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Preface

Northern fulmars are seabirds which feed exclusively at sea, and as such, they are useful indicators of ocean health. Marine plastic pollution is an ever-increasing and global issue that affects the northern fulmar as they are frequently found to have ingested plastic. In this report we investigate whether the amount of ingested plastic affects the concentration of certain plastic-adsorbed toxicants in their tissues. Marine plastic pollution is a field of utmost importance. It is our hope that this continues to be an area which receives increased attention in order to elucidate the potential harmful effects plastics have on the northern fulmar and ocean health, in general.

This study was made possible by the financial support from the Nordic Council of Ministers. We would also like to take this opportunity to express our gratitude to Jan van Franeker (IMARES), Dorte Herzke (NILU) and Tycho Anker-Nilssen (NINA) for contributing with datasets for this report.

Summary

Marine plastic pollution is a widespread and increasing problem. Due to the chemical and physical properties of plastic, it tends to persist in the marine environment over long periods of time where it has the potential to harm fauna and flora. Among the many threats posed by plastic, ingestion of plastic is frequently observed in a variety of species. Seabirds, and especially the Procellariiformes, are commonly found with high levels of ingested plastics. Apart from the physical dangers of ingested plastics (e.g. internal injuries and lodging in the digestive system), there is concern that the chemicals added to and adsorbed to the plastic could be absorbed by the bird and exert toxic effects. The aim of this study was to investigate this by expanding upon and comparing two datasets on northern fulmars (*Fulmarus glacialis*) in relation to the contaminant concentration in selected tissues and ingested plastics.

Fulmars from the Faroe Islands were all bycatch victims from longline fisheries caught in 2011 and fulmars from Norway were predominantly bycatch from fisheries in 2012 and 2013, supplemented with a few individuals found beached. Upon dissection, plastic content in the stomach was quantified and tissues (liver for the Faroese fulmars and muscle and liver for the Norwegian fulmars) were frozen for subsequent chemical analyses. Tissues were analysed for a suite of persistent organic pollutants: polychlorinated biphenyls, polybrominated diphenyl ethers, perfluoroalkyl and polyfluoroalkyl substances, metabolites, organophosphate flame retardants, dichlorodiphenyltrichloroethane and other pesticides. The data were then analysed statistically to examine whether there were associations between the level of ingested plastic and contaminant concentration in the fulmars, in addition to comparing contaminant burdens between Faroese and Norwegian fulmars.

After correcting for the multiple testing, there were no statistically significant differences in contaminant concentrations between the various plastic ingestion groups. The contaminant concentrations in liver in Faroese and Norwegian fulmars were not significantly different after correcting for the multiple testing. Thus, it appears that ingested plastic is not a significant route of exposure to the adsorbed contaminants analysed herein for the fulmar.

1. Introduction

Over five trillion pieces of plastic pollute the surface of the world's oceans according to a recent estimate (Eriksen *et al*., 2014) and marine plastic pollution is recognized as an area of global concern (Thompson *et al*., 2009; UNEP 2011; Bergmann *et al*., 2015). The majority of marine plastics are consumer products (e.g. food packaging, cigarette filters, bottles, bags, commercial and recreational fishing gear) and manufacturing pellets (Derraik, 2002). The common denominator for each piece of marine plastic is that it was the improper handling by humans, either accidentally or deliberately, which led to it ending up in the ocean (Sheavly and Register, 2007).

Plastics are synthetic polymers and are composed of a variety of different chemical classes such as polyethylene, polypropylene, polystyrene, polyethylene terephthalate and polyvinyl chloride (Andrady, 2011). Although there is some chemical and biological degradation of the plastics, the rates of degradation are generally low and the plastics are persistent in the environment (Shah *et al*., 2008; Andrady, 2011). Nevertheless, the plastics break into smaller fragments as a result of mechanical weathering and degradation, eventually forming microplastics. However, microplastics can also enter the ocean from primary sources, such as airblasting media, cosmetics, manufacturing pellets and waste from plastic production plants (Moore, 2008; Fendall and Sewell, 2009; Andrady, 2011). The field is lacking a formal classification system, but it is generally agreed upon that microplastics are defined as those with an upper diameter/length less than 5 mm (Hidalgo-Ruz *et al*., 2012).

All plastics, from micro to macro-plastics, are potentially harmful to marine fauna. Wildlife become entangled in six-pack rings and abandoned fishing nets, for instance, which frequently lead to strangulation/drowning, wounds and an impaired ability to forage and/or avoid predation. Marine plastics also act as a transport vector for invasive, alien species which may disrupt local ecosystems. Furthermore, plastics are ingested by a wide range of marine fauna from sea turtles to seabirds (Barnes, 2002; Derraik, 2002; Moore, 2008; Gregory, 2009; Tourinho *et al*., 2010; Kühn *et al*., 2015). Indeed, several studies have demonstrated the susceptibility of seabirds to ingest plastics (e.g. Moser and Lee, 1992;

Robards *et al*., 1995; van Franeker *et al*., 2011; Trevail *et al*., 2015). Several studies also note the high frequency with which Procellariiformes (which includes albatrosses, shearwaters and petrels) are found to have ingested plastics compared to other seabird species, likely due to a combination of their inability to regurgitate hard materials, surface feeding behaviour and how they mistake plastics for prey (Moser and Lee, 1992; Robards *et al*., 1995; Tourinho *et al*., 2010). Potential consequences of plastic ingestion are internal injuries, ulcers, a false sense of satiation and subsequent emaciation and dehydration, and the risk of the plastic lodging in the digestive system (Pettit *et al*., 1981; Azzarello and Van Vleet, 1987; Fry *et al*., 1987; Pierce *et al*., 2004). Additionally, toxic chemicals such as the persistent organic pollutants (POPs) are known to adsorb to plastics (Carpenter *et al*., 1972; Mato *et al*., 2001; Rios *et al*., 2007; Teuten *et al*., 2009; Rochman *et al*., 2013). There are some indications that the POPs on ingested plastic become bioavailable and absorbed by seabirds (Ryan *et al*., 1988; Tanaka *et al*., 2013). However, there has also been indications that the contaminant exposure from ingested plastics is negligible compared to exposure from prey (Herzke *et al*., 2016), thus highlighting the need for further research in this area.

The northern fulmar (*Fulmarus glacialis*, hereafter fulmar) is a long-lived fulmarine petrel species breeding in the North Atlantic and North Pacific Oceans. The fulmar feeds exclusively at sea, where its diet consists mainly of crustaceans, cephalopods, fish and offal from fishing vessels (Mehlum and Gabrielsen, 1993; Anker-Nilssen *et al*., 2000). In addition, studies have found plastic in fulmar stomachs, as mentioned above (Moser and Lee, 1992; van Franeker *et al*., 2011; Trevail *et al*., 2015; van Franeker and Law, 2015). The Oslo-Paris Conventions (OSPAR) on the protection of the marine environment in the North-East Atlantic have created a set of ecological quality objectives (EcoQO) to aid in this endeavour. For the fulmar the EcoQO states that less than 10% of beached fulmars collected over a 4–5 year period should have stomach plastic content exceeding 0.1 grams (Heslenfeld *et al*., 2009). This goal, however, has proved hard to achieve (OSPAR, 2011; van Franeker *et al*., 2011; van Franeker and Law, 2015).

Driven by a receding sea ice cover, geopolitics and economics, an increase in maritime traffic in the Arctic is expected in the future (Brigham, 2011; Kerr, 2012). Despite the ban of disposing waste to sea by MARPOL Annex V (International Maritime Organization), ships are a considerable source of marine plastics and an increase in marine plastic pollution is anticipated to accompany the rise in maritime traffic (Ryan *et al.*, 2009; van Franeker *et al.*, 2011). Considering that arctic fulmars already exceed the EcoQO, it is concerning and adds urgency to the research into possible detrimental effects of plastic ingestion by fulmars.

