SPFO-Report: 952/2006
TA-number: 2175/2006
ISBN-number: 82-7655-289-7

Nona- and deca- brominated diphenylethers in seabird eggs from Northern Norway and Svalbard

Report 952/2006

Photo: Hallvard Strøm (Norwegian Polar Institute). Chicks and eggs of glaucous gull.

Lisa B. Knudsen, Anuschka Polder, Siri Føreid, Elisabeth Lie, Geir W. Gabrielsen, Rob Barrett, Janneche Utne Skåre

October 2006

1. PREFACE

This project determined the levels of nona-brominated flame retardants (BDE-206, BDE-207 and BDE-208)) and deca-BDE (BDE-209) in seabird eggs from Northern Norway and Svalbard. Eggs of herring gulls (*Larus argentatus*), Atlantic puffins (*Fratercula arctica*) and black-legged kittiwakes (*Rissa tridactyla*) from Røst and Hornøya (Northern Norway) were collected in 1983, 1993 and 2003, and of glaucous gulls (*Larus hyperboreus*) from Bjørnøya (Svalbard) in 1997 and 2002.

The present study was a collaboration between the Norwegian Polar Institute, Tromsø University Museum, the Norwegian School of Veterinary Science and National Veterinary Institute, Norway. Funding was provided by the Norwegian Pollution Control Authority (SFT).

The present study is part of an ongoing monitoring project commenced in 1972 assessing contaminant burdens, as well as spatial and temporal trends, of a suite of halogenated organic contaminants (HOCs) and mercury in seabird eggs (Fimreite et al., 1974; Barrett et al., 1985; 1996). In 2005 a SFT report on BDE-28, BDE-47, BDE-100, BDE-99, BDE-154, BDE-153, BDE-183, BDE-209, α- hexabromocyclododecane (HBCD), β-HBCD, γ-HBCD, total-HBCD and mercury, using the same eggs as in the present study, were completed (Knudsen et al., 2005).

Norwegian School of Veterinary Science, October 2006

Lisa Bjørnsdatter Knudsen
Research Assistant, Norwegian School of Veterinary Science

Anuschka Polder
Head Engineer, Norwegian School of Veterinary Science

Siri Føreid
Engineer, Norwegian School of Veterinary Science

Elisabeth Lie
PostDoc, Norwegian School of Veterinary Science

Geir Wing Gabrielsen
Research Leader, Ecotoxicology Programme, Norwegian Polar Institute

Rob Barrett
Associate Professor, Tromsø University Museum

Janneche Utne Skåre
Professor, Norwegian School of Veterinary Science/National Veterinary Institute, Norway

2. CONTENTS

3. SAMMENDRAG

Det har til nå vært lite informasjon om nivåer av nona- og deka- bromerte difenyletere i frittlevende dyr. Formålet med denne studien var å kartlegge nivåer av nona-bromerte difenyletere (BDE-206, BDE-207, BDE-207) og deka-BDE (BDE-209) i sjøfuglegg. Egg fra gråmåke (*Larus argentatus*), lundefugl (*Fratercula arctica*) og krykkje (*Rissa tridactyla*) ble samlet inn i 1983, 1993 og 2003 på Røst og Hornøya (Nord-Norge). Egg fra polarmåke (*Larus hyperboreus*) ble samlet inn i 1997 og 2002 på Bjørnøya (Svalbard). Analysene av BDE-209 ble foretatt i fjor og finnes også i SPFO rapport 942/2005.

Nona og deka-BDE
Nivåer av nona-BDEer og BDE-209 fra ikke målbare verdier til deler per milliard (ng/g) ble rapportert i denne studien. Nona-BDE and BDE-209 ble funnet i henholdsvis 26 og 11 av totalt 96 analyserte egg. Konsentrasjoner av samtlige nona-BDEer og BDE-209 ble kun funnet i fem egg.

Maksimum konsentrasjoner av nona-BDEer i denne studien var høyere enn det som tidligere har vært rapportert i mort (*Rutilus rutilus*), abbor (*Perca fluviatilis*) og gjedde (*Esox lucius*) fra Østersjøen. Videre var de høyeste nivåer av BDE-209 i denne studien høyere enn BDE-209 i ugleegg fra Belgia og i tidligere undersøkte egg fra polarmåke fra Bjørnøya. Nivåene av BDE-209 var imidlertid lavere enn tidligere rapporterte verdier i vandrefalkegg (*Falco peregrinus*) fra Grønland og Sverige.

Ingen signifikante geografiske trender eller tidstrender av nona-BDE og BDE-209 ble funnet i denne studien. Det var imidlertid få prøver og lave konsentrasjoner i denne studien.

Kongenermønster
Det relative bidraget av de individuelle nona-BDE kongenerne og BDE-209 til ∑PBDE (BDE-28, 47, 100, 99, 154, 153, 183, 206, 207, 208 and 209) varierte betydelig. BDE-209 bidro med 0 – 27.8 % til ∑PBDE. BDE-208, BDE-207 and BDE-206 bidro med henholdsvis 0 – 2.4 %, 0 – 13.1 % og 0 – 3.5 % til ∑PBDE. Vi ser ingen klare sammenhenger i kongener mønsteret av deka-BDE og nona-BDE i denne studien. Det er imidlertid viktig å påpeke at konsentrasjoner av samtlige nona-BDEer og BDE-209 kun ble rapportert i fem egg.

4. SUMMARY

There are information gaps regarding levels of nona- and deca- brominated diphenyls in wildlife. The aim of the present study was to determine levels and temporal trends of nona-brominated diphenylethers (BDEs) (BDE-206, BDE-207 and BDE-208) and deca-BDE (BDE-209) in seabird eggs. Eggs were collected from herring gulls (*Larus argentatus*), Atlantic puffins (*Fratercula arctica*) and black-legged kittiwakes (*Rissa tridactyla*) in 1983, 1993 and 2003 at Røst and Hornøya (Northern Norway). Furthermore, eggs of glaucous gulls (*Larus hyperboreus*) were collected in 1997 and 2002 at Bjørnøya (Svalbard). The analyses of BDE-209 was done in 2005, and the results can be found in the SPFO report 942/2005.

Nona-BDEs and deca-209
The nona-BDEs and BDE-209 in the present study were in the range non-detect to the parts per billion level. Of a total of 96 eggs 26 eggs were reported with nona-BDEs and 11 eggs with deca-BDE concentrations above the maximum concentrations detected in the procedural blanks. Concentrations above the procedural blanks for all three nona-BDE congeners and BDE-209 were only found in five eggs.

The maximum concentrations of nona-BDEs in the present study were higher than that previously reported in roach (*Rutilus rutilus*), perch (*Perca fluviatilis*) and pike (*Esox lucius*) from the Baltic Sea. Furthermore, the highest levels of BDE-209 were higher than that reported for eggs of little owl from Belgium and from earlier studies of the glaucous gull from Bjørnøya. Levels of BDE-209 in the present study were lower than those reported for peregrine falcon (*Falco peregrinus*) eggs from Greenland and Sweden.

