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Summary

Plastic pollution is of worldwide concern. However, international commercial advances into the Arctic are occurring without knowledge of the existing threat posed to the local marine environment by plastic litter.

Here, we quantify plastic ingestion by northern fulmars, *Fulmarus glacialis*, from Svalbard, at the gateway to future shipping routes in the high Arctic. Plastic ingestion by Svalbard fulmars does not follow the established decreasing trend away from human marine impact. Of 40 individuals, 87.5% had ingested plastic, averaging at 0.08g or 15.3 pieces per individual. Plastic ingestion levels in Svalbard exceed the ecological quality objective defined by OSPAR for European seas, highlighting an urgent need for mitigation of plastic pollution in the Arctic, and international regulation of future commercial activity.

Preliminary analises of new data for plastic ingestion by fulmars in Iceland support the arguments above, and reveal that annual variation in plastic may be significant: an area warranting further study. The updated monitoring average shows that 84% of northern fulmars in Iceland have ingested plastic. Levels in Iceland still exceed OSPAR monitoring targets.

In addition, this report presents an increase in variability of tissue contaminant load with plastic ingestion, although differences are not significant. This is the case for multiple classes of contaminants, including PCBs, PBDEs, DDTs, chlordanes and other pesticides, that could either be adsorbing to the surface of plastic pieces whilst in sea water or leaching from within the plastic (e.g. flame retardants). This further emphasises the need for mitigation of plastic pollution and strict enforcement of legislation in the future.

This report highlights future research needs, as well as policy needs to regulate and mitigate this major environmental problem.

Norwegian Summary — Sammendrag på norsk

Plastforurensning i verdenshavene er et økende problem. Økt trafikk i arktiske områder skjer uten god nok kunnskap om hvordan plastforurensningen vil påvirke det marine miljø i Arktis.

I denne studien har vi kvantifisert plastforurensning i magene til havhester, *Fulmarus glacialis*, fra Svalbard. Plastinntak hos havhester fra Svalbard følger ikke en nedadgående trend fra sentral-Europa (kildeområde) mot Arktis.

Av 40 havhester undersøkt hadde 87,5 % inntatt plast, med et gjennomsnitt av 0,08 gram og 15,3 plastdeler per individ. Plastinntak hos havhester fra Svalbard overgår en grenseverdi (Ecological Quality Objective (EcoQO)) som er etablert av OSPAR i europeiske havområder. Dette viser at problemet med plast i havene er økende, og at det er behov for internasjonal regulering knyttet til plastutslipp.

Foreløpige analyser av nye data av plastinntak i havhester fra Island støtter argumentene ovenfor og viser store årlige variasjoner i inntak hos havhester. Overvåking av plast hos havhester fra Island i en periode på 5 år viser i gjennomsnitt at 84 % av fuglene hadde plast i magene. I havhestene på Island overgår 35,7 % av fuglene grenseverdien (EcoQO) på mer enn 0,1 gram plast i magene.

I tillegg viser denne rapporten stor variasjon i persistente organiske miljøgifter (POPs) hos havhestene. Stoffer analysert var PCBer, PBDEer, DDTer, klordaner og andre pestisider. Det ble ikke påvist noen signifikant forskjell i POPs hos havhester med lite og mye plast i magene. Denne studien omfatter ikke analyse av plaststoffer som flatalater og bispenoler, men indikerer at fugler kan bli påvirket av miljøgiftene som inngår i plastbitene.

Rapporten viser også forskningsbehov og viktigheten av å regulere og gjøre tiltak rettet mot dette store miljøproblemet.

Introduction

The threat of pollution in the Arctic is rising as commercial activity increases, enabled by rapid sea ice decline (Kerr 2012) and driven by economics and geopolitics (Brigham 2011). The Arctic is currently an area of low human impact (Halpern et al. 2008), however, increased shipping may put areas of high biodiversity at risk (Humphries and Huettmann 2014). Indeed, shipping density has previously been linked to the prevalence of plastic ingestion by marine life (Van Franeker et al. 2011; Kühn and Van Franeker 2012). The global plastic industry is continuously expanding (Plastics Europe 2013), the use of disposable plastic products persists (WRAP 2014), and it is likely that the already significant amounts of plastic litter entering the marine environment will increase (Law and Thompson 2014). The deleterious impacts of plastic litter are numerous, including transport of pollutants (Zarfl and Matthies 2010) and invasive species (Barnes 2002), entanglement with and ingestion by marine fauna (Laist 1997), as well as economic costs (Leggett et al. 2014). Considering this, there is an urgent need for a quantitative assessment of pollution levels in the Arctic. Such an assessment can provide information for development of international regulation to protect the marine environment for the future (Brigham 2011), as well as a tool for monitoring potential impacts of future commercial activity.

Plastic ingestion has been documented in over 100 species of seabird (Laist 1997), which has led to the identification of species with characteristics that make them suitable as biological monitors of trends in plastic pollution (Van Franeker et al. 2011). Northern fulmars have been extensively used as an indicator species for plastic pollution levels in the northern hemisphere since they were first used for monitoring around the North Sea in the 1980s (Van Franeker et al. 2011). At present, data exist for much of the North-East Atlantic (Van Franeker et al. 2011; Kühn and Van Franeker 2012), the Canadian Arctic (Provencher et al. 2009) and the eastern North Pacific (Avery-Gomm et al. 2012; Donnelly-Greenan et al. 2014). Northern fulmars are entirely oceanic feeders, and omnivorous foraging behaviour renders them particularly vulnerable to plastic ingestion (Van Franeker et al. 2011). Fulmars tend not to regurgitate hard prey items, but they remain in the muscular stomach until they are broken down to a size that can pass through the gut. Therefore, stomach plastic contents represent a recent period prior to death, and thus plastic pollution in the local area (Van Franeker et al. 2011).

Within Europe, northern fulmars (*Fulmarus glacialis*) are defined by OSPAR (The convention for the protection of the marine environment of the North-East Atlantic) as an indicator species. OSPAR recommendations state that for acceptable ecological quality (EcoQO), less than 10% of the monitored population of northern fulmars should have more than 0.1g of plastic in the stomach (Heslenfeld et al. 2009).