1.1 Aim of Study

This report aims to statistically analyse two datasets on plastic ingestion by northern fulmars caught in the Faroe Islands and Norway and investigate the tissue concentrations of POPs. When possible, the contaminant levels in the Faroese and Norwegian fulmars are compared.

2. Materials and methods

This report expands upon previous work by Trevail (2014) and Herzke *et al*. (2016). For the ingested plastics, all data herein has been published previously in van Franeker *et al*. (2013; as part of a larger dataset) and Herzke *et al*. (2016). Trevail (2014) and Herzke *et al*. (2016) report some contaminant data, but further analyses have been performed on the tissue samples and the data on perfluoroalkyl and polyfluoroalkyl substances and metabolites are reported here for the first time.

The materials and methods used by the authors of the different datasets are described in detail in van Franeker *et al*. (2013), Trevail (2014) and Herzke *et al*. (2016). However, they will be briefly outlined here.

For the Faroe Island dataset, first published in van Franeker *et al*. (2013) and Trevail (2014), the northern fulmars (*Fulmarus glacialis*) were all victims of long-line fisheries, caught in 2011. Dissections, tissue sampling and stomach analyses were conducted at the IMARES lab in the Netherlands following the protocol outlined in van Franeker (2004). Out of 200 fulmars caught, 27 were chosen for liver tissue analysis at the Norwegian Institute for Air Research (NILU), Tromsø, Norway and are the birds used for this study. The sub-samples were chosen on the basis of birds with no, moderate (0.03 to 0.08 g; 2–12 pieces) and high (0.27 to 1.42 g; 7–152 pieces) levels of ingested plastics.

The fulmars used in the Norway dataset were predominantly caught on long-lines in Northern Norway in 2012 and 2013 (n = 72) while three individuals were found beached in Rogaland county, as reported in Herzke *et al*. (2016). Out of the 75 fulmars, 30 were chosen for chemical analyses at NILU, Tromsø and are the birds used for this study. Again, the subsample was divided into birds with no, moderate (0.01 to 0.21 g; 1–14 pieces) and high (0.11 to 0.59 g; 15–106 pieces) levels of ingested plastics. Muscle tissue was analysed for all three groups and for the "high" group liver tissue was also analysed. The fulmars were dissected at the Norwegian Institute for Nature Research, Trondheim, Norway and results published in Herzke *et al*. (2016).

2.1 Analyses of contaminants

The samples were analysed for polychlorinated biphenyls (PCBs), metabolites (Faroe dataset only), polybrominated diphenyl ethers (PBDEs), dichlorodiphenyltrichloroethanes (DDTs), pesticides (Faroe dataset only), perfluoroalkyl and polyfluoroalkyl substances (PFASs), and organophosphate flame retardants (OPFRs; Faroe dataset only). A complete list of analytes is provided in Appendix A.

All the classes of chemicals listed above can sorb to marine plastic from the ambient seawater (Mato *et al*., 2001; Teuten *et al*., 2009; Rios *et al*., 2010; Rochman *et al*., 2013; Llorca *et al*., 2014). Flame retardants, however, are also commonly added to the plastic in the manufacturing process. Thus, PBDEs and OPFRs will be present in plastic as a result of both adsorption and intentional addition (Alaee *et al*., 2003; Talsness, 2008). Similarly, due to the desirable physicochemical properties of PFASs, they are also frequently added to plastic products (Lang *et al*., 2016).

2.1.1 Chemical analyses

Two grams of tissue (liver or muscle) was homogenized with sodium sulphate (Merck, Darmstadt, Germany) and frozen overnight. The following day internal standards were added to each sample. Then the homogenates were extracted three times using 50 mL cyclohexane:acetone (the ratio differs in the two studies) for one hour. In total, the extraction was achieved with 150 mL over 3 hours. The extract was then concentrated to 0.5 mL/1 mL before running the samples on a gel permeation chromatography (GPC) system (Waters ® Envirogel™ GPC Cleanup column). Following the GPC cleanup, 50 µL isooctane was added to the samples before the final cleanup step. The Florisil® Cleanup ensured the removal of lipids from the samples. Lastly, the samples were concentrated to approximately 200 µL using nitrogen gas (N_2, 99% purity, AGA, Oslo, Norway) before 20 µL of recovery standard was added to each sample.

In addition to running the tissue samples, a laboratory blank and standard reference material (SRM 1945; National Institute of Standards and Technology, Gaithersburg, USA) were analysed concurrently as part of the quality control.

2.2 Statistics

Within each dataset, fulmars from the "absent", "moderate" and "high" groups were compared to examine whether level of ingested plastic affects tissue concentrations of selected contaminants. Contaminant burden in liver tissue for fulmars from the "high" groups in both datasets was also compared to examine if there was a regional difference.

All statistics were performed in Excel (2013, Microsoft Corp.) and SigmaPlot (version 13.0.0, 2014, Systat Software Inc.). Excel was used for descriptive statistics and all other tests were performed in SigmaPlot.

A statistical significance level of $p < 0.05$ was set. When testing for differences in contaminant load between ≥ 3 groups, data that passed the Shapiro-Wilk test for normality and the Brown-Forsythe test for homogeneity of variance were analysed using one-way ANOVA (analysis of variance) and Tukey's post hoc test. If the data failed the Shapiro-Wilk test and/or the Brown-Forsythe test, it was analysed using Kruskal-Wallis one-way analysis of variance on ranks (hereafter Kruskal-Wallis) and Dunn's post hoc test. When comparing just two groups, data that passed the Shapiro-Wilk and Brown-Forsythe tests were analysed using the Student's t-test and data that failed the assumptions of normality and equal variance were analysed using the Mann-Whitney Rank Sum test. Data was not transformed if it failed the Shapiro-Wilk and/or the Brown-Forsythe tests. The Holm-Šidàk correction was applied to account for the multiple comparisons and the consequent increased risk of committing a Type I error.

2.2.1 Data below the limit of detection

A minimum of 70% of the individuals in each group had to have values above the limit of detection (LOD) for each contaminant in order to be included in statistical analyses. Please refer to Appendix B for a complete list of contaminants eliminated due to this and the list of LOD values for each contaminant.

If less than 30% of the samples were below LOD, those that were below were replaced with a value of zero in order to incorporate them into the analyses. Substitution with zero was done as that was the method chosen by Trevail (2014).

3. Results

3.1 Faroe Island dataset

The reader is referred to Trevail (2014) and van Franeker *et al*. (2013) for the original publication of the data.

3.1.1 Plastics

The mean, standard deviation (SD), median and range of the plastic ingested by the "absent", "moderate" and "high" groups are presented in Table 1, as well as statistics for all 27 fulmars.

3.1.2 Contaminants

The mean, SD, median and range for the analysed contaminants are summarized in Tables 2, 3, 4 and 5. The tables are segmented for each group ("absent", "moderate", and "high") to assist in the comparison. For PFASs, only the "absent" and "high" groups were sampled.

After applying the Holm-Šidàk correction, there were no statistically significant differences in contaminant concentrations between the various groups.

Table 1: Summary of plastic ingestion data for the subset of northern fulmars (Fulmarus glacialis) caught near the Faroe Islands in 2011 that were sampled for this contaminant study

Group		Mean ± SD	Median	Range	*n*
Absent	Mass (g)	0	0	0	9
	Pieces	0	0	0	9
Moderate	Mass (g)	0.06 ± 0.02	0.05	0.03–0.08	9
	Pieces	4.78 ± 3.07	4	2–12	9
High	Mass (g)	0.63 ± 0.35	0.56	0.28–1.42	9
	Pieces	36.8 ± 44.7	26	7–152	9
All	Mass (g)	0.23 ± 0.35	0.05	0.00–1.42	27
	Pieces	13.9 ± 29.9	4	0–152	27

Note: The data are summarized as mean ± standard deviation (SD), median and range. *n* denotes the sample size.

