Environmental Science and Pollution Research
https://doi.org/10.1007/s11356-018-1844-2

RESEARCH ARTICLE

Contrasting congener profiles for persistent organic pollutants and PAH monitoring in European storm petrels (*Hydrobates pelagicus*) breeding in Ireland: a preen oil versus feathers approach

Heidi Acampora[1] · **Philip White**[1] · **Olga Lyashevska**[1] · **Ian O'Connor**[1]

Received: 2 November 2017 / Accepted: 20 March 2018

Abstract

Persistent organic pollutants (POPs) and polycyclic aromatic hydrocarbons (PAHs) are anthropogenic contaminants of environmental concern due to their persistence in the environment and capacity to accumulate in biota. Many of these contaminants have been found to have ill effects over wildlife and humans. Birds are known to be particularly affected through endocrine disruption and eggshell thinning. POPs have been banned or restricted through the Stockholm Convention (2001), making monitoring essential for tracking effects of regulation. Seabirds have been used as monitoring tools for being top predators and consuming a diverse array of prey in different trophic levels. Non-destructive sampling has become widely popular using feathers and preen oil, as opposed to carcasses and internal organs. This study aimed to set baseline levels of POP and PAH concentration in a highly pelagic and abundant seabird in Ireland and the Atlantic, the European storm petrel, *Hydrobates pelagicus*, and to investigate the profiles of contaminant congeners in preen oil and feathers, comparatively. Mean concentrations in preen oil followed: PCB (10.1 ng/g *ww*) > PAH (7.1 ng/g *ww*) > OCP (5.4 ng/g *ww*) > PBDE (3.9 ng/g *ww*), whilst mean concentrations in feathers followed the order: PAH (38.9 ng/g *ww*) > PCB (27.2 ng/g *ww*) > OCP (17.9 ng/g *ww*) > PBDE (4.5 ng/g *ww*). Congener profiles highly differed between preen oil and feathers, and little correlation was found between the matrices. These results demonstrate that the sampling of a single matrix alone (preen oil or feathers) might produce confounding results on contamination in seabirds and that more than one matrix is recommended to obtain a full picture of contamination by persistent organic pollutants.

Keywords Persistent organic pollutants · European storm petrel · POPs · PCBs · PAHs · OCPs · PBDEs

Highlights
• First assessment of levels for persistent organic pollutants and PAHs in European storm petrels in Ireland
• A comparison between matrices: preen oil vs feathers
• PCBs, PAHs, OCPs and PBDEs were detected and quantified for 32 individuals
• Congener profiles highly differed between feathers and preen oil

Responsible editor: Philippe Garrigues

Electronic supplementary material The online version of this article (https://doi.org/10.1007/s11356-018-1844-2) contains supplementary material, which is available to authorized users.

✉ Heidi Acampora
heidiacampora@gmail.com

[1] Marine and Freshwater Research Centre, Galway-Mayo Institute of Technology, Dublin Rd, Galway, Ireland

Introduction

Persistent organic pollutants (POPs) are chemical compounds used for various industrial purposes. Most of them are of anthropogenic origin. Polycyclic aromatic hydrocarbons (PAHs) however might also arise from natural sources, such as the incomplete combustion of fossil fuels. POPs include polychlorinated biphenyls (PCBs), organochlorine pesticides (OCPs) and polybrominated diphenyl ethers (PBDEs). Along with PAHs, these are lipophilic compounds that accumulate in the environment and in biological tissues and magnify through food webs (Jaspers et al. 2006b).

PCBs are used in a range of industrial products due to their stabilising nature and low flammability (Brinkman and De Kok 1981; Stockholm Convention 2001). PCBs have been connected to endocrine disruption and in certain regions and consequently have been linked to population decline in birds

Published online: 05 April 2018

(Gilbertson et al. 1976; Jones and de Voogt 1999; Stockholm Convention 2001).

OCPs have been found to negatively impact the nervous, immune and endocrine systems of birds, consequently affecting reproduction (Furness 1993). Some OCP compounds have been correlated with failed reproduction in fish-eating bird populations (Bosveld and van den Berg 2002; Giesy et al. 1994; Kubiak et al. 1989).

Various PBDE congeners have been reported to cause endocrine disruption and developmental abnormalities (Darnerud 2008; Eng et al. 2012; Winter et al. 2013) in birds. Although they are highly hydrophobic and supposedly difficult to leach out from plastics, studies have shown that stomach oil, present in Procellariiform birds, such as European storm petrels (*Hydrobates pelagicus*), may facilitate and accelerate leaching due to their high lipid content (Tanaka et al. 2015).

PAHs have been shown to bioaccumulate in invertebrate species (Meador et al. 1995) and are known to be highly toxic, carcinogenic and mutagenic (Laffon et al. 2006; Stockholm Convention 2001). In the aquatic environment, they tend to accumulate in sediments (MacRae and Hall 1998), affecting benthic organisms at the bottom of the food web (Roscales et al. 2011b). However, recent studies have revealed that bioaccumulation via food webs is negligible for PAHs (Nfon et al. 2008; Perugini et al. 2007; Wan et al. 2007).

The Stockholm Convention established a list of POPs that were to be banned or restricted; this treaty signed by various countries in 2001 came into effect in 2004 (Stockholm Convention 2001). Levels of PCBs and DDT (dichlorodiphenyltrichloroethane, an OCP) have decreased recently due to restrictions imposed by legislation (Hammer et al. 2016; Jones and de Voogt 1999; Miller et al. 2014; Roscales et al. 2016). It is part of the treaty that countries monitor the amounts of the listed pollutants and the convention has also been amended with emerging POPs, which are new contaminants that are not or have only recently been regulated (Trumble et al. 2012). POPs magnify through the food webs via contamination of lipid-rich food (Fromant et al. 2016; Matthies et al. 2016; Safe and Hutzinger 1984) and are globally distributed through long-range atmospheric and oceanic transport mechanisms (Jones and de Voogt 1999). As seabirds feed on a variety of trophic levels, contaminants might have been bioaccumulated in prey before ingested by them, making seabirds ideal candidates for persistent organic pollutant monitoring (Borlakoglu et al. 1990a, b).

The variation in congener profiles in organisms and trophic levels can be attributed to differences in bioaccumulation and metabolism, which can vary in different matrices (Wang et al. 2015). For a long time, POPs have been monitored through tissue such as liver, muscle or brain (Falkowska et al. 2016; Mallory and Braune 2012; Sagerup et al. 2009), but subsequent research has called for non-destructive ways of monitoring. The use of eggs has become widespread (Elliott et al. 2005; Jörundsdóttir et al. 2010). A single egg can provide baseline levels for a whole clutch (Van den Steen et al. 2006), but various species of seabirds produce a single egg per season, making this type of monitoring more sensitive to restrictions. Blood sampling is also considered a non-destructive and efficient approach, but it requires more cautious training and restrictive storage of samples in the field (Van Den Brink and Bosveld 2001). The use of feathers (Jaspers et al. 2006b) and, subsequently, preen oil has proved successful. Preen oil has been shown to provide levels comparable to internal organs (Jaspers et al. 2008).

Many studies have focused on non-migratory species to reflect local contamination (Chen et al. 2013; Jaspers et al. 2006a), but few studies have taken migratory pelagic species, from which migratory routes are known and could account for local and/or transboundary contamination. This study focused on a highly pelagic species, the European storm petrel, which is very abundant in the North Atlantic (over 99,000 breeding pairs in Ireland (Mitchell et al. 2004)). The aim of this research was to (1) set baseline levels for four different classes of pollutants in a European storm petrel population breeding in Ireland and (2) to investigate the relative contribution of pollutants and their profiles in breast feathers and in preen oil.

Material and methods

Preen oil and feather sampling

European storm petrels (*Hydrobates pelagicus*) were caught ($n = 32$) using mist nets at Portacloy, County Mayo, Ireland, under licence numbers C124/2015 and C125/2015 from Ireland's National Parks and Wildlife Service (NPWS) in August 2015. Each bird was weighed, had its wing span measured and was ringed. Preen oil was collected using sterile cotton swabs and metal forceps once the preen gland was exposed and gently pressed to express the oil. Swabs were placed in individual sterile glass jars with foil covered lids. Additionally, four breast feathers were collected from each bird and placed into individual paper envelops. Upon completion of sampling and ringing, the birds were released. Preen oil swabs were kept frozen at – 80 °C, whilst feathers were kept at room temperature. Extraction methods were performed according to Espin et al. (2012), Jaspers et al. (2008) and Van Den Brink (1997).