No clear spatial and temporal trends of nona-BDEs and BDE-209 were found in the present study. However, few samples and low contaminant levels make it difficult to reveal trends.

Congener pattern
The relative contribution of the individual congeners to ∑PBDEs was reported to be highly variable. For example, BDE-209 contributed between 0 – 27.8 % to ∑PBDEs (BDE-28, 47, 100, 99, 154, 153, 183, 206, 207, 208 and 209), whereas BDE-208, BDE-207 and BDE-206 contributed between 0 – 2.4 %, 0 – 13.1 %, 0 – 3.5 % to ∑PBDEs, respectively. We could not find any clear relationship in the congener pattern of nona-BDEs and BDE-209 in the present study. However, only five eggs were reported with concentrations of all nona-BDEs and deca-BDE.

5. BACKGROUND

Three major technical PBDE mixtures, i.e. penta, octa, and decaBDEs, have been produced and used in electronics and textiles to prevent fire (BSEF, 2006). The total global market demand of the three technical BDEs was 67,440 tonnes in 2001 (BSEF, 2006) (Figure 1).

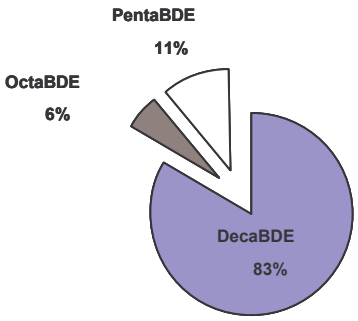

Figure 1. Total percentage global marked demand of penta, octa and decaBDEs.

The technical PBDE-mixtures, i.e., pentaBDE, octaBDE and decaBDE, mainly consist of penta-BDEs, octa-BDEs or deca-BDE, respectively, but also contain low levels of other BDEs (Figure 2).

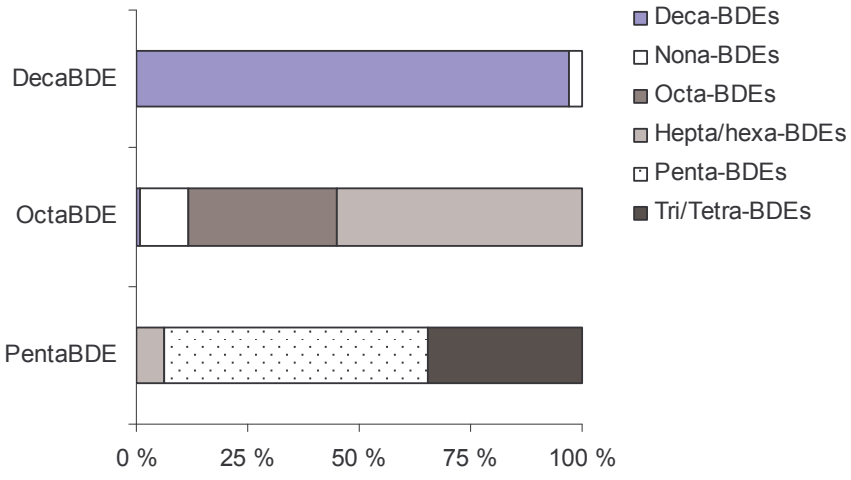

Figure 2. General composition of technical mixtures of penta, octa, and decaBDEs (de Wit et al., 2002).

PBDEs are additive flame retardants, e.g., they do not form chemical bounds, and can as such continuously leak out during manufacture and from products in use and as waste. As a result, PBDEs have been detected ubiquitously throughout the environment, including in seabird eggs from species such as the glaucous gull (*Larus hyperboreus*) from Bjørnøya

(Verreault et al., 2004), and the peregrine falcon (*Falco peregrinus*) from Greenland (Vorkamp et al., 2005). The PBDEs, and especially the penta and octa-BDEs, have been shown to be toxic (Birnbaum and Staskal, 2004). In concert with this, the industry in the U.S. voluntarily discontinued the production of penta and octaBDEs in 2005 (Great Lakes Chemical Corporation, 2005). In Japan, no use of commercial pentaBDE (called tetraBDE in Japan, but estimated to be the same product as commercial pentaBDE in Europe and USA) and octaBDE was recognized after 1991 and 2000, respectively (Watanabe and Sakai, 2003). Furthermore, the countries within the European Union banned the pentaBDEs and octaBDEs in 2004 (Cox and Efthymiou, 2003). There is currently no restriction in the production of decaBDE, reaching 56,418 tonnes in 2003 (BSEF, 2006).

As shown in Figure 2, commercially manufactured decaBDE is almost entirely composed of 2,2',3,3',4,4',5,5',6,6'-deca-BDE (BDE-209), but also contains low levels of 2,2',3,3',4,4',5,5',6-nona-BDE (BDE-206), 2,2',3,3',4,4',5,6,6'-nona-BDE (BDE-207) and 2,2',3,3',4,5,5',6,6'-nona-BDE (BDE-208) (de Wit, 2002). There has been very little information about levels and effects of nona-BDEs and BDE-209 in wildlife. Until recently, higher brominated flame retardants were considered to be poorly bioavailable to wildlife organisms (e.g., Hardy, 2002), but nona-BDEs and BDE-209 have recently been reported in humans and wildlife species (Jakobsson et al., 2002; Burreau et al., 2004; Lindberg et al., 2004; Verreault et al., 2005; Sørmo et al., in press) indicating that some absorption occur.

Nona-BDEs not only originate from technical decaBDE mixtures, but may also be formed during debromination of BDE-209 in an anaerobic environment (Birnbaum and Staskal, 2004; Gerecke et al., 2005), or in animals (Kierkegaard et al., 1999; Mörck et al., 2003; Thuresson et al., 2005). The nona-BDEs and deca-BDE are suggested to be further degraded to octa-BDEs (Gerecke et al., 2005; Thuresson et al., 2005) (Figure 3). Nevertheless, the metabolism of BDE-209 is probably more complex than illustrated above and in Figure 3. For example, hydroxylated metabolites have been reported in experimental studies feeding rats with BDE-209 (Mörck et al., 2003).

As the production and use of penta and octaBDE are being phased out, higher brominated BDEs, and in particular deca-BDE, has become a significant environmental issue and a subject for ongoing discussions (EU, 2005). In the present study we addressed the information gap regarding levels of nona-BDE in wildlife by analyzing these compounds in eggs of herring gulls (*Larus argentatus*), Atlantic puffins (*Fratercula arctica*), black-legged kittiwakes (*Rissa tridactyla*) and glaucous gulls. Furthermore, the congener-specific accumulation of nona-BDEs and BDE-209 were assessed. Sampling was conducted in 1983, 1993/1997 and 2002/2003 at Røst, Hornøya (Northern Norway) and Bjørnøya (the Barents Sea) to evaluate possible regional and temporal trends. Eggs of seabirds have previously been recognized as appropriate for long-term monitoring of contaminants in the marine environment (Barrett et al., 1985; 1996; Braune et al., 2001).