Table 2: Summary of the concentrations (ng/g ww liver) of perfluoroalkyl and polyfluoroalkyl substances (PFASs) detected in liver tissue from the subset of northern fulmars (Fulmarus glacialis) caught near the Faroe Islands in 2011 that were sampled for this contaminant study

PFASs (ng/g)	Mean ± SD	Absent Median	Range	*n*	Mean ± SD	High Median	Range	*n*
PFOS	9.13 ± 4.63	8.15	2.96–15.9	9	10.5 ± 3.24	9.83	7.44–18.0	9
PFHpA	0.13 ± 0.10	0.10	<LOD–0.27	9	0.12 ± 0.09	0.09	0.02–0.25	9
PFNA	0.85 ± 0.56	0.98	0.22–1.58	9	1.13 ± 0.70	0.80	0.46–2.69	9
PFDcA	0.96 ± 0.73	1.03	0.153–2.29	9	1.26 ± 0.54	0.98	0.65–2.26	9
PFUnDA	3.86 ± 2.29	3.38	1.23–7.86	9	4.47 ± 2.00	3.68	2.57–8.65	9
PFTrDA	4.20 ± 2.02	4.15	1.70–6.69	9	4.47 ± 1.84	4.02	2.50–7.61	9
∑PFAS	19.1 ± 9.51	17.3	6.98–32.0	9	21.9 ± 8.06	17.1	15.2–39.4	9

Note: The data are presented as mean ± standard deviation (SD), median and range. *n* denotes the sample size. No statistically significant differences were detected between fulmars with no ingested plastic (termed "absent") and a high plastic load in their stomachs (termed "high").

Table 3: Summary of the concentrations (ng/g ww liver) of metabolites detected in liver tissue from the subset of northern fulmars (Fulmarus glacialis) caught near the Faroe Islands in 2011 that were sampled for this contaminant study

Metabolites (ng/g)	Mean ± SD	Absent Median	Range	*n*	Mean ± SD	High Median	Range	*n*
PCP	0.32 ± 0.29	0.22	0.08–0.94	8	0.14 ± 0.07	0.13	0.07–0.29	8
3-OH-PCB153	0.52 ± 0.54	0.44	<LOD–1.62	8	1.04 ± 0.71	1.03	0.20–2.19	8
4-OH-PCB146	2.16 ± 1.04	2.06	0.94–3.80	8	4.53 ± 4.23	3.38	0.58–13.4	8
3-OH-PCB138	0.19 ± 0.14	0.16	<LOD–0.41	8	0.39 ± 0.23	0.45	0.06–0.67	8
4-OH-PCB187	2.91 ± 1.80	2.52	1.11–5.55	8	5.36 ± 7.01	3.13	0.53–22.1	8
4-OH-PCB172	0.30 ± 0.25	0.26	<LOD–0.82	8	0.47 ± 0.61	0.24	<LOD–1.76	8
4'-OH-PCB193	0.20 ± 0.29	0.12	<LOD–0.90	8	0.28 ± 0.48	0.10	<LOD–1.45	8
3-MeSO-PCB91	0.01 ± 0.01	0.01	0.003–0.02	9	0.02 ± 0.03	0.02	0.003–0.09	9
3-MeSO-PCB101	0.36 ± 0.21	0.34	0.10–0.69	9	1.44 ± 2.01	0.46	0.03–6.19	9
4-MeSO-PCB101	0.21 ± 0.11	0.22	0.08–0.41	9	0.60 ± 0.64	0.26	0.03–1.58	9
3-MeSO-PCB87	0.16 ± 0.09	0.14	0.03–0.31	9	0.44 ± 0.74	0.15	<LOD–2.31	9
4-MeSO-PCB110	0.07 ± 0.04	0.06	0.02–0.15	9	0.15 ± 0.22	0.08	<LOD–0.72	9
3-MeSO-PCB149	0.11 ± 0.06	0.12	0.03–0.20	9	0.35 ± 0.61	0.14	0.02–1.96	9
4-MeSO-PCB149	0.11 ± 0.05	0.11	0.05–0.19	9	0.27 ± 0.28	0.22	0.03–0.98	9
4-MeSO-PCB132	0.04 ± 0.02	0.04	0.02–0.07	9	0.07 ± 0.05	0.06	0.01–0.17	9
3-MeSO-PCB141	0.07 ± 0.05	0.07	0.02–0.16	9	0.19 ± 0.35	0.07	0.01–1.11	9
4-MeSO-PCB141	0.09 ± 0.05	0.10	0.03–0.16	9	0.11 ± 0.09	0.10	0.01–0.30	9
4-MeSO-PCB174	0.02 ± 0.01	0.02	0.01–0.03	9	0.03 ± 0.03	0.02	<LOD–0.10	9

Note: The data are presented as mean ± standard deviation (SD), median and range. *n* denotes the sample size.
No statistically significant differences were detected between fulmars with no ingested plastic (termed "absent") and a high plastic load in their stomachs (termed "high").

Table 4: Summary of the concentrations (ng/g ww liver) of polychlorinated biphenyls (PCBs) and polybrominated diphenyl ethers (PBDEs) detected in liver tissue from the subset of northern fulmars (Fulmarus glacialis) caught near the Faroe Islands in 2011 that were sampled for this contaminant study

Chemical (ng/g)	Mean ± SD	Absent Median	Range	*n*	Mean ± SD	Moderate Median	Range	*n*	Mean ± SD	High Median	Range	*n*
PCB-28	0.71 ± 0.29	0.70	0.34–1.34	9	0.88 ± 0.46	0.73	0.41–1.73	8	0.78 ± 0.36	0.81	0.19–1.26	9
PCB-52	0.18 ± 0.15	0.13	0.04–0.53	9	0.51 ± 0.63	0.15	0.02–1.59	8	0.22 ± 0.26	0.10	<LOD–0.72	9
PCB-99	30.3 ± 13.1	30.5	10.7–53.0	9	23.3 ± 15.7	14.8	9.22–46.0	8	50.6 ± 59.3	34.3	3.69–199	9
PCB-101	1.07 ± 0.79	0.80	0.20–2.72	9	0.66 ± 0.73	0.398	0.07–2.22	8	1.21 ± 1.52	0.38	<LOD–4.10	9
PCB-105	19.6 ± 7.50	16.9	8.35–32.2	9	25.4 ± 15.3	17.2	11.6–50.3	8	21.2 ± 18.6	18.2	3.10–63.6	9
PCB-118	56.0 ± 24.1	49.3	23.2–93.4	9	69.0 ± 38.9	49.7	33.9–123	8	61.4 ± 57.5	49.7	7.17–196	9
PCB-138	118 ± 49.1	109	48.5–219	9	114 ± 72.5	77.9	41.9–213	8	173 ± 160	128	15.7–544	9
PCB-153	332 ± 125	327	150–523	9	265 ± 146	205	146–525	8	343 ± 250	267	41.9–806	9
PCB-180	116 ± 30.3	119	69.3–157	9	158 ± 102	111	77.4–348	8	102 ± 55.1	92.8	25.4–177	9
∑PCB	674 ± 243	661	320–1049	9	656 ± 377	483	331–1268	8	753 ± 583	594	97.3–1989	9
PBDE-47	0.75 ± 0.56	0.44	0.27–1.97	9	0.52 ± 0.38	0.59	0.06–1.10	8	0.58 ± 0.56	0.32	0.11–1.70	9
PBDE-153	0.31 ± 0.18	0.32	<LOD–0.60	9	0.35 ± 0.27	0.24	<LOD–0.74	8	0.77 ± 0.74	0.63	<LOD–2.47	9
∑PBDE	1.06 ± 0.60	0.80	0.50–2.29	9	1.59 ± 0.91	1.91	0.44–2.86	8	1.35 ± 0.92	1.63	0.19–2.83	9

Note: The data are presented as mean ± standard deviation (SD), median and range. *n* denotes the sample size.
No statistically significant differences were detected between fulmars with no ingested plastic (termed “absent”), a moderate plastic load (termed “moderate”) and a high plastic load in their stomachs (termed “high”).