Preen oil extraction

All utensils were washed with methanol (Merck SupraSolv). Cotton swabs were removed from glass containers using forceps and transferred into glass beakers and were spiked with internal standards. Sample jars were also rinsed with methanol, which was added to the sample. In total, 150 ml of methanol was poured into each beaker (in three aliquots) and the contents agitated for 1 min. The combined aliquots were transferred to another beaker and heated gently on a hot plate (60 °C max) to remove excess solvent. The remaining 1 ml were transferred into labelled GC vials. Samples were kept frozen at – 20 °C until subsequent analysis using gas chromatography-mass spectrometry (GC-MS).

Feather extraction

All utensils were washed with methanol (Merck SupraSolv). Four feathers from each bird were placed in individual beakers. Feathers were washed with distilled water, using forceps to separate the barbs, and stirred. Feathers were left soaking for 20 min and then left to dry in folded tissue paper for 2 h or until fully dried. After drying, each sample was weighed. Each feather sample (4) was placed inside a beaker and had internal standards added along with 15 ml of 37% HCl (Merck EMSURE) and 20 ml of 2:1 *v/v* n-hexane (VWR Analar Normapur)/acetone (Merck SupraSolv). Beakers were covered with tin foil and placed in an oven at 37 °C for approximately 15 h. Forty milliliters of a 3:1 *v/v* n-hexane/acetone solvent mixture was added to the sample before they were placed in a clean separatory funnel and shaken vigorously. The aqueous layer was removed to a beaker. The remaining organic layer was decanted into previously labelled glass vials before the aqueous layer was re-extracted with 20 ml of fresh 2:1 n-hexane/acetone solvent mixture. The organic layers were combined and transferred into a TurboVap LP (Biotage) and evaporated under nitrogen (40 °C, 7.35 psi max) to 1 ml. This was subsequently transferred to pre-labelled GC vials using disposable glass pipettes. Samples were kept frozen at − 20 °C until subsequent analysis using GC/MS.

Gas chromatography-mass spectrometry

Preen oil and feather solvent extracts were then analysed for PCBs, PAHs, OCPs and PBDEs. Analysis were performed for the most common congeners formed during the production of PCBs and PBDEs, the most commonly used OC pesticides and the PAHs found on the Environmental Protection Agency (EPA) list (Supplementary Tables B 1–4). An Agilent GC-MS (5977E) with 'Masshunter' software run in EI mode with a J&W DB-1 30 m x 0.25 mm × 0.25 μm column was used. The inlet was operated in splitless mode with the temperature at 260 °C, the ion source at 230 °C and the quadrupole at 150 °C. The auxiliary transfer line was set at 280 °C. Varying column oven temperature programs were used for different compound suites (Supplementary Table E). Helium was the carrier gas. The individual analytes were initially identified by MS scan (50–550 *m/z*) using individuals and the NIST 'structural elucidation' software. Once the compounds were identified, the machine was then operated in SIM (single ion monitoring) and quantification was achieved spiking samples with 100 mg of internal standards (PBDE: ^{13}C PBDE—47,153; PAH: acenaphthene$_{-d10}$, phenanthrene$_{-d10}$, chrysene$_{-d12}$ and perylene$_{-d12}$; PCB (OCP—external standards): ^{13}C-PCB 52 and 153) corresponding to a concentration of 20 ng/g for each standard (Supplementary Table A). Quality was satisfied using procedural blanks and solvent blanks, system suitability standards and the analysis of certified reference materials (CRMs) (Supplementary Table C). Cod liver oil (Commission of the European Communities, Community Bureau of Reference—BCR. Reference Material No. 349. Chlorobiphenyls in cod liver oil no. 0831) and NIST 1947 (Lake Michigan Fish Tissue. US Department of Commerce, National Institute of Standards and Technology, Gaithersburg, MD, 20899) were used. During the instrumental analysis, quality control aspects used included the scrutiny of system control standards to ensure that there was no instrumental drift. A system suitability standard of known concentration (~ 10 ng/g) was included after every 10 samples were analysed. The estimated sample concentrations were only judged to be correct if the system suitability standard showed a concentration range between 90 and 110%. In the case where the concentration of system suitability standard was outside this range, the instrument was stopped and the sample data discarded. Instrumental maintenance was completed and the system was re-verified, and the analysis continued from the last passed system suitability sample. Supplementary information found in Table D contains all the produced quality data. Analysis was performed according to Acampora et al. (2017) and White et al. (2014).

Statistical analysis

A principal component analysis (PCA) was conducted in R (R Core Team 2015) version 3.2.3 using 'prcomp' package. The directions of maximum variance in high-dimensional data were identified and projected onto a smaller dimensional subspace whilst retaining most of the information. The calculation was done by a singular value decomposition of the centred and scaled data matrix. Original variables were transformed into a set of values of orthogonal variables called principal components (PCs). PCs are normalised linear combinations of the original predictors in a data set. PC1 captures the maximum variance and determines the direction of highest variability in the data. The following principle components (e.g. PC2, PC3) capture the remaining variance. This analysis was used to identify the main contributors to the burden of each contaminant in each matrix (preen oil and feather) and to investigate any differences between the two matrices. Congeners that had over 50% of values below the level of quantification (LOQ) were excluded from statistical analysis (Behrooz et al. 2009; Jaspers et al. 2007b; Jaspers et al. 2006b). Values below level of detection (LOD) were excluded. Levels above LOD, but below LOQ, were turned into LOD/2. Pearson's correlation was calculated for each contaminant group to see if there was any correlation between the two matrices sampled (preen oil and feathers) at the individual level and through aggregated data.

Enrichment factors (EF) were calculated for three main PCB commercial mixtures (Aroclor 1248, 1254 and 1260) using the same method used by Borlakoglu et al. (1990a, b), in which PCB abundance concentration is compared to that of popular industrial mixtures. An enrichment factor > 1 suggests the accumulation of the pollutant, rather than the excretion, whilst an enrichment factor < 1 suggests the metabolising of the pollutant and its removal from adipose tissue by the body (Borlakoglu et al. 1990a, b).

Additionally, potential sources of PAHs were calculated using the ratio phenanthrene/anthracene (P/A) and fluoranthene/pyrene (Fl/Py) (Webster et al. 2000). A P/A ratio < 10 indicates a pyrogenic source, whilst a > 10 ratio indicates a petrogenic source. A Fl/Py ratio > 1 suggests a pyrogenic source of contamination, and a < 1 ratio indicates a petrogenic source.

Results

In total, 16 PCB congeners were detected in preen oil and feathers. The average total PCB in preen oil was 10.01 ng/g *ww* preen oil (range 2.75–18.71 ng/g *ww*). The average sum of seven PCB indicators (PCB 28, − 52, − 101, − 118, − 153, − 138, − 180) was 7.28 ng/g *ww* preen oil (range 2.37–14.62 ng/g *ww*). In feathers, the average total PCB was 27.2 ng/g *ww* feather (range 4.23–136.7 ng/g *ww*). The average sum of seven PCBs was 20.2 ng/g *ww* feather (range 3.24–99.3 ng/g *ww*) (Table 1; Fig. 1). Results from the PCA analysis for preen oil showed that over 81% of the variance was explained by the first three components (cumulative proportion for principal components (PC): PC1 = 0.56, PC2 = 0.71, PC3 = 0.81). The congeners that had a higher relative contribution to PCB burden in preen oil were PCB − 156, − 153 and 18, respectively. For feathers, similarly, the first three components explained 82% of the variance (PC1 = 0.58, PC2 = 0.73, PC3 = 0.82). The highest contributing congeners were PCB 105 (closely with PCB 28), − 170 and − 209, respectively.