Plastic pollution typically decreases away from areas of high human impact and commercial activity, coinciding with an increase in latitude (Barnes 2002; Barnes 2005; Kühn and Van Franeker 2012). The main sources of plastic in the ocean are accidental losses during transport, irresponsible human behaviour, improper waste management and loss during natural disasters. However, there is no complete or recent information regarding plastic ingestion by northern fulmars at the highest latitudes in Europe. Svalbard, in the European Arctic, is an area of high seabird biodiversity (Humphries and Huettmann 2014) that is likely to experience a substantial increase in shipping traffic in the future (Smith and Stephenson 2013) and therefore potential increases in plastic pollution. Although the Arctic has long lost its wilderness status (France 1992), measurement of the extent of anthropogenic litter in the region is lacking.

Effects of plastic ingestion on seabirds

Ingestion of plastic debris has direct negative effects on seabirds, such as internal wounds and blockage of the digestive tract (Gregory 2009), as well as causing secondary stress (Sievert and Sileo 1993; Auman et al. 1998) and uptake of organic contaminants that are of high environmental concern (Ryan et al. 1988; Colabuono et al. 2010; Tanaka et al. 2013). Contaminants can leach from within the plastic particles (e.g. colourants, flame-retardants, and softeners used within plastic products) or can have adsorbed to the outside of the plastic particle from seawater (Teuten et al. 2007).

Exposure to these compounds can occur through natural prey via bioaccumulation (Borgå et al. 2001); however, direct consumption of plastic is considered an additional source. This is particularly evident for seabirds of the order Procellariiformes that are generally found with highest levels of plastic ingestion, hypothesized to be because of the structure of the gizzard and the fact that they do not regurgitate hard items (Tourinho et al. 2010). PCBs (*Polychlorinated biphenyls*) in tissues of female Great Shearwaters (*Puffinus gravis*) have been found to directly correlate with the amount of plastic ingested by the individual (Ryan et al. 1988). Similarly, PBDEs (polybrominated diphenyl ethers) are found in tissues of short-tailed shearwaters (*Puffinus tenuirostris*) that are present in ingested plastic but not in tissues of natural prey (Tanaka et al. 2013).

Organochlorine pollutants, including those mentioned above, have been proven to result in a plethora of detrimental effects in seabirds. Species will metabolise contaminants (albeit at different rates (Helgason et al. 2010)) and resulting changes in hepatic enzyme activity (for example, EROD activity) are detectable with varying contaminant concentrations (Verreault et al. 2013). POPs have been found to result in endocrine effects, such as disrupted hormone ratios in northern fulmars (*Fulmarus glacialis*) and black-legged kittiwakes (*Rissa tridactyla*) (Nøst et al. 2012; Verreault et al. 2013). In glaucous gulls (*Larus hyperboreus*), POPs have been found to correlate with decreased immunity (Sagerup et al. 2000; Sagerup et al. 2009; Sagerup et al. 2014), behavioural differences during the breeding period (Bustnes et al. 2001), decreased levels of reproduction (Bustnes et al. 2003) and may contribute to decreased adult survival (Gabrielsen et al. 1995; Bustnes et al. 2003).

In March 2004, exceptionally large numbers of beached northern fulmars were found around the southern coasts of the North Sea, most concentrated along the coasts of Belgium, northern France, Germany, the Netherlands and southern England (Van Franeker et al. 2011). Above average numbers also occurred as far north as southern Norway. Many of the beached fulmars had feathers in a poor condition, having arrested both tail and primary moult the previous October, and an unusually large number were adult females (Van Franeker et al. 2011). Increased mortalities continued into May and June, including several individuals carrying eggs that were found dead at large distances from any colonies. Such findings are contrary to the usual behaviour of long-lived petrel species to abandon reproduction if adult survival is threatened, for example by poor body condition (Chastel et al. 1995). Feather moult, feather condition, and reproductive behaviour are all regulated by the endocrine system (Van Franeker et al. 2011). The observed disruption of these could therefore be indicative of a hormonal disruption from chemicals associated with ingested plastic (Van Franeker et al. 2011). At the time, no funding was available to test this hypothesis and therefore this interpretation is merely speculative.

In the March 2004 mortality event, the delay in feather moult showed that hormone disruption began in the previous autumn and persisted until a threshold level of tissue chemical load was reached at a time of high-energy demand. Such hormonal disruption could have population effects if widespread, such as the 2004 fulmar wreck (Van Franeker et al. 2011). It is therefore important that if related to plastic pollution, any correlative link that exists between cause and effect is determined. This will enable policy makers to be informed about the adverse effects of chemicals entering the environment, as well as the magnitude and extent of the threat posed to marine ecosystems by plastic pollution (Depledge et al. 2013; Rochman et al. 2013)

Study aims and objectives

To assess plastic pollution levels in the European Arctic, this study quantified the amount of plastic ingested by northern fulmars from Spitsbergen, the largest island of the Svalbard archipelago and the Westfjords of Iceland. In the 1980s, plastic ingestion by fulmars from Spitsbergen and Bjørnøya (Bear Island, mid-way between Spitsbergen and mainland Norway) was observed during diet studies (Gjertz et al. 1985; Lydersen et al. 1985; Van Franeker 1985; Lydersen and Gjertz 1989). However, the mass of plastic was not recorded and data from Spitsbergen are incomplete, therefore comparison to other data is not possible. This study will be the first dedicated study of plastic ingestion by arctic fulmars in this area, and thus represents a valuable northwards expansion of Atlantic/North Sea monitoring efforts. The study in Iceland will be a repeat of data collected in 2011, yet at a different time of year.

Secondly, to determine the potential for effects of plastic on seabirds, contaminant concentrations were studied in northern fulmars, as an indicator species, from the Faroe Islands across a range of plastic ingestion amounts.

This report presents plastic ingestion results from Svalbard and Iceland, alongside analysis of spatial trends, as well as a comparison of contaminant loads in liver tissue of fulmars with no ingested plastic or high levels of ingested plastic.