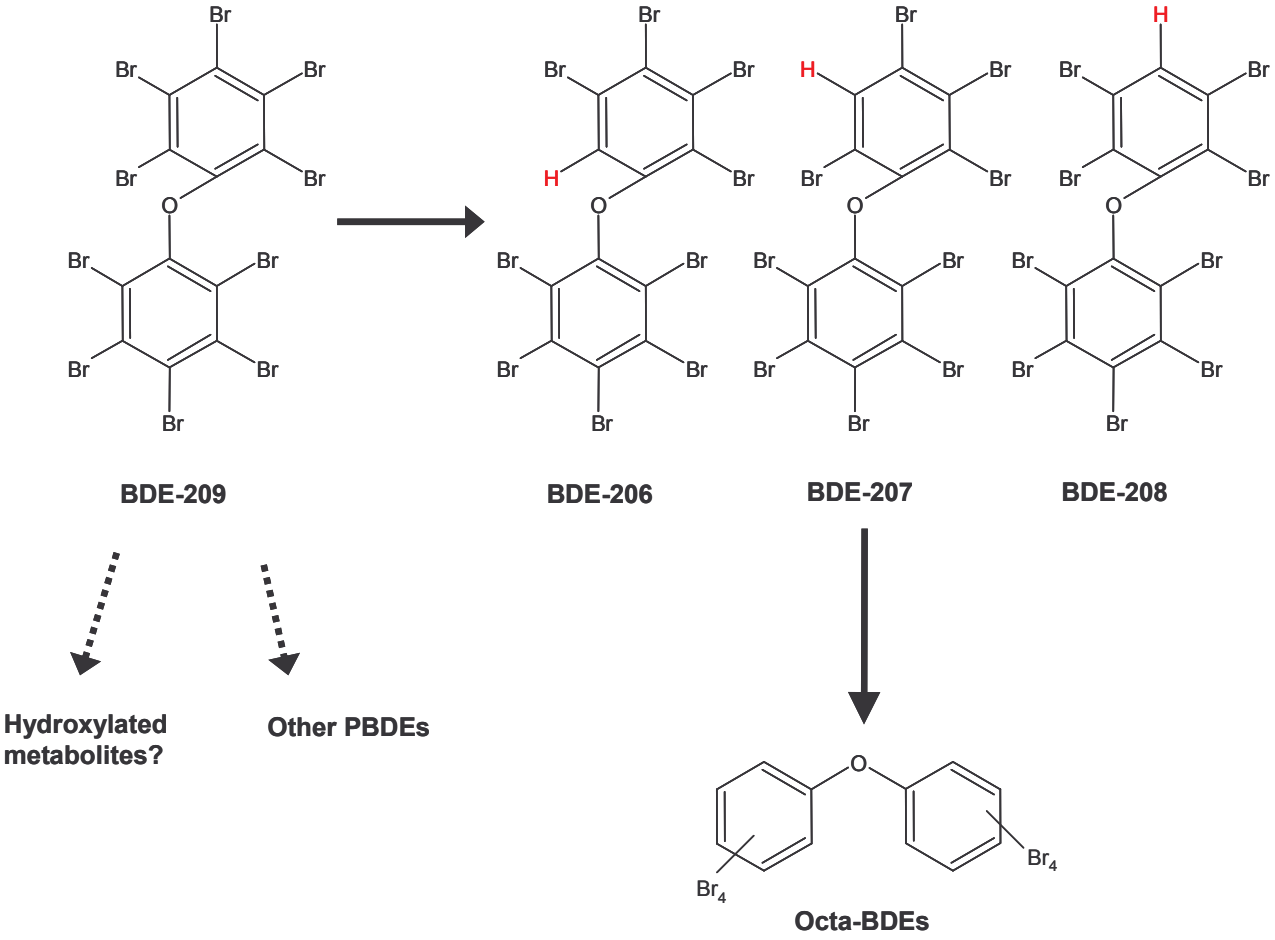

Figure 3. Simplified degradation scheme of BDE-209.

6. MATERIALS AND METHODS

6.1 SPECIES STUDIED AND FIELDWORK

A total of 96 fresh eggs were collected from herring gulls, black-legged kittiwakes, Atlantic puffins and glaucous gulls in 1983, 1993/1997, and 2002/2003 (Table 1). The Atlantic puffins and black-legged kittiwakes feed mainly on fish such as herring and capelin, whereas the herring gulls and glaucous gulls have a more varied and opportunistic diet composed of fish, seabird chicks and eggs (Anker-Nilssen et al., 2000). Bjørnøya, Hornøya and Røst (Figure 4) were selected as sampling areas to assess if there were a north-south (spatial) gradient in the dataset. Due to the lack of herring gull eggs from Røst in 1983, herring gull eggs from Hekkingen (Figure 4), an island a little further north, were included instead.

After sampling, the individual eggs were homogenized and kept frozen (-20°C) until analysis. All the eggs were analyzed in 2005. The egg homogenates may have dried slightly during storage, consequently, concentrations should be reported only on a lipid normalized (lw) basis.

Table 1. Number of eggs analyzed for nona-BDEs and BDE-209 in 2005 from herring gulls (H. gull), black-legged kittiwakes, Atlantic puffins and glaucous gulls collected from Røst, Hekkingen (in brackets), Hornøya and Bjørnøya in 1983, 1993/1997 and 2002/2003.

	Røst/(Hekkingen)			Hornøya			Bjørnøya	
Species	**1983**	**1993**	**2003**	**1983**	**1993**	**2003**	**1997**	**2002**
H. gull	(5)	5	5	5	5	5	-	-
Puffin	5	4	5	5	5	5	-	-
Kittiwake	5	5	5	5	5	5	-	-
Glaucous gull	-	-	-	-	-	-	4*	3

* 4 sub-samples of 201 pooled eggs.

Figure 4. Map showing the localities from which eggs were collected in 1983, 1993/1997 and 2002/2003.

6.2 CHEMICAL ANALYSES

6.2.1 DETERMINATION OF NONA-BDES AND BDE-209

The chemical analyses of PBDEs were done at the Laboratory of Environmental Toxicology at the Norwegian School of Veterinary Science in Oslo. BDE-209 was analyzed in 2005 and the results presented in Knudsen et al. (2005).