Fifteen PAH congeners were detected; however, five of these were only detected in feather samples, but not in preen oil. The average total PAH in preen oil was 7.1 ng/g *ww* preen oil (range 1.81–14.1 ng/g *ww*). In feathers, the average total PAH was 38.9 ng/g *ww* feather (range 9.59–218.9 ng/g *ww*) (Table 2; Fig. 1). The P/A ratio for PAHs in preen oil indicated a pyrogenic source (P/A = 54.5), whilst the Fl/Py ratio suggested a petrogenic source (Fl/Py = 0.42). These ratios for feathers, however, for both P/A (19.3) and Fl/Py (0.3)

Table 1 PCB congeners detected in preen oil and feathers are listed, including their mean concentration ± standard deviation (SD) and enrichment factors in comparison to three of the most popular commercial mixtures (Aroclor 1248, 1254 and 1260). *NM* not measured, *PO* preen oil, *FE* feathers, *ww* wet weight, 7 PCBs = PCB 28, - 52, - 101, - 118, - 138, - 153, and - 180

PCB	Preen oil (ng/g *ww*) ± SD	Feathers (ng/g *ww*) ± SD	Enrichment factor (Aroclor 1248)		Enrichment factor (Aroclor 1254)		Enrichment factor (Aroclor 1260)	
			PO	FE	PO	FE	PO	FE
PCB 18	0.04 ± 0.08	0.66 ± 0.75	0.13	0.74	1.70	9.75	8.50	48.74
PCB 31	0.15 ± 0.19	1.58 ± 1.71	0.34	1.06	6.61	20.72	46.27	145.03
PCB 28	0.19 ± 0.21	1.79 ± 1.78	0.41	1.18	12.12	34.50	76.74	218.48
PCB 52	0.84 ± 0.89	4.18 ± 4.88	1.83	2.75	1.90	2.85	42.57	63.86
PCB 44	1.45 ± 4.7	1.59 ± 1.68	1.46	1.15	3.22	2.53	248.05	194.56
PCB 101	1.16 ± 0.58	5.36 ± 5.81	7.46	10.41	1.76	2.45	4.50	6.28
PCB 118	1.29 ± 0.7	3.90 ± 4.87	6.65	6.09	2.13	1.95	32.56	29.83
PCB 105	0.55 ± 0.45	1.19 ± 1.34	4.59	3.00	2.22	1.45	30.24	19.77
PCB 149	0.25 ± 0.23	1.29 ± 2.43	9.13	14.31	0.83	1.29	0.34	0.54
PCB 153	2.53 ± 5.6	2.73 ± 4.05	42.17	23.28	4.81	2.66	1.93	1.07
PCB 138	0.72 ± 0.48	1.55 ± 2.52	21.45	13.89	1.52	0.98	1.34	0.87
PCB 156	0.22 ± 0.23	0.37 ± 0.57	65.71	33.99	3.21	1.66	5.05	2.61
PCB 180	0.55 ± 0.36	0.72 ± 1.49	31.91	12.49	10.00	3.91	0.59	0.23
PCB 170	0.09 ± 0.11	0.23 ± 0.50	13.94	10.73	2.14	1.65	0.27	0.21
PCB 194	0.06 ± 0.06	0.10 ± 0.11	NM	NM	73.58	35.65	0.36	0.17
PCB 209	0.02 ± 0.03	0.02 ± 0.01	NM	NM	NM	NM	NM	NM
∑ *all PCBs*	10.10 ± 0.66	27.26 ± 1.52						
∑ *7 PCBs*	7.28 ± 1.78	20.23 ± 1.56						

suggested petrogenic sources. Results from the PCA analysis for preen oil showed that the first five components retained 75% of the variation (PC1 = 0.26, PC2 = 0.42, PC3 = 0.55, PC4 = 0.66, PC5 = 0.75). Congeners that most contributed to the PAH burden in preen oil were pyrene, fluorene, fluoranthene and benzo(a)pyrene together and benzo(ghi)perylene, respectively. In feathers, the four first components explained 78% of the variation (PC1 = 0.38, PC2 = 0.54, PC3 = 0.68, PC4 = 0.78). The most contributing congeners for PAH burden in feathers were fluoranthene, dibenzo(a, h)anthracene, benzo(a)pyrene and indeno(1, 2, 3, CD)pyrene, respectively.

Fifteen OCP congeners were detected in preen oil and feathers. The average total OCP in preen oil was 5.4 ng/g *ww* preen oil (range 3.19–12.9 ng/g *ww*). The average total OCP in feathers was 17.9 ng/g *ww* feather (range 3.48–46.2 ng/g *ww*) (Table 3; Fig. 1). PCA results for preen oil showed that 64% of the total variance was explained by the first five components (PC1 = 0.19, PC2 = 0.34, PC3 = 0.47, PC4 = 0.56, PC5 = 0.64). Congeners that contributed most to OCP burden in preen oil were pp-DDD, isodrin, heptachlor epoxide and op-DDT, respectively. In feathers, the first five components explained 71% of the variation (PC1 = 0.26, PC2 = 0.41, PC3 = 0.54, PC4 = 0.63, PC5 = 0.71). Congeners that most contributed to the OCP burden in feathers were dieldrin together with HCB, endosulphan B, isobenzan and aldrin, respectively.

Seven PBDE congeners in total were detected in preen oil and feathers. One of these, however, was only detected in feathers (PBDE 183). The average total PBDE in preen oil was 3.91 ng/g *ww* preen oil (range 1.74–34.4 ng/g *ww*). For the feathers, the average total PBDE was 4.59 ng/g *ww* feather (range 1.96–15.9 ng/g *ww*) (Table 4; Fig. 1). Table 5 lists the average concentration of PBDE congeners in comparison to three main commercial mixtures (penta, octa and deca). Concentrations in preen oil and feathers are higher in all congeners compared to commercial mixtures, except in PBDE 47 and − 99 or when not available (not measured). PCA results for preen oil have shown that the three first components explained 86% of the variation (PC1 = 0.48, PC2 = 0.73, PC3 = 0.86). The congeners that most contributed to PBDE burden in preen oil were PBDE 100, − 183 and − 153, respectively. In feathers, the four first components explained 79% of the variation (PC1 = 0.25, PC2 = 0.49, PC3 = 0.65, PC4 = 0.79). Congeners that contributed most to the PBDE burden in feathers were PBDE 28, − 100, − 153 and − 183, respectively.

Results from Pearson's correlation matrices showed no correlation or a weak correlation between preen oil and feathers for all contaminants investigated at the individual level (Fig. 2). A positive correlation was only found for pp-DDE and PBDE 47 and − 154. On aggregated data however, PCB showed a moderate correlation (0.61), whilst PAH showed a strong correlation (0.78) between feathers and preen oil.

Discussion

Data for persistent organic pollutant levels in seabirds in Ireland are limited and dates back between the 1960s and 80s (Earley 1987; Knight and Walker 1982; Koeman et al. 1967; Moore and Tatton 1965). PCBs and some OCPs have been measured in egg, brain and adipose tissue of guillemots, puffins, razorbills (Borlakoglu et al. 1990b), tern species (Koeman et al. 1967), shags and cormorants (Borlakoglu et al. 1990b; Earley 1987; Knight and Walker 1982; Moore and Tatton 1965). This data is of low comparability to this work due to the difference in matrices (destructive vs non-destructive sampling) and species, but it provides a measure of the presence and levels of POPs in Irish seabird populations in the latter half of the twentieth century. To our knowledge, no data is available for POPs in European storm petrels in Ireland.

The first aim of this work was to set baseline concentration values for the main persistent organic pollutants and PAHs found in European storm petrels' breeding in Ireland. The mean concentration of PCBs in preen oil was 10.1 ng/g *ww* preen oil (∑ 7 PCBs = 7.28 ng/g *ww*), whilst for feathers was 27.2 ng/g *ww* feather (∑ 7 PCBs = 20.2 ng/g *ww*). The levels in feathers differ from the ones found in Leach's storm petrels (*Oceanodroma leucorhoa*), a similar species in the UK, which are higher (∑ 7 PCBs = 36.2 ng/g), and in Canada, which are lower (∑ 7 PCBs = 10.6 ng/g) (Megson et al. 2014). Leach's in Alaska also had lower concentration levels in comparison to our samples from European storm petrels in ∑ PCB in liver composites (0.24 ng/g). Differences can be explained by the difference in matrices and the levels of industrialisation (Roscales et al. 2011b).

Enrichment factor calculations suggest that the great majority of PCBs are being accumulated rather than excreted by European storm petrels. For the three main commercial mixtures, 78, 93 and 73% of the congeners in preen oil are being accumulated in comparison to Aroclor 1248, 1254 and 1260, respectively. In feathers, 92, 93 and 66% are accumulated rather than excreted. When compared to Aroclor 1248, the metabolisation is of low chlorinated congeners as opposed to Aroclor 1260, in which low enrichment factors are present in high-chlorinated congeners. In theory, low molecular weight compounds are easier to metabolise and can be excreted over time (Borlakoglu et al. 1990a, b; Ludwig et al. 1996). Research has shown though that fish-eating seabirds have low capacity to metabolise PCBs efficiently, regardless of their molecular weight (Borlakoglu et al. 1990a, b; Walker 1990).