Materials and methods

This project is registered in the Research in Svalbard (RiS) database, within the Svalbard Science Forum (project ID: 6355). Permission was granted by Sysselmannen, the governing body in Svalbard, to shoot a sample of 40 fulmars outside of the breeding season for broad range of research purposes, in collaboration with other studies. This method was selected because of the absence of longline fisheries, which normally would provide bycatch individuals. Nor is it feasible to collect beached individuals because of rapid scavenging (e.g. by polar foxes, arctic skuas, glaucous gulls etc.) and the general inaccessibility of beaches. A sample size of 40 has been recommended to quantify plastic ingestion with statistical confidence (Van Franeker and Meijboom 2002).

In Iceland, permission was granted by The Environment Agency of Iceland, Umhverfisstofnun, to shoot a sample of 40 fulmars for scientific purposes. 37 birds were shot from the 13th to the 15th of October 2013, and the remaining three were shot on the 17th of February 2014. For the purposes of this study, they will be treated as a single sample.

Ethics

Sampling was carried out in accordance with ethical guidelines in current Norwegian legislation, and all efforts were made to minimise suffering. Collaboration with other studies (e.g. stable isotope analysis studies and radioactive nucleotide studies by Japanese scientists investigating the fall out from Fukushima, 2011) ensured maximum sampling from sacrificed individuals.

Methods

Fulmars were sampled in Isfjord, Svalbard (78.3°N, 16.1°E) from 21st to 23rd September 2013. Dissections were undertaken at the University Centre in Svalbard, Longyearbyen, following the protocol used by the North Sea monitoring programme (Van Franeker 2004) to determine age and sex as well as morphological characteristics. Samples were collected of the breast muscle, subcutaneous fat, liver, kidney and head (for brain tissue) for future toxicology studies. At no point during dissections were rubber gloves worn in order to avoid plastic contamination of the samples. Dissection tools were rinsed in ethanol between samples to avoid contamination. Stomachs were collected whole, and both the proventriculus and gizzard were rinsed over a 1mm sieve. Stomach plastic contents were characterised and quantified with IMARES (Texel, Netherlands) according to the North Sea monitoring protocol (Van Franeker et al. 2011): plastic pieces were counted and weighed by category on a Sartorius electronic scale accurate to 0.0001 g. Plastics were sorted into industrial plastics: raw plastic pellets produced by plastic manufacturers, and user plastics: all forms of plastic used by consumers, such as fragments of hard plastics, sheets, threads or foams.

Contaminant studies

In 2011, over 200 fulmars that were bycatch victims of longline fisheries were dissected, the stomach plastic contents quantified, and liver tissue sampled. For this study a subsample of 18 of these individuals were used (analytical time constraints prevented any greater number). The 18 samples included nine birds that had no plastic in their stomachs and the nine birds that had the highest amount of plastic in their stomachs (0.27 to 1.42 grams). Because of time constraints, the samples from the Svalbard birds were not used in this study, but will instead be used for future analytical work when multiple tissues can be investigated.

All contaminant analysis was carried out at the Norwegian Institute for Air Research (NILU) in Tromsø, Norway. Methods were followed as per Herzke et al. (2003). A single procedure was used to extract a suite of organic pollutants and metabolites from the tissue, and then remove biological compounds from the sample, such as lipids, during a series of "clean-up" steps. Finally the concentrations of contaminants were calculated and visualised using gas chromatography with mass spectrometry.

Data analysis and presentation

Data were compared to other regions where plastic loading in northern fulmar stomachs has been monitored. Data were provided from Jan Van Franeker for the most recent five year period (2997-2011) in the English Channel, the North Sea (comprising of East England, the Scottish Islands, Belgium, Germany, the Netherlands, Denmark and the North Sea coasts of Sweden and South Norway) and the Faroe Islands, published within the "Save the North Sea" monitoring work (Van Franeker and SNS Fulmar Study Group 2013) as well as for Iceland for 2011 (Kühn and Van Franeker 2012). Summary data for Arctic Canada were also used for comparison (Mallory et al. 2006; Mallory 2008; Provencher et al. 2009). Study locations are given in Figure 1.

All data analyses were carried out using R version 3.1.0. Population averages are presented as the arithmetic mean (unless otherwise stated) using all individuals, including those with no ingested plastic. Data were not normally distributed before or after relevant transformation (Shapiro-Wilk $p<0.05$), therefore non-parametric tests (Mann-Whitney U test or Kruskal-Wallis) were used to compare regional differences in plastic ingestion. In addition, the geometric mean and OSPAR EcoQO performance were calculated to minimise the effect of outliers, as per previous monitoring work (Kühn and Van Franeker 2012).

For the fulmars from Iceland, dissections were carried out in summer of 2014, and thus data are not completely available yet. A preliminary analysis is presented.

Results

As determined from dissections, five out of the 40 fulmars were adults, seven were second year birds (i.e. chicks of 2012) and the remainder were sub-adults (ca. 3-5 or more years old, having never bred before). None of the fulmars sampled in September had bred that summer. Sexes were equally represented with 21 females, and 19 males. All birds were of the arctic "coloured" type (colour phases L, D and DD as in Van Franeker (2004)). No fulmars were of the light plumage colour that dominates in colonies south of the Arctic.