The content of the eggs were homogenized in a food blender. The egg homogenates (~3 g) were weighed in 80 mL centrifugation tubes and spiked with the internal standards of ^{13}C-BDE-209. The lipids were extracted twice using cyclohexane and acetone (3:2) and an ultrasonic homogenizer. The supernatants of both extractions were merged and concentrated to about 1 mL using a Zymark® evaporation system at 40°C, and by a gentle flow of nitrogen. The concentrated lipid extracts were quantitatively transferred to volumetric flasks, and the volume adjusted to 5 mL with cyclohexane. The lipid determination was done gravimetrically using 1 mL aliquot of the sample. For cleanup (i.e. removal of lipids) the rest of the lipid extracts were treated twice with ultra clean (purity 98.8%) concentrated H_2SO_4 and concentrated to about 0.3 mL using a gentle flow of nitrogen. The sample concentrates were transferred to dark gas chromatography (GC) vials to minimize the degradation. For more details of the methods used for extraction and cleanup see Murvoll et al. (2005). See Table 2 for product specifications.

A programmable temperature vaporization (PTV) injector was used to inject the nona-BDEs and BDE-209. A PTV injector made it possible to inject large volumes of the

sample, whereas a short column reduced the time in the gas chromatograph (GC), i.e., reduced the degradation. 1.0 μL were injected for analysis on a gas chromatograph (GC)-mass spectrometry (MS) quadrupol detector. The separation and identification of the nona-BDEs and BDE-209 was performed by a 10 m long DB-5-MS column. Nona-BDEs and BDE-209, in addition to ^{13}C-BDE-209, were monitored using electron capture negative ionization (ECNI) in selected ion monitoring (SIM) at *m/z* 484.5 and 486.5, and 494.5 and 498.5, respectively. See Table 3 and Figure 5 for details of the temperature programme and the operations of the mass spectrometer, respectively.

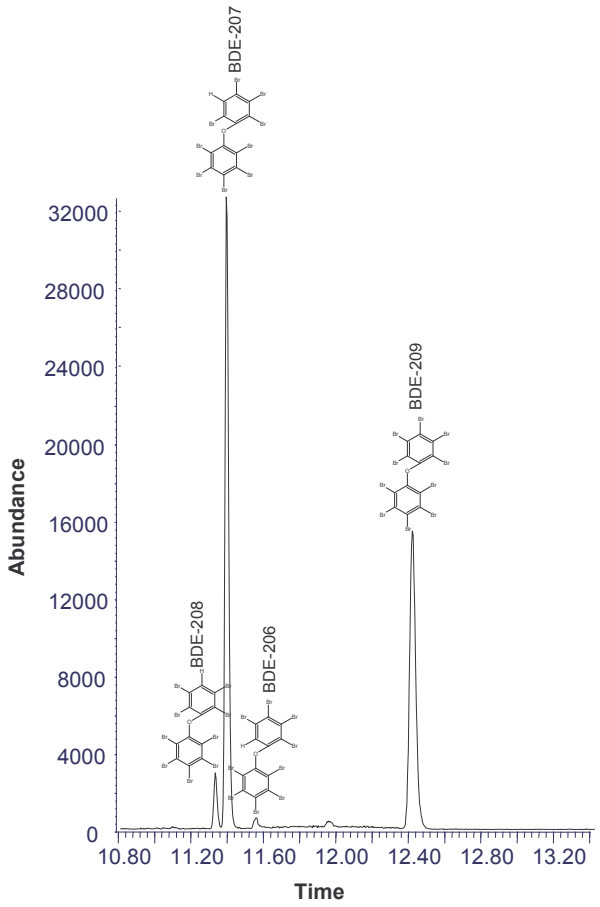

Figure 5. GC-ECNIMS chromatogramme of an egg sample. Nona-BDEs and BDE-209, in addition to ^{13}C-BDE-209 were monitored using negative chemical ionization (NCI) in selected ion monitoring (SIM) at *m/z* 484.5 and 486.5, and 494.5 and 498.5, respectively.

The PBDEs were quantified using a 12 points calibration curve of the quantification standards (pure standards of individual PBDE congeners analysed in the present study) and the internal standards. Detection limits of individual compounds were determined as three times the noise level (Table 4). Blank samples were analyzed continuously during the experiment (2 blank/20 samples) to monitor for laboratory contamination. In the present study the procedural blanks for BDE-206, BDE-207, BDE-208 and BDE-209 (N = 13 for nona-BDEs and N = 11 for BDE-209 (different N for the procedural blanks because nona and deca-BDEs were analyzed at different time points) were reported to

contain background contribution of these congeners. The concentrations of BDE-206, BDE-207, BDE-208 and BDE-209 in the procedural blanks were highly variable (Table 4). Because of the variable levels, procedural blanks could not be applied to correct for contamination in the samples. Corrections using the procedural blanks can only be applied if contamination in blank samples is relatively constant (Covaci et al., 2003). The concentrations of nona-BDEs and BDE-209 in the seabird eggs were only reported if the concentrations were above maximum concentrations reported in the procedural blanks. Values below the maximum blanks were set to not detect (nd).

Internal standards were added to all samples in the present study, including blank samples, control samples (seal blubber) (1 control/20 samples) and blind samples (hen egg) (1 blind/20 samples), to detect and correct changes in congener concentrations during the extraction, clean-up and detection, e.g. caused by degradation on the GC-MS. BDE-209 might be debrominated to other BDE congeners, e.g. nona-BDEs, on the GC-MS. This was not observed in the present study as no ^{13}C-nona-BDEs were detected, indicating that there is little degradation from deca-BDE to nona-BDEs through sample preparation and at the GC.

Internal standards and recovery standards (a known amount of all PBDE congeners analyzed for) were added to 2 hen eggs/20 samples to calculate relative recovery (Table 4).

The laboratory is accredited for the methods of routinely analyzed PBDEs congeners, i.e., BDE-28, BDE-47, BDE-100, BDE-99, BDE-154, BDE-153, BDE-183, and BDE-209 in biological samples according to the requirements of the NS-EN ISO/IEC 17025 (TEST 051). The laboratory's accredited analytical quality was approved in several international intercalibration tests, of which the most relevant and up to date were: FIRE 2003 (EU-project internal exercise on BFRs in food and egg of common tern) and QUASIMEME 2003 (Exercise 565, round 33 on BFRs). An accreditation of the method of analysing nona-BDEs in biological samples is currently in preparation. Nevertheless, normal quality-assurance/quality-control procedures were applied and approved.

6.3 DATA ANALYSES

Because only 27 % and 11 % of the nona-BDEs and BDE-209, respectively, were detected above the highest concentrations found in the procedural blank only exploratory statistics were conducted on the datasets. The relative concentrations of nona- and deca-BDEs were calculated as a proportion to ∑PBDEs (BDE-28, 47, 100, 99, 154, 153, 183, 206, 207, 208 and 209). For details of the concentrations of BDE-28, 47, 100, 99, 154, 153 and 183 see Knudsen et al. (2005).

Table 2. Product specifications.