Table 5: Summary of the concentrations (ng/g ww liver) of dichlorodiphenyltrichloroethane (DDT) and other pesticides detected in liver tissue from the subset of northern fulmars (Fulmarus glacialis) caught near the Faroe Islands in 2011 sampled for this contaminant study

Chemical (ng/g)	Mean ± SD	Absent Median	Range	*n*	Mean ± SD	Moderate Median	Range	*n*	Mean ± SD	High Median	Range	*n*
p,p'-DDE	346 ± 195	322	138–804	9	286 ± 224	185	94.9–724	8	390 ± 255	406	32.8–812	9
p,p'-DDD	5.71 ± 3.10	5.87	2.31–12.5	9	14.0 ± 14.5	8.31	2.63–38.9	8	6.82 ± 5.63	5.91	1.02–18.8	9
∑DDT	352 ± 195	335	140–811	9	300 ± 236	191	101–759	8	396 ± 258	412	34.2–817	9
HCB	22.7 ± 5.22	24.1	13.7–30.6	9	25.6 ± 8.14	23.1	18.5–39.8	8	24.4 ± 9.89	27.3	6.68–40.0	9
Oxy-chlordane	173 ± 57.0	183	95.3–262	9	232 ± 91.0	223	105–385	8	210 ± 149	165	37.9–482	9
trans-chlordane	1.60 ± 0.90	1.34	0.65–3.29	9	1.80 ± 0.79	1.40	1.12–3.13	8	2.48 ± 2.26	1.80	0.27–7.77	9
trans-nonachlor	6.73 ± 4.76	5.44	2.00–15.3	9	9.08 ± 6.52	8.59	1.54–18.6	8	8.17 ± 7.78	5.46	1.08–24.1	9
cis-nonachlor	0.75 ± 0.56	0.57	0.17–1.96	9	0.71 ± 0.56	0.72	<LOD–1.49	8	0.60 ± 0.62	0.26	0.10–1.72	9
Mirex	30.5 ± 13.1	29.9	14.9–60.9	9	35.0 ± 22.4	22.8	17.0–70.3	8	24.4 ± 15.4	20.1	5.44–46.8	9

Note: The data are presented as mean ± standard deviation (SD), median and range. *n* denotes the sample size.
No statistically significant differences were detected between fulmars with no ingested plastic (termed "absent"), a moderate plastic load (termed "moderate") and a high plastic load in their stomachs (termed "high").

3.2 Norway dataset

The reader is referred to Herzke *et al*. (2016) for the original publication of the data.

3.2.1 Plastics

The mean, standard deviation (SD), median and range of the plastic ingested by the "absent", "moderate" and "high" groups are presented in Table 6, as well as statistics for all 30 fulmars.

Table 6: Summary of plastic ingestion data for the subset of northern fulmars (Fulmarus glacialis) caught near Norway in 2012 and 2013 that were sampled for this contaminant study

Group		Mean ± SD	Median	Range	*n*
Absent	Mass (g)	0	0	0	9
	Pieces	0	0	0	9
Moderate	Mass (g)	0.079 ± 0.070	0.062	0.009–0.213	10
	Pieces	5.80 ± 4.64	4.5	1–14	10
High	Mass (g)	0.313 ± 0.182	0.224	0.115–0.593	11
	Pieces	40.6 ± 31.9	24	15–106	11
All	Mass (g)	0.136 ± 0.176	0.078	0.00–0.593	30
	Pieces	17.4 ± 26.8	9	0–106	30

Note: The data are summarized as mean ± standard deviation (SD), median and range. *n* denotes the sample size.

3.2.2 Contaminants

The mean, SD, median and range for the analysed contaminants are summarized in Tables 7, 8, 9 and 10. The tables are segmented for each group ("absent", "moderate", and "high") to assist in the comparison. Please note that for these birds, muscle tissue was analysed for all three groups and, in addition, liver of the "high" group. PFASs were analysed in liver tissue of all three groups.

After applying the Holm-Šidàk correction, there were no statistically significant differences in contaminant concentrations between the various groups.

Figure 7: Summary of the concentrations (ng/g ww liver) of perfluoroalkyl and polyfluoroalkyl substances (PFASs) detected in liver tissue from the subset of Norwegian northern fulmars (Fulmarus glacialis) collected in 2012 and 2013 that were sampled for this contaminant study

PFASs (ng/g)	Mean ± SD	Absent Median	Range	*n*	Mean ± SD	Moderate Median	Range	*n*	Mean ± SD	High Median	Range	*n*
PFOSA	0.39 ± 0.29	0.33	<LOD–880	10	0.37 ± 0.22	0.44	0.12–0.60	5	0.18 ± 0.11	204	0.05–0.36	10
PFHxS	0.30 ± 0.20	0.31	<LOD–0.63	10	0.33 ± 0.17	0.39	0.15–0.48	5	0.35 ± 0.11	0.36	0.19–0.48	10
PFOS	6.19 ± 4.70	5.32	<LOD–13.7	10	10.8 ± 6.71	8.82	4.60–20.3	5	6.42 ± 5.20	4.76	<LOD–13.3	10
PFOA	0.16 ± 0.07	0.15	0.05–0.26	10	0.15 ± 0.08	0.14	0.08–0.27	5	*	*	*	*
PFNA	1.24 ± 0.10	0.85	0.18–3.28	10	1.25 ± 0.61	0.93	0.64–2.02	5	0.98 ± 0.61	0.87	0.19–2.03	10
PFDcA	0.88 ± 0.72	0.62	0.17–1.99	10	1.20 ± 0.92	0.82	0.42–2.45	5	0.99 ± 0.62	1.16	0.15–2.07	10
PFUnDA	2.94 ± 2.22	1.87	0.68–6.12	10	4.79 ± 3.24	3.47	1.87–8.75	5	2.90 ± 1.63	3.30	0.50–5.36	10
PFDoDA	0.75 ± 0.45	0.63	0.20–1.38	10	0.97 ± 0.53	0.79	0.46–1.59	5	0.84 ± 0.41	0.91	0.25–1.46	10
PFTrDA	3.63 ± 1.89	3.43	1.93–6.47	10	3.24 ± 1.64	2.83	1.59–5.70	5	6.21 ± 4.87	5.25	2.06–19.4	10
PFTeDA	0.70 ± 0.32	0.73	0.25–1.28	10	0.80 ± 0.36	0.70	0.44–1.39	5	1.08 ± 0.78	0.87	0.37–3.13	10
8:2 FTS	0.03 ± 0.02	0.03	0.004–0.05	10	*	*	*	*	0.05 ± 0.06	0.03	<LOD–0.21	10
∑PFAS	17.2 ± 9.97	15.2	4.58–32.7	10	24.0 ± 13.5	15.5	12.0–40.8	5	20.0 ± 9.78	22.2	7.01–34.3	10

Note: The data are presented as mean ± standard deviation (SD), median and range. *n* denotes the sample size.
No statistically significant differences were detected between fulmars with no ingested plastic (termed "absent"), a moderate plastic load (termed "moderate") and a high plastic load in their stomachs (termed "high").
* Excluded due to >70% of samples below LOD.