∑ PAH concentrations in feathers (38.9 ng/g *ww*) were fivefold higher than the ones found for preen oil (7.1 ng/g *ww*). This value is similar, yet higher for the white-faced storm petrel (*Pelagodroma marina*), a related species reported by (Roscales et al. 2011a) to have a mean concentration of 29.8 ng/g *ww* in the liver. Sources of PAHs for preen oil

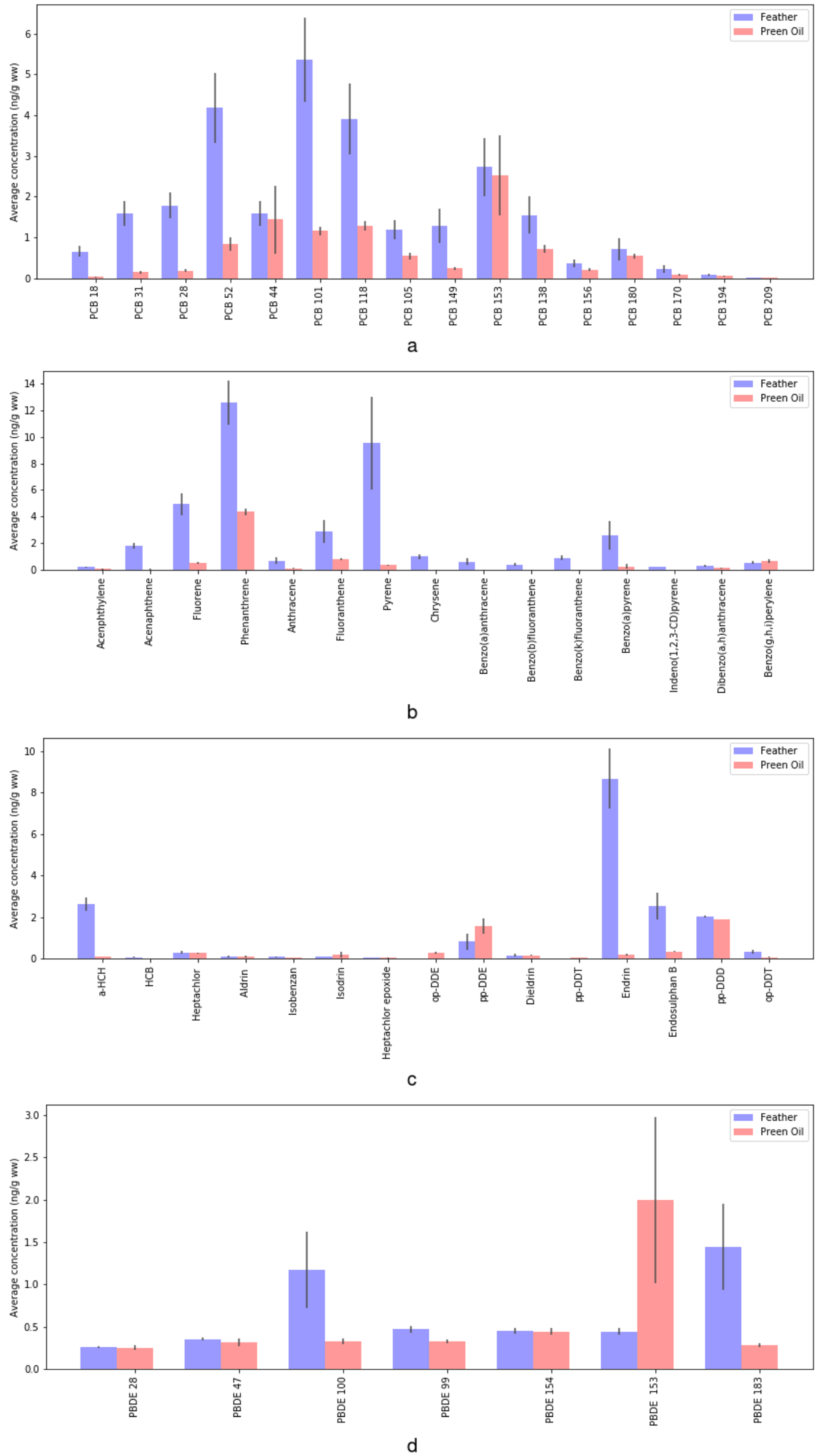

Fig. 1 Average concentration (ng/g *ww*) of PCBs (**a**), PAHs (**b**), OCPs (**c**) and PBDEs (**d**) comparatively for feathers and preen oil per congener

Table 2 PAH congeners and their mean concentration ± standard deviation (SD) in preen oil and feather samples. *ND* not detected

PAH	Preen oil (ng/g *ww*)	Feathers (ng/g *ww*)
Acenphthylene	0.04 ± 0.11	0.19 ± 0.20
Acenaphthene	0.01 ± 0.05	1.78 ± 1.33
Fluorene	0.50 ± 0.46	4.91 ± 4.65
Phenanthrene	4.36 ± 1.47	12.59 ± 9.27
Anthracene	0.08 ± 0.15	0.65 ± 1.34
Pyrene	0.80 ± 0.45	9.55 ± 4.93
Fluoranthene	0.34 ± 0.23	2.87 ± 19.80
Benzo(a)anthracene	ND	0.60 ± 0.95
Chrysene	ND	0.98 ± 1.46
Benzo(k)fluoranthene	ND	0.88 ± 0.46
Benzo(b)fluoranthene	ND	0.38 ± 0.88
Benzo(a)pyrene	0.24 ± 1.17	2.59 ± 6.20
Indeno(1,2,3-CD)pyrene	ND	0.20 ± 0.11
Dibenzo(a,h)anthracene	0.09 ± 0.25	0.28 ± 0.57
Benzo(g,h,i)perylene	0.65 ± 0.72	0.52 ± 0.54
∑ *PAH*	7.10 ± 1.06	38.96 ± 3.59

suggested high pyrogenic and low petrogenic origins. Whilst for feathers, both ratios indicated petrogenic sources. Petrogenic sources indicate that the PAHs detected originated from petroleum and crude oils (Stogiannidis and Laane 2015), which is fitting for a bird that spends most of its time at sea, whilst a pyrogenic source indicates that PAHs are derived from the combustion of fuels (Stogiannidis and Laane 2015). Since preen oil is renewed and metabolised constantly, it makes sense that it could contain PAH from different

Table 3 OCP mean concentration ± standard deviation (SD) per congener measured in preen oil and feather samples

OCP	Preen oil (ng/g *ww*)	Feathers (ng/g *ww*)
a-HCH	0.11 ± 0.05	2.63 ± 1.79
HCB	0.02 ± 0.03	0.07 ± 0.18
Heptachlor	0.28 ± 0.12	0.30 ± 0.37
Aldrin	0.10 ± 0.21	0.10 ± 0.18
Isobenzan	0.07 ± 0.05	0.10 ± 0.11
Isodrin	0.19 ± 0.73	0.11 ± 0.07
Heptachlor epoxide	0.04 ± 0.03	0.05 ± 0.04
op-DDE	0.30 ± 0.22	0.03 ± 0.02
pp-DDE	1.57 ± 1.97	0.82 ± 2.23
Dieldrin	0.16 ± 0.19	0.17 ± 0.39
pp-DDT	0.05 ± 0.06	0.02 ± 0.02
Endrin	0.20 ± 0.23	8.66 ± 8.2
Endosulphan B	0.35 ± 0.16	2.54 ± 3.57
pp-DDD	1.89 ± 0.03	2.02 ± 0.19
op-DDT	0.08 ± 0.09	0.33 ± 0.44
∑ *OCP*	5.40 ± 0.54	17.94 ± 2.19

Table 4 PBDE mean concentration ± standard deviation (SD) per congener given in preen oil and feathers. *ND* not detected

PBDE	Preen oil (ng/g *ww*)	Feathers (ng/g *ww*)
PBDE 28	0.25 ± 0.15	0.25 ± 0.03
PBDE 47	0.31 ± 0.24	0.35 ± 0.11
PBDE 100	0.32 ± 0.19	1.17 ± 2.5
PBDE 99	0.32 ± 0.13	0.46 ± 0.21
PBDE 154	0.44 ± 0.24	0.45 ± 0.19
PBDE 153	1.99 ± 5.54	0.44 ± 0.21
PBDE 183	0.28 ± 0.11	1.44 ± 2.86
∑ PBDE	3.91 ± 0.58	4.59 ± 0.42

sources, whilst feathers could be retaining them from time spent at sea. PAHs have been demonstrated not to have high degrees of bioaccumulation (Nfon et al. 2008; Perugini et al. 2007; Wan et al. 2007); thus, it is more likely that they have been acquired from the environment rather than from prey items.