Standard/instrument	Product specifications
Internal standard ^{13}C	Cambridge Isotope Laboratories, Inc., Andover, MA, USA
Ultrasonic homogenizer	4710 Series, Cole- Palmer Instruments, Chicago
Zymark® evaporation system	TurboWap II, Zymark Corporation, Hopkinton, MA, USA,
Standards	Cambridge Isotope Laboratories, Inc., Andover, MA, USA
PTV injector	Agilent Technologies
GC	Agilant 6890 series, Agilent Technologies, Avondale, PA, USA
MS	Agilent 5973 network Agilent Technologies, Avondale, PA, USA
DB-5-MS column	10 m × 0.25 mm I.D., 10 µm film thickness; J&W Scientific, Agilent Technologies

Table 3. Temperature programme

Temperature (°C)	Minutes
1) 80	2
2) 80 - 315	25°C min^{-1}
3) 315	10

Table 4. Quality assurance nona-BDEs and BDE-209.

Qualtity assurance parameter	Range
Recoveries*:	
Recovery BDE-206	104-111 %
Recovery BDE-207	124-144 %
Recovery BDE-208	97-109 %
Recovery BDE-209	86-111 %
Detection limits:**	
Detection limit BDE-206	8-13 pg/g wet weight (ww)
Detection limit BDE-207	8-13 pg/g ww
Detection limit BDE-208	20-33 pg/g ww
Detection limit BDE-209	30-125 pg/g ww
Procedural blanks:	
Procedural blank BDE-206 (N=13)	0-0.76 ng/ml
Procedural blank BDE-207 (N=13)	0-8.18 ng/ml
Procedural blank BDE-208 (N=13)	0-1.26 ng/ml
Procedural blank BDE-209 (N=11)	0.03-2.62 ng/ml

* The recoveries were within the acceptable range.

** Signal-to-noise ration of 1:3

7. RESULTS AND DISCUSSION

7.1 NONA-BDES AND BDE-209

The present study is to our knowledge the first study to report levels and temporal trends of nona-BDEs in arctic seabird eggs. The concentrations of nona-BDE and BDE-209 analyzed in individual eggs of herring gull, black-legged kittiwake, Atlantic puffin and glaucous gull are presented in Table 5.

An Atlantic puffin egg collected at Hornøya in 1993 exhibited the highest BDE-206 and BDE-207 concentrations (5.5 and 61.7 ng/g lipid weight (lw), respectively) reported in the present study (Table 5). In contrast, the maximum concentrations of BDE-208 and BDE-209 were reported in a glaucous gull and herring gull egg sampled in 2002/2003 at Bjørnøya and Hornøya, respectively (14.5 ng/g lw and 128 ng/g lw) (Table 5).

The highest levels of nona-BDEs and BDE-209 in the present study were higher than that previously reported in fish, eggs of little owl (*Athene noctua*) (BDE-209 only) and glaucous gulls (BDE-209 only) (Table 6). However, the BDE-209 levels in the present study were much lower than that previously reported in peregrine falcon eggs from Greenland and Sweden (Table 6).

There were large differences in concentrations of nona-BDEs and BDE-209 in the eggs. Of a total of 96 eggs 26 eggs were reported with nona-BDEs and 11 eggs with deca-BDE concentrations above the maximum concentrations detected in the procedural blanks (Table 5).

The highly variable levels of nona-BDEs and BDE-209 reported in the present study might be an indication of:

- Intra- and inter- specific divergences in biotransformation capacities (Walker, 1990; Sanderson et al., 1998)
- Individual-specific specialization on feeding and feeding rate (Bustnes et al., 2000; Hobson et al., 2002; Borgå et al., 2005)
- Species-specific differences in winter migration and exposure.
- Contamination at the laboratory (Thomsen et al., 2001).

As the sample size was very low in the present study it is difficult to ascertain unequivocal evidence of intra- and inter- specific differences in congener accumulation and retention. In all probability all of the above proposed sources might have an influence on the contaminant pattern.

Table 5. Nona-BDE and BDE-209 concentrations (ng/g lw) measured in individual eggs of herring gulls, Atlantic puffins and black-legged kittiwakes collected from Hornøya and Røst in 1983 (A), 1993 (B) and 2003 (C), and in eggs of glaucous gulls from Bjørnøya in 1997 and 2002 (D). The relative contribution of nona-BDEs and BDE-209 to ∑PBDE is presented. - = not detected (below maximum procedural blank).

A) Nona-BDE and BDE-209 concentrations (ng/g lw) measured in individual eggs of herring gulls, Atlantic puffins and black-legged kittiwakes collected from Hornøya and Røst in 1983.

Year	Location	Species	Lipid %	BDE-206	BDE-207	BDE-208	BDE-209	∑PBDEs	% BDE-206	% BDE-207	% BDE-208	% BDE-209
1983	H. gull	Hornøya	12.6	-	-	-	-	816	-	-	-	-
1983	H. gull	Hornøya	10.7	1.0	11.2	-	-	373	0.3	3.0	-	-
1983	H. gull	Hornøya	12.6	-	-	-	2.2	511	-	-	-	0.4
1983	H. gull	Hornøya	10.9	-	-	-	-	629	-	-	-	-
1983	H. gull	Hornøya	8.1	1.5	-	-	-	256	0.6	-	-	-
1983	H. gull	Hekkingen	9.0	-	-	-	-	161	-	-	-	-
1983	H. gull	Hekkingen	10.4	-	-	-	-	165	-	-	-	-
1983	H. gull	Hekkingen	9.7	-	-	-	-	161	-	-	-	-
1983	H. gull	Hekkingen	8.7	2.3	-	-	-	723	0.3	-	-	-
1983	H. gull	Hekkingen	9.5	-	-	-	-	839	-	-	-	-
1983	Puffin	Hornøya	11.8	-	-	-	-	118	-	-	-	-
1983	Puffin	Hornøya	14.2	1.7	-	-	-	136	1.2	-	-	-
1983	Puffin	Hornøya	12.4	3.4	-	-	-	163	2.1	-	-	-
1983	Puffin	Hornøya	13.8	-	-	-	-	136	-	-	-	-
1983	Puffin	Hornøya	10.8	-	-	-	-	116	-	-	-	-
1983	Puffin	Røst	13.3	-	-	-	-	49	-	-	-	-
1983	Puffin	Røst	12.2	-	-	-	-	66	-	-	-	-
1983	Puffin	Røst	12.8	-	-	-	-	112	-	-	-	-
1983	Puffin	Røst	10.8	-	-	-	-	82	-	-	-	-
1983	Puffin	Røst	11.9	-	-	-	-	74	-	-	-	-
1983	Kittiwake	Hornøya	9.2	-	-	-	-	111	-	-	-	-
1983	Kittiwake	Hornøya	10.2	2.8	-	-	-	81	3.5	-	-	-
1983	Kittiwake	Hornøya	9.3	-	-	-	-	85	-	-	-	-
1983	Kittiwake	Hornøya	10.5	1.4	-	-	-	104	1.3	-	-	-
1983	Kittiwake	Hornøya	9.7	1.6	15.5	-	-	166	1.0	9.3	-	-
1983	Kittiwake	Røst	6.7	-	-	-	19.2	189	-	-	-	10.1
1983	Kittiwake	Røst	8.2	-	-	-	-	263	-	-	-	-
1983	Kittiwake	Røst	7.8	-	-	-	-	136	-	-	-	-
1983	Kittiwake	Røst	6.2	-	-	-	-	269	-	-	-	-
1983	Kittiwake	Røst	6.4	-	-	-	-	209	-	-	-	-

B) Nona-BDE and BDE-209 concentrations (ng/g lw) measured in individual eggs of herring gulls, Atlantic puffins and black-legged kittiwakes collected from Hornøya and Røst in 1993.