Photo: Geir Wing Gabrielsen

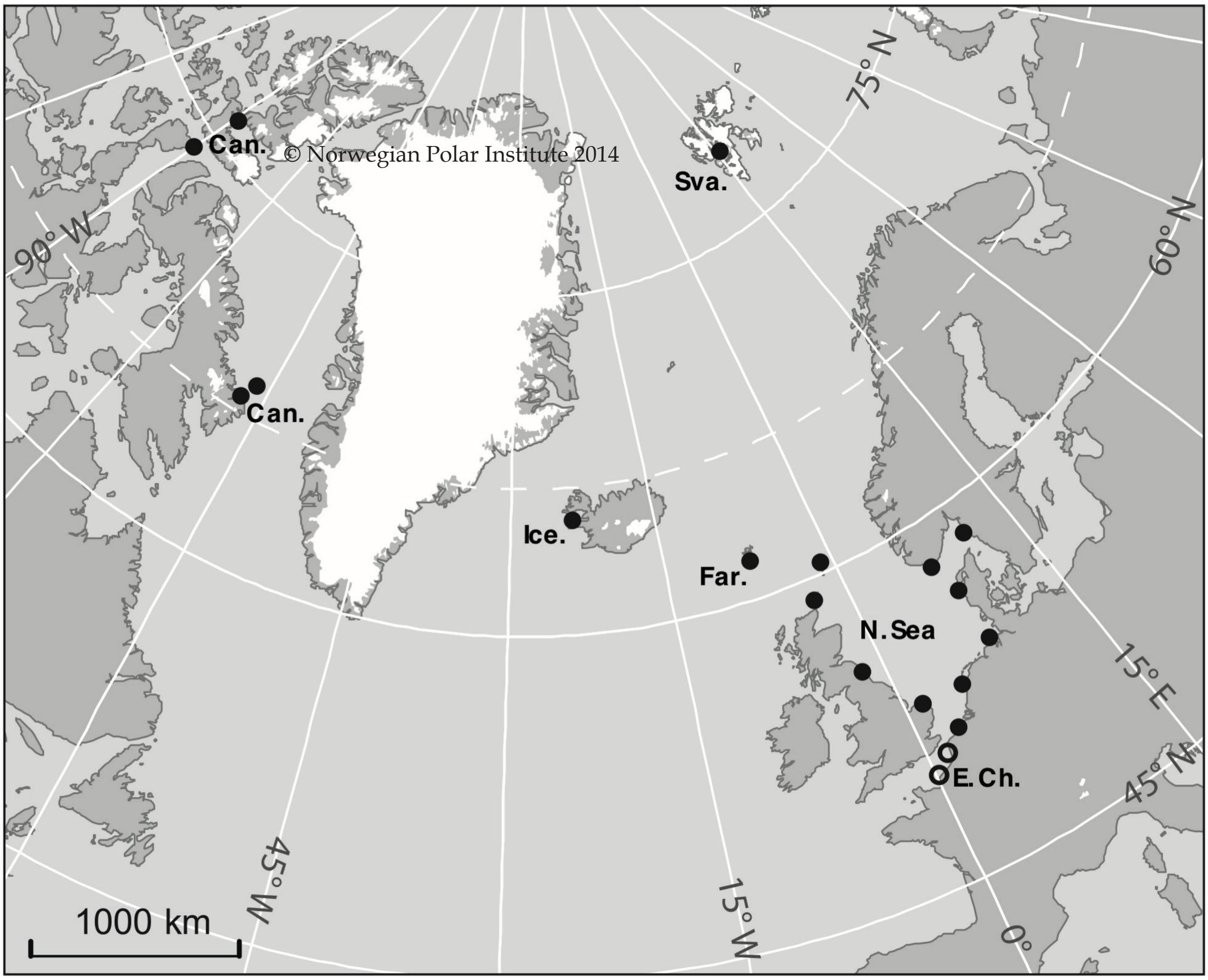

Figure 1
Map showing all study locations used for regional comparison of plastic ingestion by northern fulmars: Svalbard (Sva.), Arctic Canada (Can.), Iceland (Ice.), Faroe Islands (Far.), the North Sea (N. Sea) and the English Channel (E. Ch.; hollow circles). White shading indicates ice cover. Dashed line gives limit of the Arctic Circle around 66˚ 33′ 44′′ N.

Plastic ingestion by fulmars in Svalbard

In Svalbard, 87.5 % of northern fulmars had ingested plastic (i.e. the incidence rate), equating to an average of 15.3 pieces (± s.e. = 5.5, n=40) of plastic per individual, or an average total mass of 0.08 g (± 0.02 g) per individual. The maximum ingested plastic both by number and weight were recorded in the same individual: 200 pieces, weighing 0.4990 g. Industrial plastic pellets made up on average 10.8 % (± 4.5 %) of the mass of all plastic ingested by individual fulmars, the remainder of which was user plastic. Examples of stomach plastic content are given in Figure 2.

In this study we found that in Svalbard, 22.5 % of northern fulmars have ingested ≥ 0.1 g of plastic, which exceeds the level defined by OSPAR as the Ecological Quality Objective for the North Sea (EcoQO; 10 %).

The amount of plastic ingested by fulmars in this study did not differ between male and female birds both in terms of mass ingested and number of pieces (Mann Whitney U-test, $p>0.05$). Similarly, we found no statistical difference in ingested plastic between the different age groups sampled in this study (Mann Whitney U-test, $p>0.05$).

Latitudinal comparison of plastic ingestion

Plastic ingestion was compared to monitoring data from multiple regions in the North-East Atlantic (Figure 1). Overall, amount of plastic ingestion differs significantly between study regions in the North Atlantic (mass and number of pieces; Kruskal-Wallis, $p<0.05$).

From the English Channel northwards to Arctic Canada, there is a decrease in plastic ingestion incidence and mass with latitude (Figure 3). However, plastic ingestion (incidence and mass) is greater in Svalbard than at lower latitudes in Arctic Canada. In addition, there is no difference in the amount of plastic ingested (mass and number of pieces) by northern fulmars from Svalbard and Iceland, (Mann-Whitney U test, $p>0.05$). In Svalbard, the incidence of plastic ingestion was higher (Figure 3A), although fewer individuals had ingested high amounts of plastic (Figure 3B). The average mass of plastic ingested is higher in Iceland (Figure 3C) unless the effect of outliers on the average is reduced: the geometric mean masses of plastic ingested in Iceland and Svalbard are similar (0.020 g and 0.024 g plastic respectively).

Preliminary analysis of Iceland data from 2013

From results in Table 3, plastic ingested by fulmars from Iceland was higher in October of 2013 than in April of 2011 (Kühn and Van Franeker 2012). Indeed, the mass of plastic ingested by fulmars in 2013/4 is significantly higher than in 2011 (Mann-Whitney U test, $p=0.02$).

Combined data are given as used for OSPAR monitoring, which studies changes based on 5-year averages. The combined Iceland data (Table 3) are still comparable to the data from Svalbard (Tables 1 and 2) and there is still no difference in the mass of ingested plastic between the two regions (Mann-Whitney U test, $p=0.41$).

