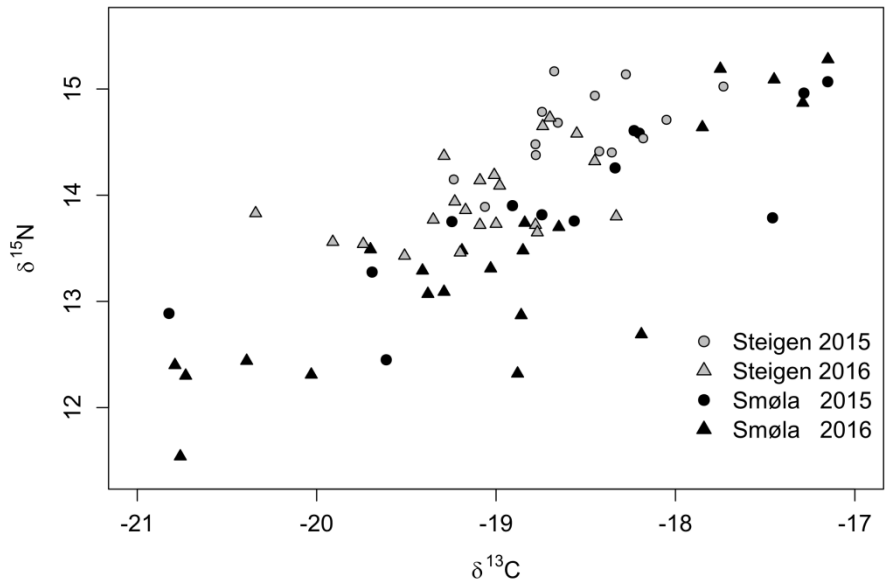


Figure S5: Stable isotope ratios of carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) in body feathers from white-tailed eagle nestlings, sampled in Smøla and Steigen (Norway), in 2015 and 2016.