Year	Location	Species	Lipid %	BDE-206	BDE-207	BDE-208	BDE-209	∑PBDEs	% BDE-206	% BDE-207	% BDE-208	% BDE-209
1993	H. gull	Hornøya	10.9	1.3	10.1	-	-	276	0.5	3.7	-	-
1993	H. gull	Hornøya	9.8	-	-	-	-	560	-	-	-	-
1993	H. gull	Hornøya	9.9	-	-	-	-	500	-	-	-	-
1993	H. gull	Hornøya	10.3	-	-	-	-	524	-	-	-	-
1993	H. gull	Hornøya	9.9	-	-	-	5.2	250	-	-	-	2.1
1993	H. gull	Røst	7.4	-	-	-	-	784	-	-	-	-
1993	H. gull	Røst	8.7	-	-	-	-	1590	-	-	-	-
1993	H. gull	Røst	10.1	-	-	-	-	1813	-	-	-	-
1993	H. gull	Røst	6.9	-	-	-	-	647	-	-	-	-
1993	H. gull	Røst	8.3	-	-	-	-	733	-	-	-	-
1993	Puffin	Hornøya	12.4	0.7	-	-	-	299	0.2	-	-	-
1993	Puffin	Hornøya	10.8	5.5	61.7	1.2	6.2	472	1.2	13.1	0.2	1.3
1993	Puffin	Hornøya	10.8	-	-	-	-	259	-	-	-	-
1993	Puffin	Hornøya	11.7	-	-	-	-	166	-	-	-	-
1993	Puffin	Hornøya	12.5	-	-	-	-	215	-	-	-	-
1993	Puffin	Røst	13.5	-	-	-	-	223	-	-	-	-
1993	Puffin	Røst	11.9	-	-	-	-	193	-	-	-	-
1993	Puffin	Røst	10.8	0.8	13.9	-	-	274	0.3	5.1	-	-
1993	Puffin	Røst	12.6	-	-	-	-	155	-	-	-	-
1993	Kittiwake	Hornøya	10.1	-	-	-	-	88	-	-	-	-
1993	Kittiwake	Hornøya	9.5	-	-	-	-	101	-	-	-	-
1993	Kittiwake	Hornøya	9.4	1.1	8.9	-	-	123	0.9	7.2	-	-
1993	Kittiwake	Hornøya	9.8	-	-	-	-	104	-	-	-	-
1993	Kittiwake	Hornøya	9.2	-	-	-	-	171	-	-	-	-
1993	Kittiwake	Røst	10.7	-	-	-	-	406	-	-	-	-
1993	Kittiwake	Røst	9.5	-	-	-	-	1185	-	-	-	-
1993	Kittiwake	Røst	10.3	-	-	-	-	351	-	-	-	-
1993	Kittiwake	Røst	8.3	-	-	-	-	280	-	-	-	-
1993	Kittiwake	Røst	8.3	-	-	-	-	353	-	-	-	-

C) Nona-BDE and BDE-209 concentrations (ng/g lw) measured in individual eggs of herring gulls, Atlantic puffins and black-legged kittiwakes collected from Hornøya and Røst in 2003.

Year	Location	Species	Lipid %	BDE-206	BDE-207	BDE-208	BDE-209	∑PBDEs	% BDE-206	% BDE-207	% BDE-208	% BDE-209
2003	H. gull	Hornøya	8.8	1.3	21.6	3.5	128	760	0.2	2.8	0.5	16.8
2003	H. gull	Hornøya	9.4	-	-	-	7.1	453	-	-	-	1.6
2003	H. gull	Hornøya	11.1	-	-	-	-	821	-	-	-	-
2003	H. gull	Hornøya	8.5	-	-	-	4.8	465	-	-	-	1.0
2003	H. gull	Hornøya	10.9	-	-	-	-	425	-	-	-	-
2003	H. gull	Røst	14.8	-	6.0	-	-	430	-	1.4	-	-
2003	H. gull	Røst	8.0	-	-	-	-	835	-	-	-	-
2003	H. gull	Røst	6.3	1.8	20.1	-	-	742	0.2	2.7	-	-
2003	H. gull	Røst	15.0	1.2	30.1	8.4	95.7	345	0.3	8.7	2.4	27.8
2003	H. gull	Røst	8.0	2.5	32.9	2.4	33.5	1072	0.2	3.1	0.2	3.1
2003	Kittiwake	Hornøya	11.1	-	-	-	-	144	-	-	-	-
2003	Kittiwake	Hornøya	8.9	-	-	-	-	271	-	-	-	-
2003	Kittiwake	Hornøya	9.0	-	-	-	-	115	-	-	-	-
2003	Kittiwake	Hornøya	9.8	1.9	18.4	-	-	197	1.0	9.4	-	-
2003	Kittiwake	Hornøya	8.7	-	-	-	-	84	-	-	-	-
2003	Kittiwake	Røst	10.5	-	-	-	-	307	-	-	-	-
2003	Kittiwake	Røst	10.1	-	-	-	-	266	-	-	-	-
2003	Kittiwake	Røst	7.3	1.3	11.5	-	-	321	0.4	3.6	-	-
2003	Kittiwake	Røst	11.0	1.0	11.0	-	-	178	0.5	6.2	-	-
2003	Kittiwake	Røst	10.6	-	-	-	-	298	-	-	-	-
2003	Puffin	Hornøya	10.9	0.9	-	-	-	83	1.1	-	-	-
2003	Puffin	Hornøya	12.6	-	-	-	-	74	-	-	-	-
2003	Puffin	Hornøya	14.5	-	-	-	-	145	-	-	-	-
2003	Puffin	Hornøya	15.1	-	-	-	-	107	-	-	-	-
2003	Puffin	Hornøya	15.3	0.6	-	-	-	89	0.7	-	-	-
2003	Puffin	Røst	12.0	-	-	-	-	110	-	-	-	-
2003	Puffin	Røst	17.1	-	-	-	-	144	-	-	-	-
2003	Puffin	Røst	11.6	-	-	-	-	61	-	-	-	-
2003	Puffin	Røst	11.2	-	-	-	-	76	-	-	-	-
2003	Puffin	Røst	16.4	-	-	-	-	56	-	-	-	-

D) Nona-BDE and BDE-209 concentrations (ng/g lw) measured in individual eggs of glaucous gulls from Bjørnøya in 1997 and 2002.