Contaminant analysis

Organochlorine contaminants extracted from liver tissues of fulmars from the Faroe Islands with either high or low levels of plastic ingestion are presented in Table 4, and visualised in Figures 4 and 5.

Mean contaminant concentrations are slightly higher when plastic ingestion is high, compared to when plastic ingestion was absent, in all groups except Mirex (Table 3). In addition, there seems to be a much greater variation in tissue contaminant loads in the group with high plastic ingestion (Figs 4 & 5). None of these differences between contaminants loads, however, were statistically significant when tested against mass and number of plastic pieces ingested (ANOVA, $p>0.05$). This applies both to individual contaminant congeners and groups of contaminant congeners.

Contaminants tested for and not found were HCHs (α-, β- and γ-hexachlorocyclohexane), heptachlor, *o,p'*-DDT and PBDEs -154 and -183.

Discussion

This study has successfully achieved a baseline value of plastic ingestion that will facilitate the future detection of changes in marine plastic pollution and potential impacts of increased commercial activity. In addition, slight contaminant uptake from plastic into body tissues of northern fulmars highlights the need for further studies to advance this field of study.

(a) L-R: Industrial pellets; probably industrial; fragments; sheets; threads; foam

(b) L-R: Industrial pellets; fragments; sheets; foam

(c) L-R: Fragments; sheets; threads; (including a thread ball)

Figure 2
Stomach plastic contents of three individual northern fulmars from Svalbard, 2013. Scale bars indicate 1 cm.

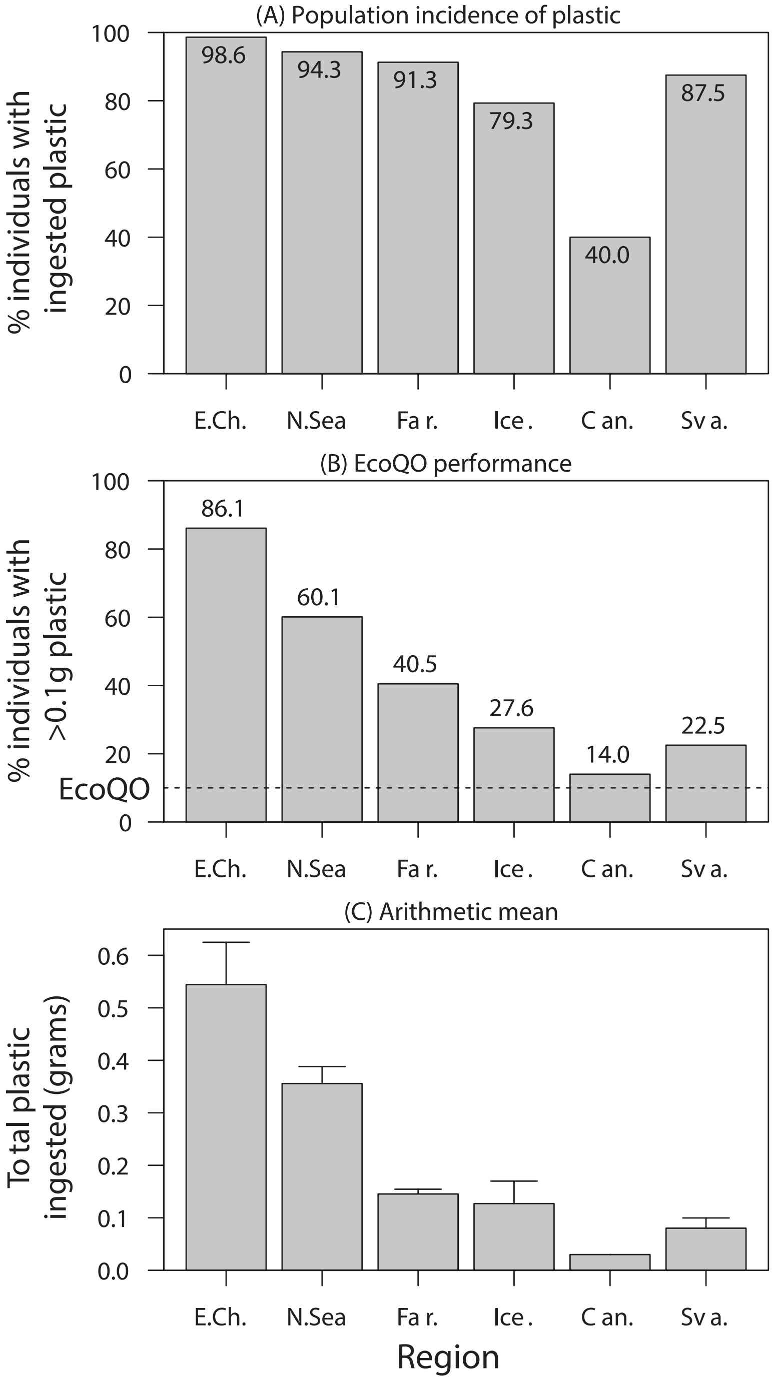

Figure 3
Regional differences in plastic ingestion by northern fulmars in the North Atlantic, according to (A) population incidence of plastic ingestion, (B) population incidence of over 0.1 gram of plastic, dashed line shows the Ecological Quality Objective (EcoQO) defined by OSPAR, and (C) arithmetic mean mass, error bars show standard error. Data have been collated for the English Channel (E.Ch., 2007-2011, n=72), the North Sea (N.Sea, 2007-2011, n=58), the Faroe Islands (Far., 2007-2011, n=699) (Van Franeker and SNS
Fulmar Study Group 2013), Iceland (Ice., 2011, n=58) (Kühn and Van Franeker 2012), Svalbard (Sva., 2013, n=40) (this study) and Arctic Canada (Can., 2002-2009, n=169) (Mallory et al. 2006; Mallory 2008; Provencher et al. 2009).

Table 1
Plastic ingested by northern fulmars from Svalbard, according to age. Averages are given in number (n) or grams (g) of plastic per individual ± standard error. Maximum ingested plastic is by a single individual in the age class. EcoQO performance gives the percentage of the population with more than 0.1g of ingested plastic.