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

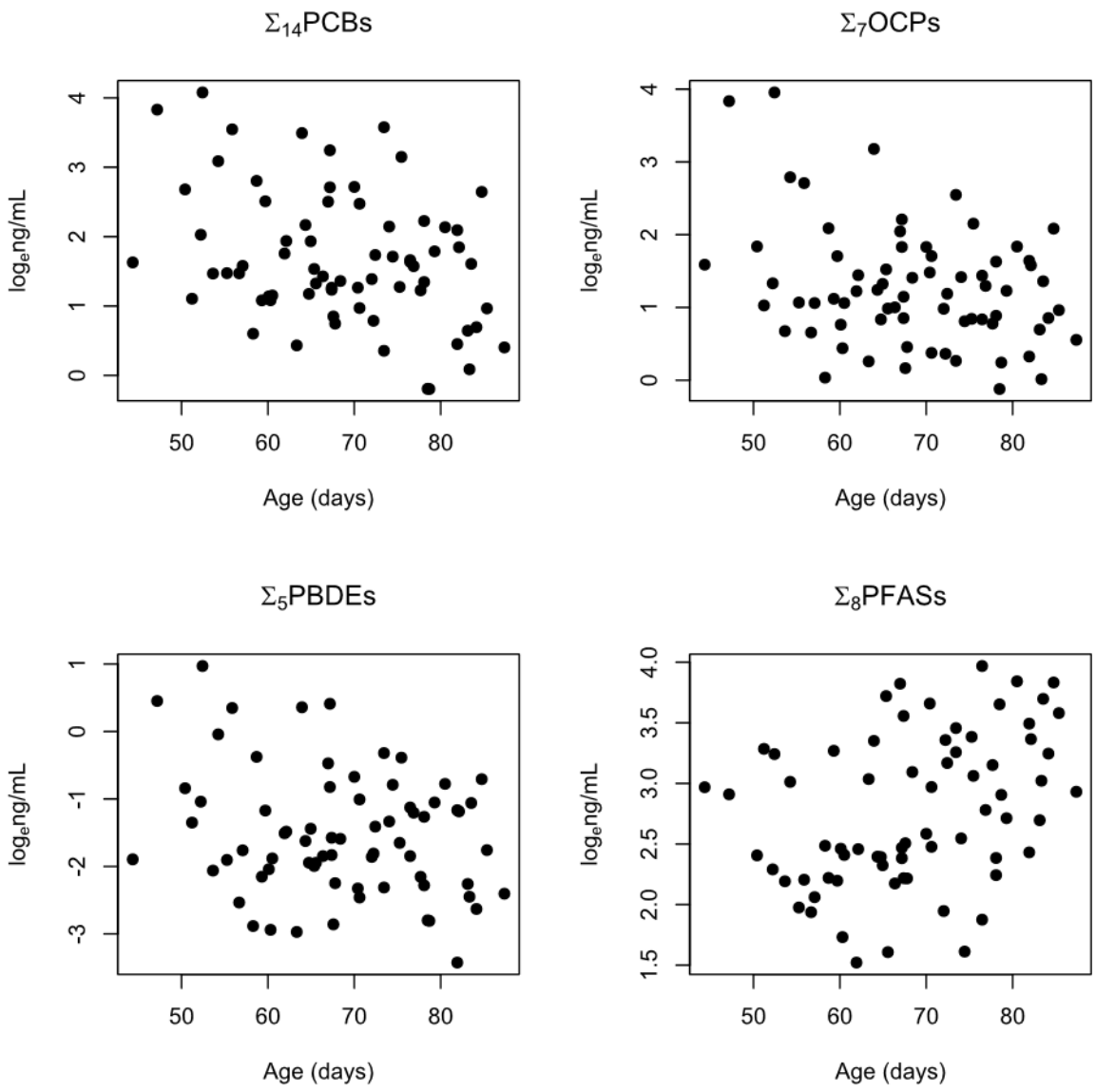


Figure S6: Plots showing the correlation between $\log_e$ concentrations of each contaminant group and age of the nestlings.

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

Table S12: Model selection table of all competing models for variation of **Σ_{14}PCBs** in plasma from white-tailed eagle nestlings from Smøla and Steigen, sampled in 2015 and 2016. Models are ranked by increasing AICc. All models include nest as a random factor. The year variable (Yr) represents 2016 and location variable (Loc) represents Steigen

	Intercept	Age	δ^{13}C	Loc	Yr	Loc:Yr	df	logLik	ΔAICc	ΔAICc	weight
m3	-3.10	-0.03	-0.36	+			6	-71.04	155.4	0.00	0.32
m4	-2.61	-0.03	-0.35				5	-72.65	156.2	0.81	0.21
m1	-3.76	-0.02	-0.35	+	+	+	8	-69.05	156.5	1.03	0.19
m2	-2.95	-0.03	-0.34	+	+		7	-70.99	157.8	2.37	0.10
m7	-2.47	-0.03	-0.34		+		6	-72.59	158.5	3.10	0.07
m6	-5.43		-0.33	+	+	+	7	-71.94	159.7	4.27	0.04
m13	3.55	-0.03		+			5	-74.96	160.8	5.43	0.02
m15	3.94	-0.03					4	-76.30	161.2	5.80	0.02
m5	2.59	-0.02		+	+	+	7	-72.71	161.2	5.81	0.02
m9	3.29	-0.03		+	+		6	-74.39	162.1	6.69	0.01
m11	3.68	-0.03			+		5	-75.75	162.4	7.02	0.01
m14	0.83			+	+	+	6	-75.20	163.7	8.31	0.01
m12	-5.02		-0.34	+			5	-76.94	164.8	9.41	0.00
m8	-4.49		-0.31	+	+		6	-76.53	166.4	10.97	0.00
m16	-4.42		-0.32				4	-79.85	168.3	12.90	0.00
m18	1.44			+			4	-80.07	168.8	13.33	0.00
m10	-3.84		-0.29		+		5	-79.38	169.7	14.28	0.00
m17	1.48				+		4	-81.30	171.2	15.79	0.00

Table S13: Model selection table of all competing models for variation of **Σ_{7}OCPs** in plasma from white-tailed eagle nestlings from Smøla and Steigen, sampled in 2015 and 2016. Models are ranked by increasing AICc. All models include nest as a random factor. The year variable (Yr) represents 2016 and location variable (Loc) represents Steigen. Two outliers were removed from these analyses to ensure normality of the residuals (n = 68).

	Intercept	Age	δ^{13}C	Loc	Yr	Loc:Yr	df	logLik	AICc	ΔAICc	weight
m1	-5.04	-0.01	-0.36	+	+	+	8	-42.51	103.50	0.00	0.38
m6	-5.75		-0.35	+	+	+	7	-43.87	103.60	0.15	0.35
m2	-4.49	-0.02	-0.36	+	+		7	-45.20	106.30	2.82	0.09
m3	-4.03	-0.01	-0.32	+			6	-46.45	106.30	2.83	0.09
m4	-3.68	-0.02	-0.32				5	-48.97	108.90	5.46	0.03
m12	-4.82		-0.31	+			5	-49.05	109.10	5.62	0.02
m7	-4.11	-0.02	-0.36		+		6	-47.90	109.20	5.72	0.02
m8	-5.24		-0.34	+	+		6	-48.47	110.30	6.86	0.01
m16	-4.47		-0.30				4	-52.58	113.80	10.35	0.00
m14	0.80			+	+	+	6	-50.97	115.30	11.86	0.00
m10	-4.81		-0.33		+		5	-52.20	115.40	11.91	0.00
m13	1.99	-0.01		+			5	-52.44	115.90	12.40	0.00
m18	1.06			+			4	-54.32	117.30	13.82	0.00
m15	2.34	-0.02					4	-54.56	117.80	14.31	0.00
m9	2.08	-0.01		+	+		6	-52.33	118.00	14.59	0.00
m11	2.43	-0.02			+		5	-54.47	119.90	16.46	0.00
m17	1.29				+		4	-57.14	122.90	19.46	0.00