Year	Location	Species	Lipid %	BDE-206	BDE-207	BDE-208	BDE-209	∑PBDEs	% BDE-206	% BDE-207	% BDE-208	% BDE-209
2002	Glaucous gull	Bjørnøya	9.2	-	-	5.4	8.9	472	-	-	1.1	1.9
2002	Glaucous gull	Bjørnøya	8.0	1.0	25.0	14.5	32.2	862	0.1	2.9	1.7	3.7
2002	Glaucous gull	Bjørnøya	8.2	-	-	-	-	496	-	-	-	-
1997	Glaucous gull	Bjørnøya	8.9	-	-	-	-	495	-	-	-	-
1997	Glaucous gull	Bjørnøya	9	-	-	-	-	541	-	-	-	-
1997	Glaucous gull	Bjørnøya	9.1	0.9	8.7	-	-	531	0.2	1.6	-	-
1997	Glaucous gull	Bjørnøya	9.1	-	-	-	-	531	-	-	-	-

Table 6. Nona-BDE and BDE-209 concentrations (ng/g lw) in bird eggs reported in the scientific literature. Values are ranges or mean concentrations.

Species	Scientific name	BDE-206	BDE-207	BDE-208	BDE-209	Study area	Source
Peregrine falcon	*Falco peregrinus*	-	-	-	3.8-250	South Greenland	Vorkamp et al., 2005
					130	South Sweden	Lindberg et al., 2004
					110	North Sweden	Lindberg et al., 2004
Glaucous gull	*Larus hyperboreus*	-	-	-	23.2-52.5 (4 out of 32 analysed eggs)	Bjørnøya	Verreault et al., 2004
Osprey	*Pandion haliaetus*	-	-	-	nd	Norway	Herzke et al., 2005
White-tailed sea eagle	*Haliaeetus albicilla*	-	-	-	nd	Norway	Herzke et al., 2005
Golden eagle	*Aquila chrysaetos*	-	-	-	nd	Norway	Herzke et al., 2005
Peregrine falcon	*Falco peregrinus*	-	-	-	nd	Norway	Herzke et al., 2005
Merlin	*Falco columbarius*	-	-	-	nd	Norway	Herzke et al., 2005
Goshawk	*Accipiter gentiles*	-	-	-	nd	Norway	Herzke et al., 2005
Tawny owl	*Strix aluco*	-	-	-	nd-337	Norway	Bustnes JO., unpublished results
Little owl	*Athene noctua*	-	-	-	17 (1 out of 40 analyzed eggs)	Belgium	Jaspers et al., 2004
Roach	*Rutilus rutilus*	0.03-0.7 A	0.09-1.9 A	0.03-3.0 A	0.6-116	Baltic Sea	Burreau et al., 2004
Perch	*Perca fluviatilis*	0.002-1.1 A	0.004-2.0 A	0.003-1.0 A	0.3-31	Baltic Sea	Burreau et al., 2004
Pike	*Resox lucius*	0.002-0.4 A	0.01-0.5 A	0.004-0.2 A	0.5-4.6	Baltic Sea	Burreau et al., 2004
Peregrine falcon	*Falco peregrinus*	nd	nd-113 B	11.1-38.8 B	nd-307 B	USA	U.S. Department of Commerce, 2004

A: Nona 1, 2 and 3 (did not have standards for nona-BDEs). B: The analyte was positively identified; the associated numerical value is the approximate concentrations of analyte in the sample.

7.1.1 Spatial and temporal trends of nona-BDEs and BDE-209

No clear spatial differences of nona-BDEs and BDE-209 were apparent in the dataset (Table 5).

Few samples and low contaminant levels make it difficult to reveal trends. A lack of geographical differences of concentrations of nona-BDEs and BDE-209 in seabird colonies could indicate that the birds were exposed to similar local exposures. Alternatively, it might be an indication of similar feeding habits, migration patterns and exposures during the winter and early spring.

There appeared to be no clear temporal trends (1983-2003) in levels of nona-BDEs and BDE-209 (Table 5; Figure 6). As for the spatial trends, few samples and low contaminant levels make it difficult to reveal temporal trends.

Little is known about the congener-specific temporal trends of nona-BDEs in the environment. Temporal trends of BDE-209 have recently been assessed in peregrine falcon eggs from South Greenland where a statistically significant increase of 6 % per year was reported between 1986 and 2003 (Vorkamp et al., 2005).

7.1.2 Congener pattern of nona-BDEs and BDE-209

Figure 6A-C shows the percentage congener distribution of the nona-BDEs, BDE-209 and ∑PBDEs determined in the present study. All congeners in the present study (BDE-206, 207, 208 and 209) were reported in only five eggs (Table 5).

The relative contribution of the individual congeners was reported to be highly variable, and that of BDE-209 was highest compared to the other congeners in four of five eggs (Table 5). Furthermore, the levels of nona-BDEs decreased in the order BDE-207 > BDE-208 > BDE-206 (Table 7).

The congener pattern of nona-BDEs in the seabird eggs of the present study was slightly different than that reported in human serum (Table 7). In human serum BDE-208 was not detected (Table 7).

Table 7. Reported congener pattern of nona-BDEs in humans (Thuresson et al., 2005).

BDE congener	Thuresson et al., % of ∑nona-BDEs	Present study % of ∑nona-BDEs
BDE-206	25	2-8
BDE-207	75	62-90
BDE-208	0	2-36