Table 8: Summary of the concentrations (ng/g ww muscle) of dichlorodiphenyltrichloroethane (DDT) and its metabolites detected in muscle tissue from the subset of northern fulmars (Fulmarus glacialis) caught in Norway in 2012 and 2013 that were sampled for this contaminant study

DDTs (ng/g)	Mean ± SD	Absent Median	Range	*n*	Mean ± SD	Moderate Median	Range	*n*	Mean ± SD	High Median	Range	*n*
p,p'-DDT	1.53 ± 1.50	0.91	0.11–4.53	9	1.56 ± 1.78	0.60	<LOD–4.41	9	0.80 ± 0.51	0.86	<LOD–1.64	11
o,p'-DDT/*p,p'*-DDD	10.3 ± 8.58	8.58	2.38–28.0	9	14.8 ± 12.8	17.6	0.73–39.3	9	8.63 ± 13.0	3.54	0.89–45.6	11
p,p'-DDE	260 ± 181	206	84.2–639	9	424 ± 345	352	21.7–1049	9	305 ± 396	122	32.4–1205	11
∑DDT	272 ± 190	209	88.1–669	9	441 ± 353	391	22.8–1076	9	315 ± 408	132	34.5–1251	11

Note: The data are presented as mean ± standard deviation (SD), median and range. *n* denotes the sample size.
No statistically significant differences were detected between fulmars with no ingested plastic (termed "absent"), a moderate plastic load (termed "moderate") and a high plastic load in their stomachs (termed "high").

Table 9: Summary of the concentrations (ng/g ww muscle) of polychlorinated biphenyls (PCBs) detected in muscle tissue from the subset of northern fulmars (Fulmarus glacialis) caught in Norway in 2012 and 2013 that were sampled for this contaminant study

PCBs (ng/g)	Mean ± SD	Absent Median	Range	*n*	Mean ± SD	Moderate Median	Range	*n*	Mean ± SD	High Median	Range	*n*
PCB-28/31	1.06 ± 0.44	0.98	0.41–1.69	9	1.06 ± 0.61	0.89	0.34–1.98	9	1.18 ± 0.70	0.96	0.37–2.73	11
PCB-52	0.33 ± 0.73	0.10	0.003–2.27	9	2.98 ± 4.14	1.20	0.02–12.3	9	0.37 ± 0.65	0.05	0.008–1.85	11
PCB-99	27.5 ± 17.1	20.7	10.0–59.6	9	40.0 ± 28.9	36.5	2.91–79.8	9	31.4 ± 34.7	16.7	4.54–112	11
PCB-101	0.61 ± 0.89	0.25	<LOD–2.74	9	0.77 ± 1.56	0.13	<LOD–4.72	9	0.06 ± 0.05	0.07	<LOD–0.15	11
PCB-105	27.5 ± 17.0	19.3	12.0–60.9	9	31.1 ± 20.9	28.2	2.61–64.1	9	27.4 ± 28.9	15.9	4.05–101	11
PCB-118	83.7 ± 52.1	63.3	34.3–189	9	98.6 ± 65.5	90.0	8.12–210	9	89.6 ± 89.3	54.8	13.4–308	11
PCB-138	113 ± 77.3	79.7	36.7–268	9	153 ± 109	142	9.82–307	9	112 ± 116	68.7	16.2–376	11
PCB-153	296 ± 188	216	117–683	9	317 ± 219	356	25.4–742	9	260 ± 205	195	39.8–741	11
PCB-170	52.8 ± 40.3	37.1	17.9–147	9	53.2 ± 39.5	58.0	4.03–135	9	40.2 ± 28.3	29.2	4.72–99.2	11
PCB-180	160 ± 122	114	54.6–448	9	158 ± 119	165	12.5–414	9	116 ± 76.3	89.5	12.5–260	11
PCB-183	17.8 ± 12.7	12.7	6.44–46.5	9	20.9 ± 14.3	18.9	1.54–47.7	9	14.5 ± 11.4	10.2	1.92–39.2	11
PCB-187	1.03 ± 1.47	0.40	0.13–4.64	9	1.61 ± 2.70	0.25	0.04–7.24	9	0.26 ± 0.16	0.18	0.07–0.53	11
PCB-189	2.20 ± 1.54	1.67	0.84–5.85	9	2.07 ± 1.65	1.88	0.14–5.60	9	1.59 ± 0.93	1.33	0.19–3.27	11
PCB-194	21.5 ± 14.6	18.7	8.03–56.1	9	20.5 ± 15.4	15.5	1.88–52.3	9	14.8 ± 8.76	12.1	1.40–25.8	11
∑PCB	805 ± 537	607	306–1966	9	901 ± 618	979	69.8–2057	9	709 ± 591	517	113–2067	11

Note: The data are presented as mean ± standard deviation (SD), median and range. *n* denotes the sample size.
No statistically significant differences were detected between fulmars with no ingested plastic (termed “absent”), a moderate plastic load (termed “moderate”) and a high plastic load in their stomachs (termed “high”).

Table 10: Summary of the concentrations (ng/g ww muscle) of polybrominated diphenyl ethers (PBDEs) detected in muscle tissue from the subset of northern fulmars (Fulmarus glacialis) caught in Norway in 2012 and 2013 that were sampled for this contaminant study

PBDEs (ng/g)	Mean ± SD	Absent Median	Range	*n*	Mean ± SD	Moderate Median	Range	*n*	Mean ± SD	High Median	Range	*n*
PBDE-28	0.05 ± 0.03	0.04	0.02–0.08	9	0.04 ± 0.03	0.04	<LOD–0.09	9	0.03 ± 0.03	0.02	0.01–0.10	11
PBDE-47	0.42 ± 0.31	0.34	0.08–1.06	9	0.49 ± 0.74	0.17	0.23–2.16	9	0.12 ± 0.05	0.10	0.06–0.23	11
PBDE-99	0.16 ± 0.15	0.11	0.03–0.51	9	0.45 ± 0.77	0.11	0.01–2.19	9	0.07 ± 0.04	0.07	0.01–0.12	11
PBDE-100	0.10 ± 0.06	0.09	0.02–0.23	9	0.12 ± 0.18	0.04	0.01–0.54	9	0.04 ± 0.02	0.02	0.02–0.08	11
PBDE-119	0.03 ± 0.01	0.03	0.01–0.04	9	0.03 ± 0.03	0.02	<LOD–0.07	9	0.02 ± 0.02	0.02	<LOD–0.05	11
PBDE-153	0.31 ± 0.15	0.30	0.14–0.65	9	0.50 ± 0.36	0.56	0.04–0.97	9	0.27 ± 0.21	0.24	0.06–0.69	11
PBDE-154	0.19 ± 0.08	0.17	0.08–0.32	9	0.25 ± 0.27	0.11	0.02–0.78	9	0.12 ± 0.07	0.12	0.05–0.26	11
∑PBDE	1.25 ± 0.63	1.18	0.49–2.57	9	1.87 ± 2.07	0.89	0.10–5.42	9	0.66 ± 0.36	0.59	0.24–1.18	11

Note: The data are presented as mean ± standard deviation (SD), median and range. n denotes the sample size.
No statistically significant differences were detected between fulmars with no ingested plastic (termed “absent”), a moderate plastic load (termed “moderate”) and a high plastic load in their stomachs (termed “high”).

3.3 Comparison between Faroese and Norwegian fulmars

The concentration of contaminants in liver tissue was compared in fulmars from the "high" groups from the Faroe Islands and Norway. The concentrations are listed in Tables 11, 12 and 13.

After applying the Holm-Šidàk correction, there were no statistically significant differences in contaminant concentrations between the Faroese and Norwegian fulmars. The concentrations were also corrected for the mass of plastic ingested by the fulmars by dividing the individual concentrations by the mass of plastic ingested by each bird. When comparing the plastic corrected concentrations, no statistically significant differences persisted after applying the Holm-Šidàk correction.