OCP concentrations were approximately three fold higher in feathers (17.94 ng/g *ww*) than in preen oil (5.40 ng/g *ww*). ∑ DDT was higher in preen oil (0.13 ng/g *ww*) and feathers (0.35 ng/g *ww*) in European storm petrels from our study than from liver composites of the related Leach's storm petrel in Alaska (0.007 ng/g) (Ricca et al. 2008). The same is true for pp-DDE, which was 0.067 ng/g for Leach's storm petrels, in comparison to our species' concentration in preen oil (1.57 ng/g *ww*) and feathers (0.82 ng/g *ww*). Additionally, concentrations were lower for ∑ DRIN in Leach's (0.007 ng/g) in comparison to 0.65 ng/g *ww* in preen oil and 9.04 ng/g *ww* in feathers in European storm petrels. Concentrations were similar for ∑ CHLOR in Leach's (0.027 ng/g) in comparison with 0.032 ng/g *ww* in preen oil and 0.35 ng/g *ww* in feathers in birds from our study (Ricca et al. 2008).

PBDE concentrations in the eggs of Leach's storm petrels were similar (3.38 ng/g) (Elliott et al. 2005) to those found in our studies with preen oil and feathers (3.91 and 4.59 ng/g *ww*), respectively. Brominated flame retardants have become of greater concern recently due to plastic pollution at sea (Jaspers et al. 2006a, b; Karlsson et al. 2006; Miller et al. 2014; Tanaka et al. 2013). Many of these components are used as plastic additives and have been found to leach from plastic products, not only at sea, but also when plastics are ingested by seabirds (Tanaka et al. 2015). Procellariiform birds (such as the European storm petrel) are a family of birds from which many species have a tendency not only to ingest plastic, but also to accumulate it in their digestive tract due to a narrow connector between the gizzard and the proventriculus, which prevents regurgitation and can consequently affect excretion (Van Franeker et al. 2011). These birds are also known for containing a specific lipid-rich stomach oil, which they use as a defence mechanism by squirting it at predators when they

Table 5 Proportion (%) of PBDE congeners found in preen oil and feathers in comparison to main PBDE commercial mixtures (penta, octa and deca). *PO* preen oil, *FE* feathers, *ND* not detected, *EF* enrichment factor

PBDE	% PO	% FE	Penta Bromkal 70-5DE			Octa DE-79%	Deca Bromkal 82-ODE %
			%	PO EF	FE EF		
PBDE 28	7.95	5.88	0.1	79.5	58.8	ND	ND
PBDE 47	8.36	8.05	42.8	0.19	0.18	ND	ND
PBDE 100	8.25	26.80	7.82	1.05	3.42	ND	ND
PBDE 99	11.25	10.70	44.8	0.25	0.23	ND	ND
PBDE 154	50.58	10.30	2.68	18.8	3.84	1.07	ND
PBDE 153	7.19	10.11	5.32	1.35	1.90	8.66	ND
PBDE 183	6.99	28.17	0.33	21.1	85.3	42	ND

feel threatened (Ackman 1989; Connan et al. 2007). Research has shown that PBDEs are more prone to leaching from plastics when immersed in stomach oil rather than in sea water (Tanaka et al. 2015), indicating that this family of birds could be more affected by potential PBDE leaching and contamination than other organisms or birds. Conversely, Herzke et al. (2016) found pollutants in microplastics not to correlate to the body burden of Northern Fulmars that had ingested plastics and attributed these to pollutants in ingested prey. Plastic ingestion has been reported for other species of petrels (Bond and Lavers 2013; Colabuono et al. 2009; Ryan 2015; Van Franeker and Bell 1988), but not for the European storm petrel. Enrichment factors were calculated in comparison to the penta-DE mixture. In preen oil, 43% of the congeners are being accumulated rather than excreted, whilst in feathers, the number increases to 71%.

There are many factors that can influence the burden of persistent organic pollutants. Feathers receive pollutants through the blood supply whilst growing. Once they are fully grown, the blood supply stops and levels receive no internal input from contamination (Jaspers et al. 2004). However, feathers can have their burdens increased by external contamination and the preening of feathers (Jaspers et al. 2008, 2007a; Van Den Brink 1997). Time and sequence of moult is therefore an important factor when quantifying levels in feathers, especially when comparing those with preen oil, which reflect current contamination, due to its regular production. In our study, storm petrels were sampled during the breeding season. The European storm petrel is known to moult their body feathers during incubation up to the winter quarters (Ginn and Melville 1983), meaning that feathers were receiving blood supply along with its contaminants and reflect local contamination. This might explain why levels in feathers were much higher than levels in preen oil in the same individuals as feathers would have the additional burden of blood, preening and external contamination, although the latter has been considered negligible (Jaspers et al. 2007a). Migration is another important factor contributing to variation in contaminant levels (Perkins and Barclay 1997). European storm petrels winter in southern Africa and spend their breeding season in Europe (Robert et al. 1998). Long migrations such as these take a toll into a bird's energy reserves. If at the beginning of a long-haul migration a bird's fat reserves are high, at the end, they are very low and the mobilisation of lipid reserves to attend a bird's demands can increase contaminant concentration, the same way starvation can (Perkins and Barclay 1997). Starving birds are expected to have higher contaminant concentration due to mobilisation of their fat reserves (Barron et al. 1995). Birds from this study were sampled in late August, meaning they were at the end of their breeding season, which was also confirmed by repairing brood patches. By then, it is expected that body mass has been regained to cope with energetic breeding demands. This assumption was supported by data on the mass of the birds sampled, which ranged between 22.1 and 27.4 g (Sanz-Aguilar et al. 2009). Breeding also means that females can transfer contaminants to eggs and alleviate their own burden (Bustnes et al. 2008). In this study, sexing of live birds was not possible; therefore, it is not possible to address sex-specific individual levels of persistent organic pollutants, but it is important to consider that during the breeding season, eggs are a pathway for excretion of such contaminants for female birds. Another factor that could influence the way pollutants are perceived during the breeding season is that some birds might change their diet during this period (Hammer et al. 2016), in order to provide more nutritious and energetic food to their young, perhaps making more use of higher trophic organisms. This difference has been observed in storm petrel populations in the Mediterranean, when compared to the Atlantic populations (Albores-Barajas et al. 2011; D'elbeei and Hemer 1998). Persistent organic pollutants are known to bio-accumulate throughout the food web, making predators more vulnerable to such contaminants (Jones and de Voogt 1999; Walker 1990). The diet of the European storm petrels in the UK consists mainly of zooplankton (52%) and a further 37% on benthic organisms (Albores-Barajas et al. 2011; D'elbeei and