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028

Table S14: Model selection table of all competing models for variation of **Σ_5PBDEs** in plasma from white-tailed eagle nestlings from Smøla and Steigen, sampled in 2015 and 2016. Models are ranked by increasing AICc. All models include nest as a random factor. The year variable (Yr) represents 2016 and location variable (Loc) represents Steigen.

	Intercept	Age	δ^{13}C	Loc	Yr	Loc:Yr	df	logLik	AICc	ΔAICc	weight
m4	-6.71	-0.03	-0.38				5	-72.48	155.9	0.00	0.24
m1	-8.50	-0.02	-0.43	+	+	+	8	-69.00	156.4	0.46	0.19
m3	-7.09	-0.03	-0.38	+			6	-71.55	156.4	0.54	0.18
m7	-7.30	-0.03	-0.43		+		6	-71.70	156.7	0.83	0.16
m2	-7.69	-0.03	-0.43	+	+		7	-70.72	157.2	1.34	0.12
m6	-10.00		-0.43	+	+	+	7	-71.17	158.2	2.25	0.08
m12	-8.73		-0.37	+			5	-75.51	161.9	6.05	0.01
m15	0.40	-0.03					4	-76.85	162.3	6.41	0.01
m13	0.10	-0.02		+			5	-76.11	163.2	7.26	0.01
m16	-8.32		-0.36				4	-77.42	163.5	7.55	0.01
m8	-9.20		-0.40	+	+		6	-75.21	163.7	7.84	0.01
m11	0.47	-0.03			+		5	-76.81	164.6	8.66	0.00
m5	-0.49	-0.02		+	+	+	7	-74.74	165.3	9.38	0.00
m10	-8.72		-0.39		+		5	-77.20	165.3	9.44	0.00
m9	0.17	-0.03		+	+		6	-76.07	165.5	9.57	0.00
m14	-2.04			+	+	+	6	-76.54	166.4	10.51	0.00
m18	-1.68			+			4	-79.49	167.6	11.71	0.00
m17	-1.49				+		4	-80.97	170.6	14.66	0.00

Table S15: Model selection table of all competing models for variation of **Σ_8PFASs** in plasma from white-tailed eagle nestlings from Smøla and Steigen, sampled in 2015 and 2016. Models are ranked by increasing AICc. All models include nest as a random factor. The year variable (Yr) represents 2016 and location variable (Loc) represents Steigen.

	Intercept	Age	δ^{13}C	Loc	Yr	Loc:Yr	df	logLik	AICc	ΔAICc	weight
m9	1.66	0.02		+	+		6	-9.10	31.5	0.00	0.56
m2	2.22	0.02	0.03	+	+		7	-8.91	33.6	2.11	0.19
m5	1.58	0.02		+	+	+	7	-8.94	33.7	2.17	0.19
m1	2.13	0.02	0.03	+	+	+	8	-8.76	35.9	4.36	0.06
m14	3.16			+	+	+	6	-20.81	54.9	23.42	0.00
m8	3.15		0.00	+	+		6	-21.43	56.2	24.66	0.00
m11	2.16	0.02			+		5	-22.73	56.4	24.86	0.00
m6	3.29		0.01	+	+	+	7	-20.80	57.4	25.88	0.00
m7	2.77	0.02	0.03		+		6	-22.59	58.5	26.99	0.00
m17	3.30				+		4	-29.30	67.2	35.69	0.00
m10	3.42		0.01		+		5	-29.30	69.5	38.01	0.00
m3	3.55	0.02	0.14	+			6	-30.48	74.3	42.77	0.00
m13	1.00	0.02		+			5	-32.35	75.6	44.12	0.00
m4	3.88	0.02	0.13				5	-36.75	84.4	52.92	0.00
m15	1.45	0.02					4	-38.18	85	53.46	0.00
m18	2.57			+			4	-42.94	94.5	62.97	0.00
m12	4.58		0.11	+			5	-42.17	95.3	63.75	0.00
m16	4.74		0.10				4	-45.36	99.3	67.81	0.00

This is a post-print version of an article published in Environmental Pollution.
The final version is available online at: https://dx.doi.org/10.1016/j.envpol.2018.12.028