Table 11: A comparison of the concentrations (ng/g ww liver) of polychlorinated biphenyls (PCBs) detected in liver tissue from the subset of northern fulmars (Fulmarus glacialis) caught in the Faroe Islands (2011) and Norway (2012–2013) that were sampled for this contaminant study

PCBs (ng/g)	Mean ± SD	Faroe Islands Median	Range	*n*	Mean ± SD	Norway Median	Range	*n*
PCB-28	0.78 ± 0.36	0.81	0.19–1.26	9	1.77 ± 1.55	1.21	<LOD–4.34	10
PCB-52	0.22 ± 0.26	0.10	<LOD–0.72	9	2.65 ± 3.21	1.45	0.48–10.2	10
PCB-99	50.6 ± 59.3	34.3	3.69–199	9	46.8 ± 58.5	26.9	10.9–178	10
PCB-101	1.21 ± 1.52	0.38	<LOD–4.10	9	0.12 ± 0.08	0.12	<LOD–0.27	10
PCB-105	21.2 ± 18.6	18.2	3.10–63.6	9	46.4 ± 57.1	29.3	9.98–187	10
PCB-118	61.4 ± 57.5	49.7	7.17–196	9	166 ± 194	108	34.9–625	10
PCB-138	173 ± 160	128	15.7–544	9	179 ± 213	105	34.4–654	10
PCB-153	343 ± 250	267	41.9–806	9	386 ± 397	290	58.1–134	10
PCB-180	102 ± 55.1	92.8	25.4–177	9	166 ± 153	124	19.3–511	10
∑PCB	753 ± 583	594	97.3–1989	9	1100 ± 1171	783	183–3830	10

Note: The data are presented as mean ± standard deviation (SD), median and range. *n* denotes the sample size.
No statistically significant differences were detected.

Table 12: A comparison of the concentrations (ng/g ww liver) of dichlorodiphenyltrichloroethane (DDT), its metabolites and polybrominated diphenyl ethers (PBDEs) detected in liver tissue from the subset of northern fulmars (Fulmarus glacialis) caught in the Faroe Islands (2011) and Norway (2012–2013) that were sampled for this contaminant study

Chemicals (ng/g)	Mean ± SD	Faroe Islands Median	Range	*n*	Mean ± SD	Norway Median	Range	*n*
p,p'-DDE	390 ± 255	406	32.8–812	9	419 ± 562	164	74.5–1634	10
p,p'-DDD	6.82 ± 5.63	5.91	1.02–18.8	9	5.36 ± 7.45	1.96	0.43–24.4	10
∑DDT	396 ± 258	412	34.2–817	9	425 ± 570	168	76.1–1660	10
PBDE-47	0.58 ± 0.56	0.32	0.11–1.70	9	0.19 ± 0.13	0.17	0.45–0.52	10
PBDE-153	0.77 ± 0.74	0.63	<LOD–2.47	9	0.56 ± 0.64	0.32	0.07–2.00	10
∑PBDE	1.35 ± 0.92	1.63	0.19–2.83	9	1.19 ± 0.95	0.84	0.20–2.97	10

Note: The data are presented as mean ± standard deviation (SD), median and range. *n* denotes the sample size. No statistically significant differences were detected.

Table 13: A comparison of the concentrations (ng/g ww liver) of perfluoroalkyl and polyfluoroalkyl substances (PFASs) detected in liver tissue from northern fulmars (Fulmarus glacialis) caught in the Faroe Islands (2011) and Norway (2012–2013)

	PFASs (ng/g)	Mean ± SD	Faroe Islands Median	Range	*n*	Mean ± SD	Norway Median	Range	*n*
	PFOS	9.13 ± 4.63	8.15	2.96–15.9	9	6.19 ± 4.70	5.32	<LOD–13.7	10
	PFNA	0.85 ± 0.56	0.98	0.22–1.58	9	1.24 ± 0.98	0.85	0.18–3.28	10
Absent	PFDcA	0.96 ± 0.73	1.03	0.15–2.29	9	0.88 ± 0.72	0.52	0.17–1.99	10
	PFUnDA	3.86 ± 2.29	3.38	1.23–7.86	9	2.94 ± 2.22	1.87	0.68–6.12	10
	PFTriDA	4.20 ± 2.02	4.15	1.70–6.69	9	3.63 ± 1.89	3.43	1.03–6.47	10
	∑PFAS	19.1 ± 9.51	17.3	6.98–32.0	9	17.2 ± 9.97	15.2	4.58–32.7	10
	PFOS	10.5 ± 3.24	9.83	7.44–18.0	9	6.42 ± 5.20	4.76	<LOD–13.3	10
	PFNA	1.13 ± 0.70	0.80	0.46–2.69	9	0.98 ± 0.61	0.87	0.19–2.03	10
High	PFDcA	1.26 ± 0.54	0.98	0.65–2.26	9	0.99 ± 0.62	1.16	0.15–2.07	10
	PFUnDA	4.47 ± 2.00	3.68	2.57–8.65	9	2.90 ± 1.63	3.30	0.50–5.36	10
	PFTriDA	4.47 ± 1.84	4.02	2.50–7.61	9	6.21 ± 4.87	5.25	2.06–19.4	10
	∑PFAS	21.9 ± 8.06	17.1	15.2–39.4	9	20.0 ± 9.78	22.2	0.70–34.3	10

Note: The data are presented as mean ± standard deviation (SD), median and range. *n* denotes the sample size.
No statistically significant differences were detected.

4. Discussion

4.1 Contaminant levels

There were no statistically significant differences in contaminant concentrations in liver tissue between fulmars from the Faroe Islands and Norway and the averaged contaminant levels are thus comparable between the two datasets.

Fulmars have extensive foraging areas and a wide-ranging behaviour (Anker-Nilssen *et al*., 2000; Weimerskirch *et al*., 2001), and as such the individuals sampled in the Faroe Islands and in Norway may belong to the same metapopulation. This would help explain why there were no significant differences in contaminant concentrations between the two datasets.

A study by Knudsen *et al*. (2007) examined contaminant levels in liver and blood of fulmars from Bjørnøya, Norway. As for the Faroese fulmars in the current study, they did not detect heptachlor, *cis*-chlordane α- and γ-HCH, but they did detect *trans*-nonachlor. The values for HCB are similar, while the concentrations for *oxy*-chlordane, *cis*-nonachlor and mirex are higher in this study. Levels of PFOS, *p,p'*-DDE and *p,p'*-DDD are considerably higher for the Faroese and Norwegian fulmars in this study. The levels for *p,p'*-DDE were substantially higher in the present study than for fulmars sampled from the Aleutian Islands, USA, but concentrations of ∑PCB were similar (Ricca *et al*., 2008). Braune *et al*. (2010) investigated levels of contaminants in livers from fulmars caught in the Canadian Arctic. The reported concentrations of *p,p'*-DDE, *trans*-nonachlor, mirex, HCB, *oxy*-chlordane, PCB-153 and PFOS are all considerably lower than observed in the present study. They did, however, detect ∑HCH which was below LOD in the current work. Another study of fulmars from the Canadian arctic also found lower contaminant levels as compared to those reported here (Foster *et al*., 2011). For instance they detected PCB-153 at 15.6 ± 11.4 ng/g ww in the liver, while the lowest concentration of PCB-153 in this study was 260 ± 205 ng/g ww in the liver tissue (for Norwegian fulmars in the "high" group). Similarly, Martin *et al*. (2004) detected PFAS concentrations at levels below those reported here in fulmar livers from the Canadian arctic.

The contaminant concentrations tend to be higher in the current work compared to existing literature on fulmars sampled in the Arctic. This may largely be due to spatial differences in pollution levels (Vander Pol *et al.*, 2004; Vorkamp *et al.*, 2004; Muir and de Wit, 2010; Mallory and Braune, 2012).

4.2 Effect of plastic ingestion on contaminant levels

After correcting for the multiple testing, there were no statistically significant differences between plastic ingestion groups in either dataset. Based upon the results herein, ingested plastic does not appear to be a significant vector for exposure of the studied, adsorbed contaminants in fulmars. This is in accordance with the conclusion reached by Herzke *et al.* (2016) that adsorbed contaminant exposure from prey is more important than contaminant exposure from ingested plastics.