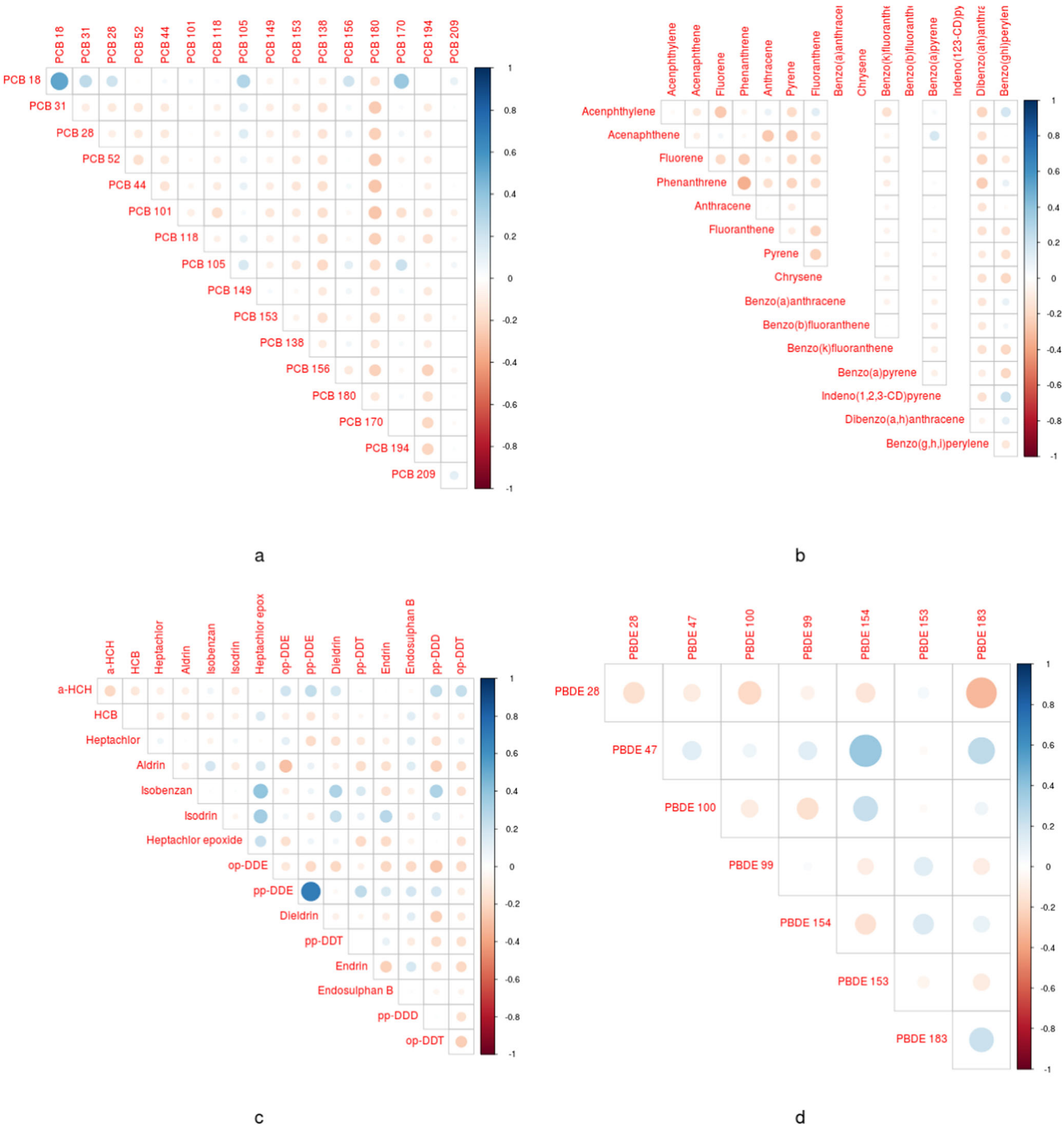

Fig. 2 A Pearson's correlation matrix for PCB (**a**), PAH (**b**), OCP (**c**) and PBDE (**d**). Blue denotes positive correlation; red denotes negative correlation. The stronger the correlation, the darker the colour and the larger the dot. A strong positive correlation is found for pp-DDE and PBDE 47 and − 154

Hemer 1998). This diet favours low trophic level organisms; thus, it is consistent with the relatively low level of contaminants found in this study. In addition, pelagic birds are less exposed to industrialisation and contamination than birds of coastal habitats (Elliott et al. 2005).

Our study, consistent with previous studies (Jaspers et al. 2011, 2008; Yamashita et al. 2007), has demonstrated that the sampling of live birds can be efficient in quantifying contaminant burden in seabird species. This study has quantified and set baseline levels for persistent organic pollutant burden in the European storm petrel. Our statistical analyses have shown none or weak correlation between preen oil and feathers at the individual level. Between groups of contaminants, a moderate and a strong correlation was seen for PCBs and PAHs.

However, congener profiles have shown to differ completely. Thus, choosing a specific matrix can show confounding levels of contaminants. Feathers had higher concentrations of pollutants in comparison to preen oil. Whilst this alone could lead to misleading results when one samples preen oil only, differences in congener profiles have shown that a single matrix might not be sufficient, but sampling different matrices in the same birds might be a more realistic and suitable way to monitor organic pollutants in birds. This factor was not confounded by the time frame feathers were collected as due to moulting dates, feathers reflected local contamination alone. As mentioned before, feather levels are expected to be higher due to the input of preen oil itself on the feathers in the act of preening, along with blood supply and potential external contamination. In this study, levels in feathers were two- to five-fold higher than in preen oil, except in PBDEs, in which similar levels were obtained in both matrices. However, the fact that congener signatures are different between matrices suggests that utilising feathers or preen oil alone is not enough to obtain an accurate picture of contamination in birds. Nevertheless, in most contaminants, congeners were present consistently between feathers and preen oil equally.

Acknowledgements This work was supported by Science Without Borders (CAPES, Brazil, BEX: 1269-13-5). Many thanks to BirdWatch Ireland. Thanks to Niall Keogh and Declan Manley for assistance in the field. Thanks to Danilo Hirota, Killian Coakley and Bill Delee for assistance in the lab.

References

Acampora H, White P, Lyashevska O, O'Connor I (2017) Presence of persistent organic pollutants in a breeding common tern (Sterna hirundo) population in Ireland. Environ Sci Pollut Res 24:13025–13035. https://doi.org/10.1007/s11356-017-8931-7

Ackman RG (1989) Marine biogenic lipids, fats and oils. CRC Press, Boca Raton

Albores-Barajas YV, Riccato F, Fiorin R, Massa B, Torricelli P, Soldatini C (2011) Diet and diving behaviour of European storm petrels Hydrobates pelagicus in the Mediterranean (ssp. melitensis). Bird Study 58:208–212. https://doi.org/10.1080/00063657.2011.560244

Barron MG, Galbraith H, Beltman D (1995) Comparative reproductive and developmental toxicology of PCBs in birds. Comp Biochem Physiol Part C Pharmacol Toxicol Endocrinol 112:1–14

Behrooz RD, Esmaili-Sari A, Ghasempouri SM, Bahramifar N, Covaci A (2009) Organochlorine pesticide and polychlorinated biphenyl residues in feathers of birds from different trophic levels of South-West Iran. Environ Int 35:285–290. https://doi.org/10.1016/j.envint.2008.07.001

Bond AL, Lavers JL (2013) Effectiveness of emetics to study plastic ingestion by Leach's storm-petrels (Oceanodroma leucorhoa). Mar Pollut Bull 70:171–175. https://doi.org/10.1016/j.marpolbul.2013.02.030

Borlakoglu JT, Wilkins JPG, Walker CH, Dils RR (1990a) Polychlorinated biphenyls (PCBs) in fisheating sea birds III: molecular features and metabolic interpretations of PCB isomers and congeners in adipose tissues. Comp Biochem Physiol C Pharmacol Toxicol Endocrinol 97(1):173–177

Borlakoglu JT, Wilkins JPG, Walker CH, Dils RR (1990b) Polychlorinated biphenyls (PCBs) in fisheating sea birds II: molecular features of PCB isomers and congeners in adipose tissue of male and female puffins (*Fratercula arctica*), guillemots (*Uria aalga*), shags (*Phalacrocorax aristotelis*) and cormorants (*Phalacrocorax carbo*) of British and Irish coastal waters. Comp Biochem Physiol C Comp Pharmacol Toxicol Endocrinol 97(1):161–171

Bosveld ATC, van den Berg M (2002) Reproductive failure and endocrine disruption by organohalogens in fish-eating birds. Toxicology 181–182:155–159. https://doi.org/10.1016/S0300-483X(02)00273-1

Brinkman UAT, De Kok A (1981) Halogenated biphenyls, terphenyls, naphthalenes, dibenzodioxins and related products, production, properties and usage. In: Chemical informations, vol 12. Elsevier, Amsterdam

Bustnes JO, Fauchald P, Tveraa T, Helberg M, Skaare JU (2008) The potential impact of environmental variation on the concentrations and ecological effects of pollutants in a marine avian top predator. Environ Int 34:193–201

Chen D, Martin P, Burgess NM, Champoux L, Elliott JE, Forsyth DJ, Idrissi A, Letcher RJ (2013) European starlings (Sturnus vulgaris) suggest that landfills are an important source of bioaccumulative flame retardants to Canadian terrestrial ecosystems. Environ. Sci. Technol. 47:12238–12247

Colabuono FI, Barquete V, Domingues BS, Montone RC (2009) Plastic ingestion by Procellariiformes in Southern Brazil. Mar Pollut Bull 58:93–96. https://doi.org/10.1016/j.marpolbul.2008.08.020

Connan M, Cherel Y, Mayzaud P (2007) Lipids from stomach oil of procellariiform seabirds document the importance of myctophid fish in the Southern Ocean. Limnol Oceanogr 52:2445–2455