	Sample Size	Incidence (%)	Mean no. of plastic pieces (g ± se)	Arithmetic mean mass (n ± se)	Max ingested plastic (g)	Geometric mean mass	Eco QO performance (%)
All	40	87.5	15.32 ± 5.51	0.08 ± 0.02	0.50	0.023	22.5
Adults	5	80	4.20 ± 2.52	0.04 ± 0.02	0.12	0.013	20
Non-adults	35	89	16.91 ± 6.25	0.09 ± 0.02	0.50	0.025	22.9

Table 2
Plastic ingested by northern fulmars from Svalbard (n=40), according to type of plastic. Averages are given in number (n) or grams (g) of plastic per individual ± standard error. Maximum ingested plastic is by a single individual.

	Incidence (%)	Mean number. of plastic pieces (n ± se)	Arithmetic mean mass (g ± se)	Max ingested plastic (g)	Geometric mean mass
ALL PLASTICS	87.5	15.32 ± 5.51	0.080 ± 0.02	0.499	0.023
Industrial plastic	23	0.45 ± 0.17	0.006 ± 0.00	0.051	0.001
User plastic	83	14.88 ± 5.41	0.074 ± 0.02	0.490	0.018
Sheet-like	35	1.53 ± 0.53	0.004 ± 0.00	0.071	0.001
Thread-like	45	1.90 ± 0.61	0.018 ± 0.01	0.318	0.002
Foamed	10	0.68 ± 0.45	0.000 ± 0.00	0.008	0.000
Fragments	80	10.72 ± 4.60	0.049 ± 0.02	0.480	0.013
Other	5	0.05 ± 0.03	0.003 ± 0.00	0.082	0.000

Table 3
Preliminary comparison of plastic ingested by fulmars from Iceland in 2011 and 2013/4 (2013 n=37; 2014 n=3), as well as combined. Averages are given in grams (g) of plastic per individual ± standard error. Maximum ingested plastic is by a single individual. EcoQO performance gives the percentage of the population with more than 0.1g of ingested plastic.

	Sample size	Incidence (%)	Arithmetic mean mass (g ± se)	Max ingested plastic (g)	Geometric mean mass	EcoQO performance
2011	58	79	0.13 ± 0.04	1.97	0.02	27.6
2013/4	40	90	0.12 ± 0.02	0.58	0.05	47.5
Combined	98	84	0.13 ± 0.03	1.97	0.03	35.7

Table 4
Stomach plastic mass and liver organochlorine concentrations of 18 northern fulmars from the Faroe Islands (longline victims in 2011) with either no ingested plastic at time of dissection (Absent; n=9) or high levels (0.3-1.4g) of ingested plastic (High; n=9). Mean mass of plastic (g) or contaminant concentration (ng g-1) ± standard error.

	Plastic	∑All [a]	∑PCBs [b]	∑PBDEs [c]	∑DDTs [d]	HCB	∑Chlordanes [e]	Mirex
Absent	0.00 ± 0.00	1444 ± 174	674 ± 81	1.4 ± 0.3	351 ± 65	23 ± 1.7	182 ± 20	31 ± 4.4
High	0.63 ± 0.12	1644 ± 357	753 ± 194	2.1 ± 0.6	398 ± 86	24 ± 3.3	221 ± 51	24 ± 5.2

a Sum of all organochlorines extracted
b ∑PCBs: sum of polychlorinated biphenyl congeners -28, -52, -99, -101, -105, -118, -138, -153 & -180
c ∑PBDEs: sum of polybrominated diphenyl ether congeners -28, -47, -99, -100, -138 & -153
d ∑DDTs: sum of p,p'-DDT, o,p'-DDE, p,p'-DDE & p,p'-DDD
e Chlordanes: sum of oxychlordane, cis- and trans-chlordane & cis- and trans-nonachlor

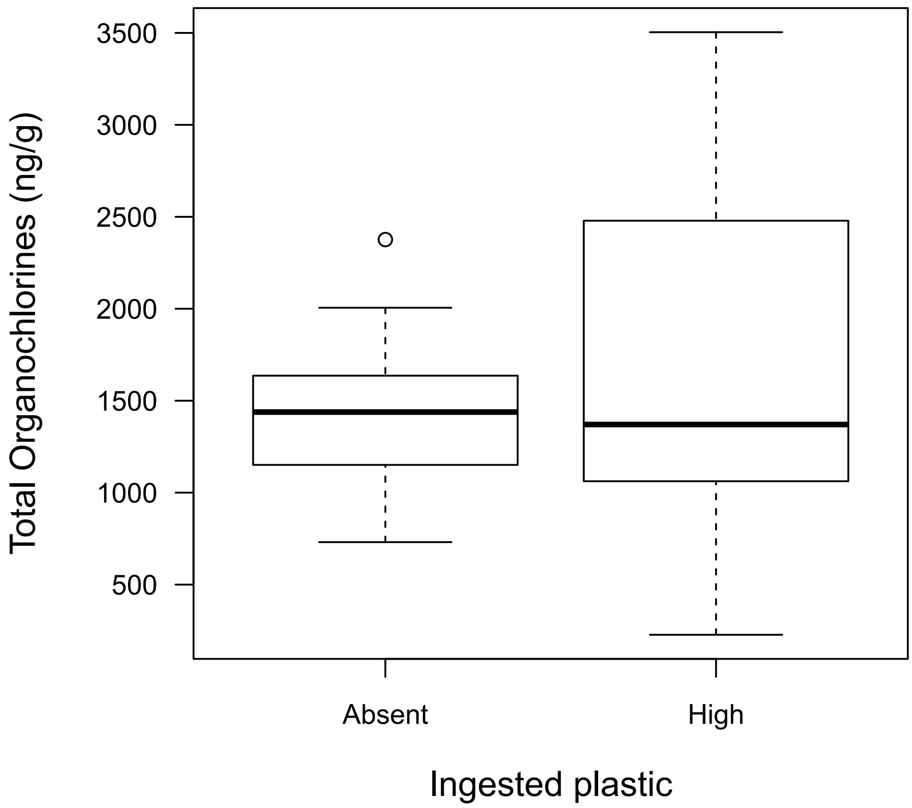

Figure 4
Liver contaminant loads of 18 northern fulmars from the Faroe Islands (longline victims in 2011) with either no ingested plastic at time of dissection (Absent; n=9) or high levels (0.3-1.4g) of ingested plastic (High; n=9). The total organochlorines are the sum of PCBs (polychlorinated biphenyl congeners -28, -52, -99, -101, -105, -118, -138, -153 & -180), PBDEs (polybrominated diphenyl ether congeners -28, -47, -99, -100, -138 & -153), DDTs (p,p'-DDT, o,p'-DDE, p,p'-DDE & p,p'-DDD), HCB (hexachlorobenzene), chlordanes (oxychlordane, cis- & transchlordane and cis- & trans-nonachlor) and Mirex.