However, there are studies reporting indications that plastic is an important route of exposure for contaminants in seabirds (Ryan *et al.*, 1988; Teuten *et al.*, 2009; Yamashita *et al.*, 2011; Tanaka *et al.*, 2013). As the sample sizes in the current study were so low that the power of the t-tests were below the desired statistical power, the likelihood of committing a Type II error increased. Likewise, the Holm-Šidàk correction is conservative and makes the detection of significant differences harder still. This may, then, have prevented the detection of some significant differences.

Finally, it is most important to stress that, given the multitude of chemicals intentionally added to plastics and degradation products of additives and polymers, the lack of association between adsorbed contaminants and plastics found in this report is not representative for the potential overall chemical impact of plastic ingestion. The study of other contaminants found in ingested plastics and their possible threat to fulmars should be prioritized in future research.

Conclusion

No statistically significant differences in contaminant concentrations were detected between plastic ingestion groups. Nor was there a significant difference between contaminant levels in liver tissue from Faroese and Norwegian fulmars. In general, ingested plastics does not appear to be a significant exposure vector for the contaminants adsorbed to plastics in fulmars. For adsorbed chemicals, the natural diet of the fulmars seems to be the most important source of exposure. However, from our analyses it is not possible to make a conclusion for plastic additives and degradation products. Such substances should be a focus for further research efforts.

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Norwegian summary

Marin plastforurensning er et omfattende og økende problem. På grunn av de kjemiske og fysiske egenskapene til plast har den en tendens til å forbli i det marine miljøet over lange tidsperioder, hvor den har et potensial til å skade fauna og flora. Blant de mange truslene plast utgjør, er inntak av plast et av de mest bekymringsfulle, da dette blir hyppig observert i flere forskjellige arter. Sjøfugler, og spesielt Procellariiformes ordenen, blir ofte funnet med en betydelig mengde plast i magene. Foruten de fysiske farene ved å innta plast (som for eksempel indre skader og at plasten setter seg fast i fordøyelsessystemet) er det en bekymring at kjemikaliene som blir tilsatt og heftet til plasten kan bli tatt opp av fuglen og medføre toksiske effekter. Målet med denne studien var å undersøke dette ved å utvide og sammenligne to datasett om miljøgiftkonsentrasjoner i utvalgte vev og inntatt plast i havhest (*Fulmarus glacialis*).

Havhestene fra Færøyene var alle bifangst i langlinefiske i 2011, og havhestene fra Norge var hovedsakelig bifangst i fiskerinæringen i 2012 og 2013, med noen individer som ble funnet strandet. Under disseksjon ble plasten i magesekken kvantifisert og vev (lever for havhest fra Færøyene og lever og muskel for havhest fra Norge) fryst ned for senere kjemisk analyse. Prøvene ble analysert for en rekke persistente organiske miljøgifter: polyklorinerte bifenyler, polybrominerte difenyl etere, perfluoroalkyl og polyfluoroalkyl-stoffer, metabolitter, organofosfat flammehemmere, diklorodifenyltrikloroetan og andre pesticider. Dataene ble så analysert statistisk for å undersøke om det var en sammenheng mellom hvor mye plast som var inntatt og hvor høy konsentrasjon det var av miljøgifter i vevet til havhestene. Samtidig ble miljøgiftbyrden sammenlignet for havhester fra Færøyene og Norge.

Etter Holm-Šidàk-korreksjonen for gjentatte sammenligninger, var det ingen signifikante forskjeller i miljøgiftkonsentrasjon mellom grupper med ingen, medium eller høyt inntak av plast. Miljøgiftnivået i lever til havhester fra Færøyene og Norge var ikke signifikant forskjellig etter korreksjonen. Dermed virker det som at inntatt plast ikke er en signifikant eksponeringsrute til de adsorberte miljøgiftene analysert for i dette studiet for havhest.

Appendix A

The samples were analysed for the following polychlorinated biphenyls (PCBs) congeners in both datasets: PCB-28/31, PCB-52, PCB-99, PCB-101, PCB-105, PCB-118, PCB-138, PCB-153 and PCB-180. Additionally, the samples from the Norwegian fulmar dataset was analysed for PCB-170, PCB-183, PCB-187, PCB-189 and PCB-194.

Samples from both datasets were analysed for polybrominated diphenyl ethers (PBDEs): PBDE-28, PBDE-47, PBDE-99, PBDE-100, PBDE-138, PBDE-153, PBDE-154 and PBDE-183. The samples from the Norwegian dataset were also analysed for PBDE-196, PBDE-197, PBDE-206, PBDE-207 and PBDE-209.

The samples from both datasets were analysed for *o,p'*-DDT, *p,p'*-DDT, *p,p'*-DDD, *o,p'*-DDE and *p,p'*-DDE. The samples from the Faroese dataset were also analysed for hexachlorobenzene (HCB), α-hexachlorocyclohexane (HCH), β-HCH, γ-HCH, heptachlor, *oxy*-chlordane, *trans*-chlordane, *cis*-chlordane, *trans*-nonachlor, *cis*-nonachlor and mirex.

For metabolites and organophosphate flame retardants, only samples from the Faroe Islands dataset were analysed: PCP, 4-OH-PCB107, 4-OH-PCB120, 4-OH-PCB130, 3-OH-PCB153, 4-OH-PCB146, 3-OH-PCB138, 4-OH-PCB163, 4-OH-PCB187, 4-OH-PCB172, 4'-OH-PCB193, 3-MeSO-PCB49, 4-MeSO-PCB49, 3-MeSO-PCB52, 4-MeSO-PCB52, 3-MeSO-PCB91, 4-MeSO-PCB91, 3-MeSO-PCB101, 4-MeSO-PCB101, 3-MeSO-PCB87, 3-MeSO-PCB110, 4-MeSO-PCB110, 3-MeSO-PCB132, 3-MeSO-PCB149, 4-MeSO-PCB149, 4-MeSO-PCB132, 3-MeSO-PCB141, 4-MeSO-PCB141, 3-MeSO-PCB174, 4-MeSO-PCB174, 4-OH-HpCS, 2-OH-BDE68, 6-OH-BDE47/55, 5-OH-BDE47, 4-OH-BDE49, 5-OH-BDE100, 4-OH-BDE103, 5-OH-BDE99, 4-OH-BDE101, 3-MeSO2-DDE. Organophosphate flame retardants were also analysed for the Faroese dataset only, in the "absent" and "high" groups: TEP, TCEP, TPrP, TCPP, TiBP, BdPhP, TPP, DBPhP, TnBP, TDCPP, TBEP, TCP, EHDP and TEHP.

Perfluoroalkyl and polyfluoroalkyl substances (PFASs) were analysed in both datasets and summarized in Table A1 below. Three PFASs were only analysed for in one dataset, and are indicated in the table.

Table A1: A summary of the perfluoroalkyl and polyfluoroalkyl substances (PFASs) analysed in muscle and liver samples from northern fulmars (Fulmarus glacialis) caught in the Faroe Islands (2011) and Norway (2012–2013)

Group	Abbreviation	Analyte	Only Faroe Islands	Only Norway
n:2 fluorotelomer sulfonic acids	6:2 FTS	6:2 fluorotelomer sulfonic acid		
	8:2 FTS	8:2 fluorotelomer sulfonic acid		x
Perfluoroalkane sulfonamides	PFOSA	Perfluorooctane sulfonamide		
Perfluoroalkane sulfonic acids	PFBS	Perfluorobutane sulfonic acid		
	PFPS	Perfluoropentane sulfonic acid	x	
	PFHxS	Perfluorohexane sulfonic acid		
	PFHpS	Perfluoroheptane sulfonic acid		
	brPFOS	Branched perfluorooctane sulfonic acid		
	PFOS	Perfluorooctane sulfonic acid		
	PFNS	Perfluorononane sulfonic acid	x	
	PFDcS	Perfluorodecane sulfonic acid		
Perfluoroalkyl carboxylic acids	PFBA	Perfluorobutanoic acid		
	PFPA	Perfluoropentanoic acid		
	PFHxA	Perfluorohexanoic acid		
	PFHpA	Perfluoroheptanoic acid		
	PFOA	Perfluorooctanoic acid		
	PFNA	Perfluorononanoic acid		
	PFDcA	Perfluorodecanoic acid		
	PFUnDA	Perfluoroundecanoic acid		
	PFDoDA	Perfluorododecanoic acid		
	PFTrDA	Perfluorotridecanoic acid		
	PFTeDA	Perfluorotetradecanoic acid		

Note: The last two columns of the table indicate analytes which were only analysed for in one dataset. If the row is blank for the last two columns, the analyte was analysed for in both datasets.