D'elbeei J, Hemer G (1998) Diet and foraging behaviour of the British storm petrel Hydrobates pelagicus in the Bay of Biscay during summer. Ardea 86:1–10

Darnerud PO (2008) Brominated flame retardants as possible endocrine disrupters. Int J Androl 31:152–160

Earley JJ (1987) Pesticides and PCBs in seabirds and coastal pollution. Biol Indic Pollut 193:200

Elliott J, Wilson LK, Wakeford B (2005) Polybrominated diphenyl ether trends in eggs of marine and freshwater birds from British Columbia, Canada, 1979-2002. Environ Sci Technol 39:5584–5591

Eng ML, Elliott JE, MacDougall-Shackleton SA, Letcher RJ, Williams TD (2012) Early exposure to 2,2′,4,4′,5-pentabromodiphenyl ether (BDE-99) affects mating behavior of zebra finches. Toxicol Sci 127:269–276. https://doi.org/10.1093/toxsci/kfs076

Espin S, Martinez-Lopez E, Maria-Mojica P, Garcia-Fernandez AJ (2012) Razorbill (Alca torda) feathers as an alternative tool for evaluating exposure to organochlorine pesticides. Ecotoxicology 21:183–190. https://doi.org/10.1007/s10646-011-0777-z

Falkowska L, Reindl AR, Grajewska A, Lewandowska AU (2016) Organochlorine contaminants in the muscle, liver and brain of seabirds (Larus) from the coastal area of the Southern Baltic. Ecotoxicol Environ Saf 133:63–72. https://doi.org/10.1016/j.ecoenv.2016.06.042

van Franeker JA, Bell PJ (1988) Plastic ingestion by petrels breeding in Antarctica. Mar Pollut Bull 19:672–674. https://doi.org/10.1016/0025-326X(88)90388-8

Fromant A, Carravieri A, Bustamante P, Labadie P, Budzinski H, Peluhet L, Churlaud C, Chastel O, Cherel Y (2016) Wide range of metallic and organic contaminants in various tissues of the Antarctic prion, a planktonophagous seabird from the Southern Ocean. Sci Total Environ 544:754–764. https://doi.org/10.1016/j.scitotenv.2015.11.114

Furness RW (1993) Birds as monitors of pollutants. In: Furness RW, Greenwood JJD (eds) Birds as monitors of environmental change. Springer Netherlands, Dordrecht, pp 86–143. https://doi.org/10.1007/978-94-015-1322-7_3

Giesy JP, Ludwig JP, Tillitt DE (1994) Dioxins, dibenzofurans, PCBs and colonial, fish-eating water birds. In: Schecter A (ed) Dioxins and health. Springer US, Boston, pp 249–307. https://doi.org/10.1007/978-1-4899-1462-0_9

Gilbertson MMRD, Morris RD, Hunter RA (1976) Abnormal chicks and PCB residue levels in eggs of colonial birds on the Lower Great Lakes (1971-73). Auk 93:434–442

Ginn HB, Melville DS (1983) Moult in birds (BTO guide). British Trust for Ornithology, Tring

Hammer S, Nager RG, Alonso S, McGill RAR, Furness RW, Dam M (2016) Legacy pollutants are declining in Great Skuas (Stercorarius skua) but remain higher in Faroe Islands than in Scotland. Bull Environ Contam Toxicol 97:184–190. https://doi.org/10.1007/s00128-016-1856-x

Herzke D, Anker-Nilssen T, Nøst TH, Götsch A, Christensen-Dalsgaard S, Langset M, Fangel K, Koelmans AA (2016) Negligible impact of ingested microplastics on tissue concentrations of persistent organic pollutants in Northern Fulmars off Coastal Norway. Environ Sci Technol. https://doi.org/10.1021/acs.est.5b04663

Jaspers V, Dauwe T, Pinxten R, Bervoets L, Blust R, Eens M (2004) The importance of exogenous contamination on heavy metal levels in bird feathers. A field experiment with free-living great tits, Parus major. J Environ Monit 6:356–360

Jaspers VLB, Covaci A, Voorspoels S, Dauwe T, Eens M, Schepens P (2006a) Brominated flame retardants and organochlorine pollutants in aquatic and terrestrial predatory birds of Belgium: levels, patterns, tissue distribution and condition factors. Environ Pollut 139:340–352

Jaspers VLB, Oorspoels S, Covaci A, Eens M (2006b) Can predatory bird feathers be used as a non-destructive biomonitoring tool of organic pollutants? Biol Lett 2:283–285. https://doi.org/10.1098/rsbl.2006.0450

Jaspers VLB, Covaci A, Van den Steen E, Eens M (2007a) Is external contamination with organic pollutants important for concentrations measured in bird feathers? Environ Int 33:766–772. https://doi.org/10.1016/j.envint.2007.02.013

Jaspers VLB, Voorspoels S, Covaci a, Lepoint G, Eens M (2007b) Evaluation of the usefulness of bird feathers as a non-destructive biomonitoring tool for organic pollutants: a comparative and meta-analytical approach. Environ Int 33:328–337. https://doi.org/10.1016/j.envint.2006.11.011

Jaspers VLB, Covaci A, Deleu P, Neels H, Eens M (2008) Preen oil as the main source of external contamination with organic pollutants onto feathers of the common magpie (Pica pica). Environ Int 34:741–748. https://doi.org/10.1016/j.envint.2007.12.002

Jaspers VLB, Rodriguez FS, Boertmann D, Sonne C, Dietz R, Rasmussen LM, Eens M, Covaci A (2011) Body feathers as a potential new biomonitoring tool in raptors: a study on organohalogenated contaminants in different feather types and preen oil of West Greenland white-tailed eagles (Haliaeetus albicilla). Environ Int 37:1349–1356

Jones K, de Voogt P (1999) Persistent organic pollutants (POPs): state of the science. Environ Pollut 100:209–221. https://doi.org/10.1016/S0269-7491(99)00098-6

Jörundsdóttir H, Löfstrand K, Svavarsson J, Bignert A, Bergman Å (2010) Organochlorine compounds and their metabolites in seven Icelandic seabird species—a comparative study. Environ Sci Technol 44:3252–3259. https://doi.org/10.1021/es902812x

Karlsson M, Ericson I, van Bavel B, Jensen J-K, Dam M (2006) Levels of brominated flame retardants in Northern Fulmar (Fulmarus glacialis) eggs from the Faroe Islands. Sci Total Environ 367:840–846

Knight GC, Walker CH (1982) A study of the hepatic microsomal monooxygenase of sea birds and its relationship to organochlorine pollutants. Comp Biochem Physiol Part C Comp Pharmacol 73: 211–221

Koeman JH, Oskamp AAG, Veen J, Brouwer E, Rooth J, Zwart P, vd Broek E, Van Genderen H (1967) Insecticides as a factor in the mortality of the sandwich tern (Sterna sandvicensis). Meded Rijksfac LandbWet Gent 32:841

Kubiak TJ, Harris HJ, Smith LM, Schwartz TR, Stalling DL, Trick JA, Sileo L, Docherty DE, Erdman TC (1989) Microcontaminants and reproductive impairment of the Forster's tern on Green Bay, Lake Michigan-1983. Arch Environ Contam Toxicol 18:706–727. https://doi.org/10.1007/BF01225009

Laffon B, Fraga-Iriso R, Pérez-Cadahía B, Méndez J (2006) Genotoxicity associated to exposure to prestige oil during autopsies and cleaning of oil-contaminated birds. Food Chem Toxicol 44:1714–1723

Ludwig JP, Colborn TL, Giesvi JP (1996) Persistent synthetic chlorinated hydrocarbons in albatross tissue samples from midway atoll. Environ Toxicol Chem 15:1793–1800

MacRae JD, Hall KJ (1998) Biodegradation of polycyclic aromatic hydrocarbons (PAH) in marine sediment under denitrifying conditions. Water Sci Technol 38:177–185

Mallory ML, Braune BM (2012) Tracking contaminants in seabirds of Arctic Canada: temporal and spatial insights. Mar Pollut Bull 64: 1475–1484

Matthies M, Solomon K, Vighi M, Gilman A, Tarazona JV (2016) The origin and evolution of assessment criteria for persistent, bioaccumulative and toxic (PBT) chemicals and persistent organic pollutants (POPs). Environ Sci Process Impacts 18:1114–1128

Meador JP, Stein JE, Reichert WL, Varanasi U (1995)Bioaccumulation of polycyclic aromatic hydrocarbons by marine organisms. In: Ware GW (ed) Reviews of environmental contamination and toxicology, vol 143. Springer, New York