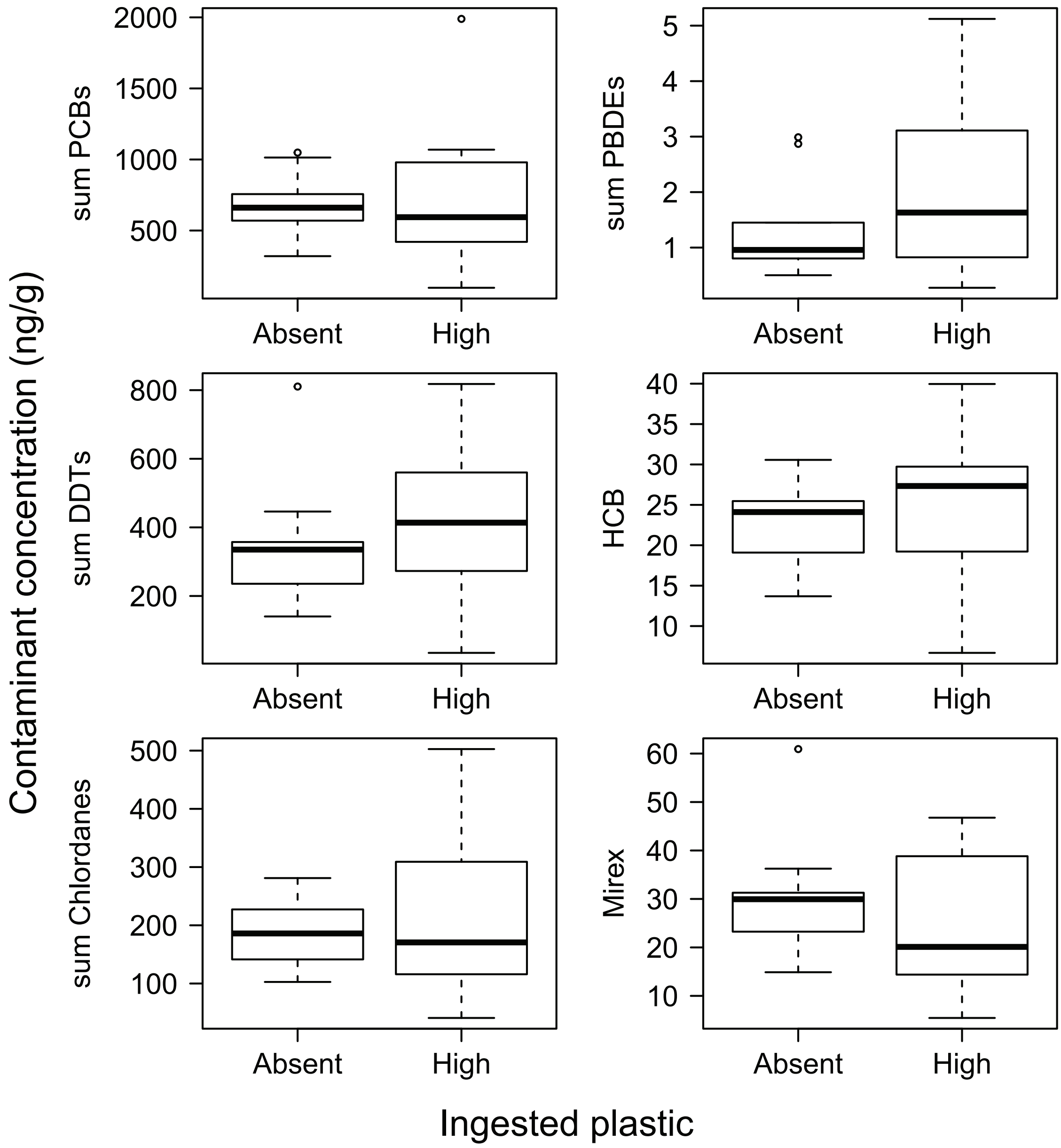

Figure 5
Liver contaminant loads of 18 northern fulmars from the Faroe Islands (longline victims in 2011) with either no ingested plastic at time of dissection (Absent; n=9) or high levels (0.3-1.4g) of ingested plastic (High; n=9). Results are given for PCBs: the sum of polychlorinated biphenyl congeners -28, -52, -99, -101, -105, -118, -138, -153 & -180, PBDEs: the sum of polybrominated diphenyl ether congeners -28, -47, -99, -100, -138 & -153, DDTs: the sum of p,p'-DDT, o,p'-DDE, p,p'-DDE & p,p'-DDD, HCB: hexachlorobenzene, chlordanes: sum of oxychlordane, cis- and trans-chlordane & cis- and trans-nonachlor, and Mirex.

Plastic ingestion by fulmars in Svalbard

Unfortunately data for the 1980s (Gjertz et al. 1985; Lydersen et al. 1985; Van Franeker 1985; Lydersen and Gjertz 1989) do not allow a proper analysis for possible changes over time in ingested quantities of plastic. The data in these early publications were only by number of items and appear contradictory between information for Spitsbergen (29% individuals with plastic (n=62), and an average of 0.75 pieces (n=20) in Gjertz et al. (1985), Lydersen et al. (1985) and Lydersen and Gjertz (1989)) and nearby Bjørnøya (82% individuals with plastic and an average of 4.5 pieces (n=22) in Van Franeker (1985)). Furthermore, from North Sea fulmar data, it appears that sizes of plastic particles have changed over time: particles have become smaller (Van Franeker and Meijboom 2002), with currently different number to mass ratios than in the 1980s, which complicates comparisons.