Appendix B

o,p'-DDT, *p,p'*-DDT, *o,p*'-DDE, PBDE-138, α-, β-, and γ-HCH and heptachlor were eliminated for all three groups in the Faroe dataset. Additionally, it excluded PBDE-28, PBDE-99, PBDE-100, PBDE-153, PBDE-154 and PBDE-183 in the absent group; and PBDE-28, PBDE-99, PBDE-100, PBDE-154 and PBDE-183 in the high group.

For metabolites and PFASs, only the absent and high groups were analysed. 4-OH-HpCS, 4-OH-PCB120, 4-OH-PCB130, 2-OH-BDE68, 6-OH-BDE47/55, 5-OH-BDE47, 4-OH-BDE49, 5-OH-BDE100, 4-OH-BDE103, 5-OH-BDE99, 4-OH-BDE101, 3MeSOPCB49, 4MeSOPCB52, 4MeSOPCB49, 6:2 FTS, PFOSA, PFBS, PFPS, PFHxS, PFHpS, brPFOS, PFNS, PFDcS, PFBA, PFPA, PFOA, PFDoDA, PFTeDA were excluded for both the absent and high group. 4-OH-PCB163, 3MeSOPCB52, 4MeSOPCB91, 3MeSO$_2$DDE, 3MeSOPCB110, 3MeSOPCB132, 3MeSOPCB174, PFHxA were excluded in the high group. 4-OH-PCB107 was excluded in the absent group.

Organophosphate flame retardants were analysed for the "absent" and "high" groups for the Faroe Islands fulmars. For TEP, TCEP, TPrP, TiBP, BdPhP, TPP, DBPhP, TnBP, TDCPP, TCP, EHDP and TEHP all individuals had values below LOD. For TCPP and TBEP some individuals had values above the detection limit, but more than 70% were below the limit and thus excluded.

For the Norway dataset, PBDE-138, PBDE-183, PBDE-196, PBDE-197, PBDE-206, PBDE-207 and PBDE-209 were excluded from all three groups for muscle tissue and for liver tissue. Additionally, PBDE-119 was eliminated from the liver tissue group. For all groups in both tissue matrices, *o,p'*-DDT was excluded from analyses and *o,p'*-DDD was excluded for the muscle tissue groups. 6:2 FTS, PFBS, PFHpS, PFDcS, PFBA, PFPA, PFHxA, PFHpA, brPFOS were eliminated for all groups. PFOA was exluded for the high group and 8:2 FTS for the low group, respectively.

Table B1: A list of the limit of detection (LOD) values in ng/g for the contaminants analysed in liver and muscle tissue in northern fulmars (Fulmarus glacialis) from the Faroe Islands and Norway

Chemical	LOD	Chemical	LOD	Chemical	LOD	Chemical	LOD	Chemical	LOD
TEP	< 0.7	PFNS	< 0.05	4-OH-PCB 187	< 0.003	4MeSOPCB110	< 0.0003	PCB-99	< 0.07
TCEP	< 0.3	PFDcS	< 0.05	4-OH-PCB 172	< 0.01	3MeSOPCB149	< 0.0002	PCB-101	< 0.08
TPrP	< 0.03	PFBA	< 0.04	4'-OH-PCB 193	< 0.003	4MeSOPCB149	< 0.0002	PCB-105	< 0.12
TCPP	< 0.4	PFPA	< 0.04	2-OH-BDE68	< 0.02	3MeSOPCB132	< 0.0002	PCB-118	< 0.04
TiBP	< 1.9	PFHxA	< 0.01	6-OH-BDE47/75	< 0.02	4MeSOPCB132	< 0.0002	PCB-138	< 0.11
BdPhP	< 0.01	PFHpA	< 0.01	5-OH-BDE47	< 0.03	3MeSOPCB141	< 0.0001	PCB-153	< 0.09
TPP	< 0.04	PFOA	< 0.01	4-OH-BDE49	< 0.04	4MeSOPCB141	< 0.0001	PCB-180	< 0.06
DBPhP	< 0.02	PFNA	< 0.01	5-OH-BDE100	< 0.04	3MeSOPCB174	< 0.0002	*o,p'*-DDT/*p,p'*-DDD	< 0.39
TnBP	< 0.04	PFDcA	< 0.01	4-OH-BDE103	< 0.04	4MeSOPCB174	< 0.0002	*p,p'*-DDT	< 0.4
TDCPP	< 0.1	PFUnDA	< 0.01	5-OH-BDE99	< 0.06	HCB	< 0.001	*o,p'*-DDE	< 1.61
TBEP	< 0.1	PFDoDA	< 0.1	4-OH-BDE101	< 0.06	α-HCH	< 0.008	*o,p'*-DDD	< 0.05
TCP	< 0.01	PFTrDA	< 0.1	3MeSOPCB52	< 0.0002	β-HCH	< 0.02	PBDE-28	< 3.2
EHDP	< 0.06	PFTeDA	< 0.1	3MeSOPCB49	< 0.0002	γ-HCH	< 0.005	PBDE-47	< 5.0
TEHP	< 0.01	PCP	< 0.05	4MeSOPCB52	< 0.00008	Heptachlor	< 0.01	PBDE-99	< 2.0
6:2FTS	< 0.05	4-OH-HpCS	< 0.001	4MeSOPCB49	< 0.00004	*Oxy*-chlordane	< 0.002	PBDE-100	< 2.0
PFOSA	< 0.05	4-OH-PCB 120	< 0.02	3MeSOPCB91	< 0.0001	*trans*-chlordane	< 0.003	PBDE-138	< 2.0
PFBS	< 0.05	4-OH-PCB 107	< 0.07	4MeSOPCB91	< 0.0002	*cis*-chlordane	< 0.004	PBDE-153	< 2.0
PFPS	< 0.05	3-OH-PCB 153	< 0.02	3MeSOPCB101	< 0.0002	*trans*-nonachlor	< 0.002	PBDE-154	< 2.0
PFHxS	< 0.05	4-OH-PCB 146	< 0.009	4MeSOPCB101	< 0.0002	*cis*-nonachlor	< 0.001	PBDE-183	< 4.0
PFHpS	< 0.05	3-OH-PCB 138	< 0.008	3MeSO2 DDE	< 0.0004	Mirex	< 0.007		
brPFOS	< 0.05	4-OH-PCB 130	< 0.02	3MeSOPCB87	< 0.0004	PCB-28	< 0.02		
PFOS	< 0.05	4-OH-PCB 163	< 0.003	3MeSOPCB110	< 0.0003	PCB-52	< 0.06		

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Contaminants in northern fulmars (*Fulmarus glacialis*) exposed to plastic

Northern fulmars are seabirds which feed exclusively at sea, and as such, they are useful indicators of ocean health. Marine plastic pollution is an ever-increasing and global issue that affects the northern fulmar as they are frequently found to have ingested plastic. In this report we investigate whether the amount of ingested plastic affects the concentration of certain plastic-adsorbed toxicants in their tissues. Marine plastic pollution is a field of utmost importance. It is our hope that this continues to be an area which receives increased attention in order to elucidate the potential harmful effects plastics have on the northern fulmar and ocean health, in general.

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