Megson D, Brown TA, Johnson GW, O'Sullivan G, Bicknell AWJ, Votier SC, Lohan MC, Comber S, Kalin R, Worsfold PJ (2014) Identifying the provenance of Leach's storm petrels in the North Atlantic using polychlorinated biphenyl signatures derived from comprehensive two-dimensional gas chromatography with time-of-flight mass spectrometry. Chemosphere 114:195–202. https://doi.org/10.1016/j.chemosphere.2014.04.061

Miller A, Elliott JE, Elliott KH, Guigueno MF, Wilson LK, Lee S, Idrissi A (2014) Spatial and temporal trends in brominated flame retardants in seabirds from the Pacific coast of Canada. Environ Pollut 195:48–55. https://doi.org/10.1016/j.envpol.2014.08.009

Mitchell PI, Newton SF, Ratcliffe Norman, Dunn TE (Eds) (2004) Seabird populations of Britain and Ireland: results of the seabird 2000 census (1998–2002). T and A.D. Poyser, London

Moore NW, Tatton J (1965) Organochlorine insecticide residues in the eggs of sea birds. Nature 207:42–43. https://doi.org/10.1038/207042a0

Nfon E, Cousins IT, Broman D (2008) Biomagnification of organic pollutants in benthic and pelagic marine food chains from the Baltic Sea. Sci Total Environ 397:190–204

Perkins CR, Barclay JS (1997) Accumulation and mobilization of organochlorine contaminants in wintering greater scaup. J Wildl Manag 61:444–449

Perugini M, Visciano P, Giammarino A, Manera M, Di Nardo W, Amorena M (2007) Polycyclic aromatic hydrocarbons in marine organisms from the Adriatic Sea, Italy. Chemosphere 66:1904–1910

R Core Team (2015) A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna

Ricca MA, Miles AK, Anthony RG (2008) Sources of organochlorine contaminants and mercury in seabirds from the Aleutian archipelago of Alaska: inferences from spatial and trophic variation. Sci Total Environ 406:308–323

Thomas RJ, Medeiros RJ, Pollard AL (2006) Evidence for nocturnal inter-tidal foraging by European Storm-petrels *Hydrobates pelagicus* during migration. Atlantic Seabirds 8(1/2):87–96

Roscales JL, Gonzalez-Solis J, Calabuig P, Jimenez B (2011a) Interspecies and spatial trends in polycyclic aromatic hydrocarbons

(PAHs) in Atlantic and Mediterranean pelagic seabirds. Environ Pollut 159:2899–2905. https://doi.org/10.1016/j.envpol.2011.04.034

Roscales JL, González-Solís J, Muñoz-Arnanz J, Jiménez B (2011b) Geographic and trophic patterns of OCs in pelagic seabirds from the NE Atlantic and the Mediterranean: a multi-species/multi-locality approach. Chemosphere 85:432–440

Roscales JL, González-Solís J, Zango L, Ryan PG, Jiménez B (2016) Latitudinal exposure to DDTs, HCB, PCBs, PBDEs and DP in giant petrels (Macronectes spp.) across the Southern Ocean. Environ Res 148:285–294. https://doi.org/10.1016/j.envres.2016.04.005

Ryan PG (2015) How quickly do albatrosses and petrels digest plastic particles? Environ Pollut 207:438–440. https://doi.org/10.1016/j.envpol.2015.08.005

Safe S, Hutzinger O (1984) Polychlorinated biphenyls (PCBs) and polybrominated biphenyls (PBBs): biochemistry, toxicology, and mechanism of action. CRC Crit Rev Toxicol 13:319–395

Sagerup K, Helgason LB, Polder A, Strøm H, Josefsen TD, Skåre JU, Gabrielsen GW (2009) Persistent organic pollutants and mercury in dead and dying glaucous gulls (Larus hyperboreus) at Bjørnøya (Svalbard). Sci Total Environ 407:6009–6016

Sanz-Aguilar A, Massa B, Lo Valvo F, Oro D, Minguez E, Tavecchia G (2009) Contrasting age-specific recruitment and survival at different spatial scales: a case study with the European storm petrel. Ecography (Cop) 32:637–646

Stockholm Convention (2001) Stockholm Convention on Persistent Organic Pollutants. p. http://chm.pops.int/

Stogiannidis E, Laane R (2015) Source characterization of polycyclic aromatic hydrocarbons byusing their molecular indices: an overview of possibilities. In: Reviews of environmental contamination andtoxicology. Springer International Publishing, Switzerland, pp 49–133

Tanaka K, Takada H, Yamashita R, Mizukawa K, Fukuwaka M, Watanuki Y (2013) Accumulation of plastic-derived chemicals in tissues of seabirds ingesting marine plastics. Mar Pollut Bull 69: 219–222

Tanaka K, Takada H, Yamashita R, Mizukawa K, Fukuwaka M, Watanuki Y (2015) Facilitated leaching of additive-derived PBDEs from plastic by seabirds' stomach oil and accumulation in tissues. Environ Sci Technol 150901174241000:11799–11807. https://doi.org/10.1021/acs.est.5b01376

Trumble SJ, Robinson EM, Noren SR, Usenko S, Davis J, Kanatous SB (2012) Assessment of legacy and emerging persistent organic pollutants in Weddell seal tissue (Leptonychotes weddellii) near McMurdo Sound, Antarctica. Sci Total Environ 439:275–283. https://doi.org/10.1016/j.scitotenv.2012.09.018

Van Den Brink NW (1997) Directed transport of volatile organochlorine pollutants to polar regions: the effect of the contamination pattern of Antarctic seabirds. Sci Total Environ 198:43–50. https://doi.org/10.1016/S0048-9697(97)05440-5

Van Den Brink NW, Bosveld ATC (2001) PCB concentrations and metabolism patterns in common terns (Sterna hirundo) from different breeding colonies in the Netherlands. Mar Pollut Bull 42:280–285. https://doi.org/10.1016/S0025-326X(00)00151-X

Van den Steen E, Dauwe T, Covaci A, Jaspers VLB, Pinxten R, Eens M (2006) Within-and among-clutch variation of organohalogenated contaminants in eggs of great tits (Parus major). Environ Pollut 144:355–359

Van Franeker JA, Blaize C, Danielsen J, Fairclough K, Gollan J, Guse N, Hansen PL, Heubeck M, Jensen JK, Le Guillou G, Olsen B, Olsen KO, Pedersen J, Stienen EWM, Turner DM (2011) Monitoring plastic ingestion by the northern fulmar Fulmarus glacialis in the North Sea. Environ Pollut 159:2609–2615. https://doi.org/10.1016/j.envpol.2011.06.008

Walker CH (1990) Persistent pollutants in fish-eating sea birds—bioaccumulation, metabolism and effects. Aquat Toxicol 17:293–324. https://doi.org/10.1016/0166-445X(90)90014-G

Wan Y, Jin X, Hu J, Jin F (2007) Trophic dilution of polycyclic aromatic hydrocarbons (PAHs) in a marine food web from Bohai Bay, North China. Environ Sci Technol 41:3109–3114

Wang J, Caccamise SAL, Woodward LA, Li QX (2015) Polychlorinated biphenyls in the plasma and preen oil of black-footed albatross (Diomedea nigripes) chicks and adults on midway atoll, North Pacific Ocean. PLoS One 10:e0123041. https://doi.org/10.1371/journal.pone.0123041

Webster L, McIntosh AD, Moffat CF, Dalgarno EJ, Brown NA, Fryer RJ (2000) Analysis of sediments from Shetland Island voes for polycyclic aromatic hydrocarbons, steranes and triterpanes. J Environ Monit 2:29–38

White P, McHugh B, Poole R, McGovern E, White J, Behan P, Foley B, Covaci A (2014) Application of congener based multi-matrix profiling techniques to identify potential PCDD/F sources in environmental samples from the Burrishoole catchment in the west of Ireland. Environ Pollut 184:449–456. https://doi.org/10.1016/j.envpol.2013.09.026

Winter V, Williams TD, Elliott JE (2013) A three-generational study of in ovo exposure to PBDE-99 in the zebra finch. Environ Toxicol Chem 32:562–568

Yamashita R, Takada H, Murakami M, Fukuwaka MA, Watanuki Y (2007) Evaluation of noninvasive approach for monitoring PCB pollution of seabirds using preen gland oil. Environ. Sci. Technol. 41:4901–4906. https://doi.org/10.1021/es0701863