As expected from North Sea data in Van Franeker et al. (2011), data in Table 1 do suggest differences between young and adult birds but the small sample size for adults prevents proper statistical evaluation in our case. However, adults and non-adults are similar in EcoQO performance and allow combined further discussion in this baseline. Later studies need to provide more insight in potential age differences and their implications for interpretation of monitoring data.

Plastic ingestion by northern fulmars in Svalbard does not follow the established trend of a decrease with latitude or distance from human marine impact (using measure of impact from Halpern et al. (2008)). Instead, plastic ingestion by fulmars from Svalbard is higher than expected. This study reports the highest levels of plastic ingestion reported in an Arctic colony of northern fulmars. Incidence of plastic ingestion, and mass of ingested plastic, are considerably higher than those recorded at lower latitudes in Arctic Canada (Mallory et al. 2006; Mallory 2008; Provencher et al. 2009). In addition, levels of plastic ingestion in Svalbard are comparable to those in Iceland, approximately 2,000km further south, contrary to the expected latitudinal decrease.

If distance from human impact was the primary driver of plastic ingestion, as previously proposed (Kühn and Van Franeker 2012), levels of plastic ingestion in fulmars from Svalbard would be expected to fall between those in Iceland and Arctic Canada. Indeed, as anticipated, levels in Svalbard are higher than Arctic Canada, where study sites are more remote from population centres. Likewise, compared to the North Sea region (Van Franeker et al. 2011), both lower ingestion amounts and higher proportions of user plastics in Svalbard reflect distance from industry and commercial shipping, in accordance with previous theory. However, Svalbard is more remote from human impact than Iceland (Halpern et al. 2008), and yet plastic ingestion amounts in fulmars from the two locations are similar, indicating a need for alternative or additional hypotheses.

The cause of elevated levels of plastic ingestion in Svalbard is uncertain, and therefore a key knowledge gap for future research has been identified. Transportation of plastic from outside of the Arctic by surface water currents is a likely explanation. Currents along the Norwegian coast may carry floating debris from the polluted North Sea up to the Barents Sea and Svalbard, thus increasing plastic ingestion levels despite the absence of dense population centres in the region. Van Sebille et al. (2012) hypothesised that converging water currents actually result in an oceanic gyre in the Barents Sea, where plastic litter would accumulate, however this is yet to be proven. Alternatively, ingested plastic may originate in or around the Barents Sea, either from the southern Barents Sea fishing fleet (Humphries and Huettmann 2014) or potential release during periods of sea ice melt (Obbard et al. 2014). To confirm or reject these hypotheses would be a useful study for the future, and will help to identify how to mitigate plastic in the Arctic.

The high levels of ingested plastic observed in Svalbard not only highlight the risk to seabirds from plastic pollution, but may also be a considered as a general warning of effects of plastic litter in the Arctic. Floating plastic debris may act as a transport vector to the Arctic for both pollutants (Zarfl and Matthies 2010) and invasive species (Barnes 2002) – both may act as important stressors with threats to biodiversity, particularly under climate warming scenarios (Serreze et al. 2007). Compounds within the plastics may have negative consequences on both wildlife and human health in the region (Oehlmann et al. 2009).

High prevalence of plastic litter in the Arctic, outside of territorial waters, emphasises the need for international mitigation of plastic litter at source, as well as strict enforcement of legislation for commercial activity in the region.

Plastic ingestion by fulmars in Iceland

From preliminary analysis, a difference in mass of plastic ingested by fulmars during the two study periods (Kühn and Van Franeker (2012) and this study) indicates that there may be annual variation in plastic ingestion that is currently unknown. This highlights a valuable area for future study.

Studies in Iceland in future years will enable a more detailed analysis of possible trends of plastic ingestion over time in the region. Long-term monitoring in the North Sea shows the importance of studying trends using 5-year averages to minimise the influence of anomalies (Van Franeker and SNS Fulmar Study Group 2013).


Photo: Geir Wing Gabrielsen

Nevertheless, this study has provided a useful contribution to monitoring of marine plastic litter in Iceland.

Despite differences within the Iceland data set, all discussions in the previous section regarding plastic ingestion in Svalbard are still valid. There continues to be no difference between plastic ingestion in Svalbard and Iceland in preliminary analysis. Finalised results from the current study will enable validation and more detailed analysis.

Contaminant uptake from ingested plastic

Organic contaminant levels in wildlife typically vary according to differing metabolism, reproductive and nutritional status, as well as variation in feeding habits, such as trophic position or whether the species or individual has previously been feeding in an area of higher or lower pollution (Borgå et al. 2001; Finkelstein et al. 2006). Although not conclusive, the results of this study support previous hypotheses that contaminant uptake can also occur via plastic ingestion (Colabuono et al. 2010; Tanaka et al. 2013). This study shows consistently increased variation in tissue contaminant load across different groups of organochlorines with higher plastic ingestion. These may be adsorbing to the surface of the plastic pieces whilst in seawater, particularly the pesticides (DDTs and Mirex, for example) or may be leaching from inside the plastic pieces, such as PBDE flame retardants (Tanaka et al. 2013).

These results highlight the need for further study in this field. A greater sample size and testing for additional compounds will allow more definitive conclusions. Contaminants tested in this study have proven negative effects on organisms (Sagerup et al. 2000; Bustnes et al. 2001; Bustnes et al. 2003; Sagerup et al. 2009; Nøst et al. 2012; Verreault et al. 2013; Sagerup et al. 2014). The study species here, northern fulmars, may not be experiencing population effects of plastic ingestion at present. However, these results may be considered an indicator of the potential harm of plastic pollution in the Arctic, and indeed elsewhere, to more vulnerable species.

Conclusions

This study provides a valuable baseline for plastic litter needed to monitor the future impacts of commercial activity in the Arctic. Furthermore, high levels of plastic ingestion in a seabird breeding area at great distance from human impact highlight the need for urgent mitigation of plastic pollution in the Arctic as well as implementation of strict regulation for future commercial activity. This need is amplified by the potential harm of plastic pollution caused by contaminant uptake into tissues.

Further research will allow a greater understanding of the effects of plastic litter on arctic wildlife.

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