Hornøya

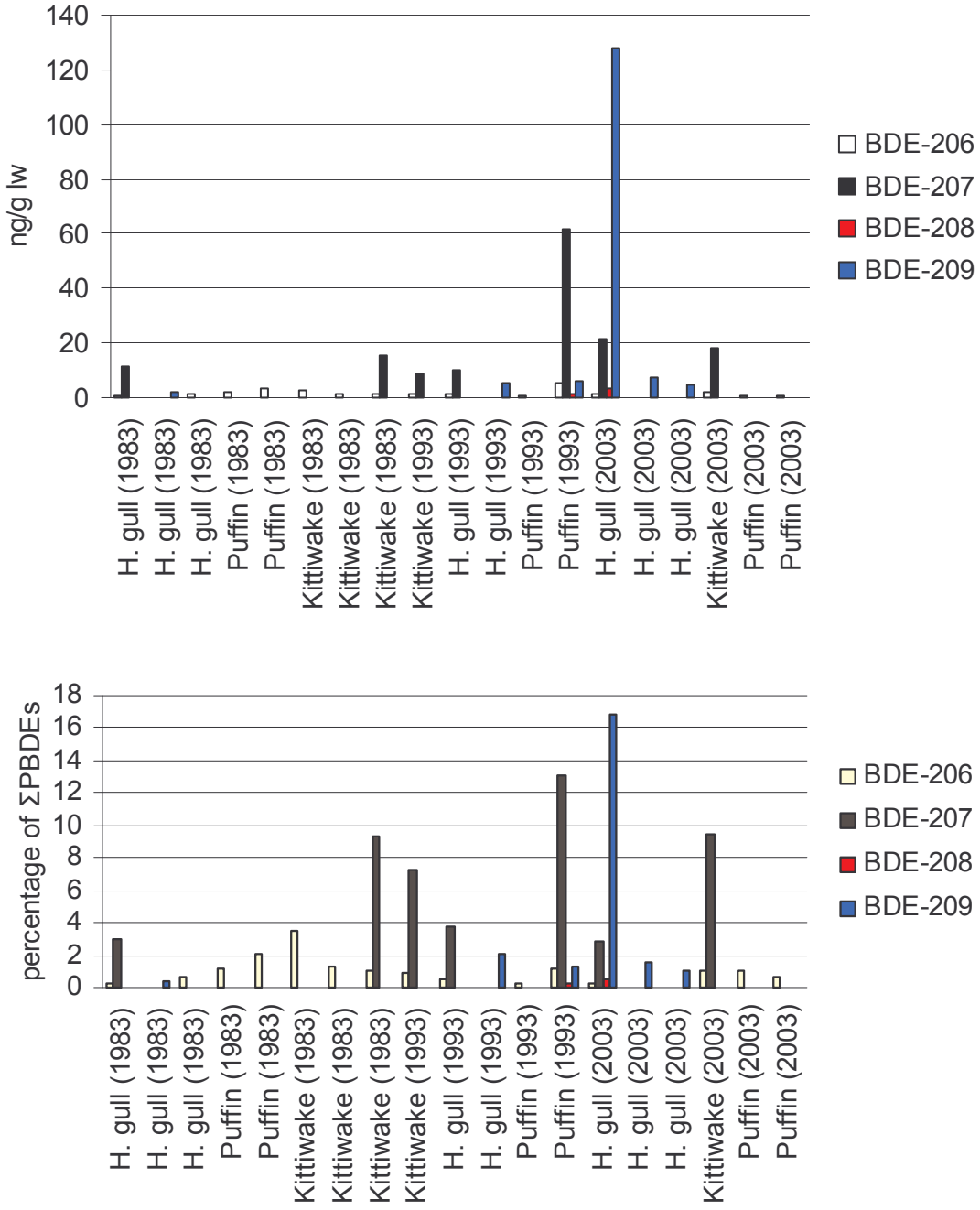

Figure 6A. Concentrations (ng/g lw; only samples above maximum procedural blank) and congener patterns of nona-BDEs and BDE-209 in individual seabird eggs sampled at Hornøya. The relative contribution of nona-BDEs and BDE-209 to ∑PBDEs is presented.

Røst

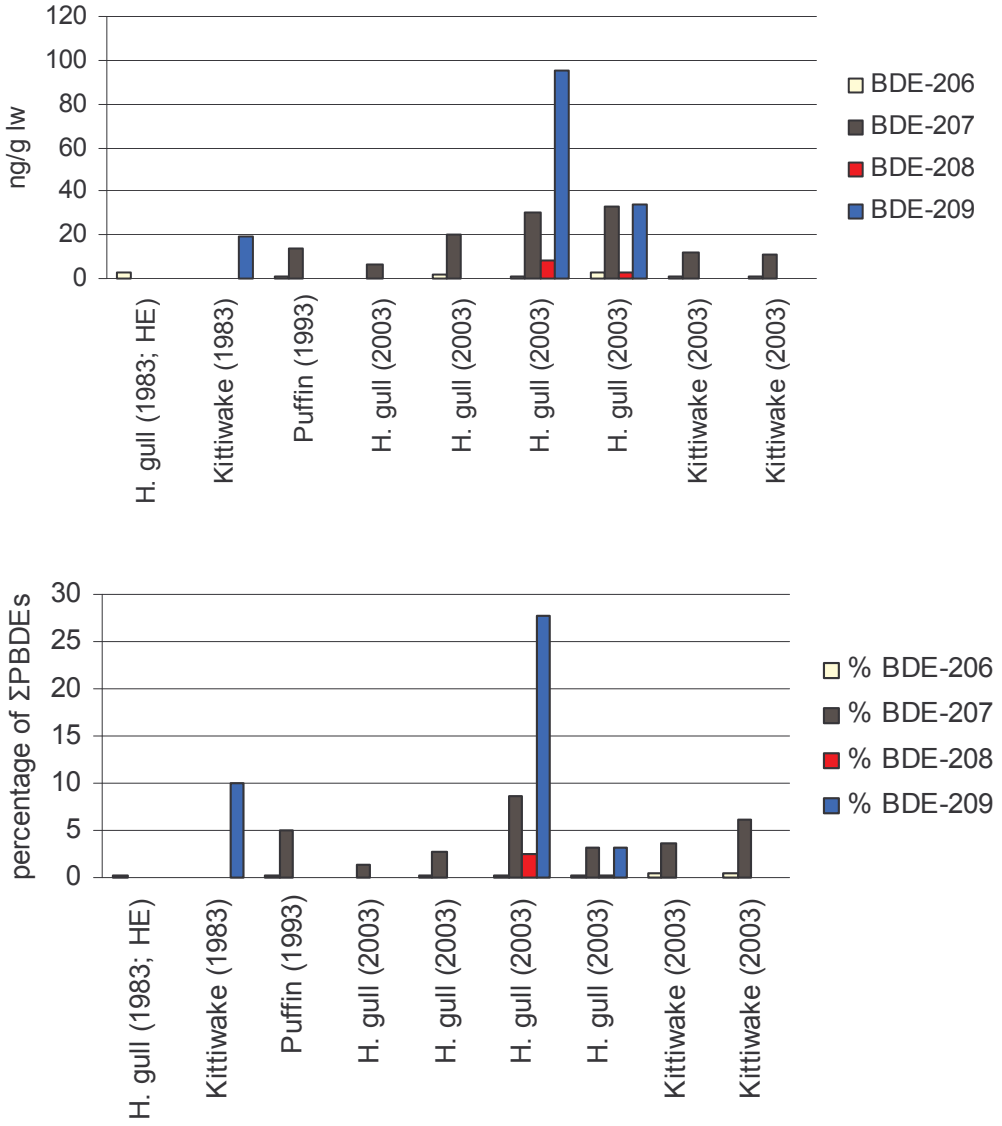

Figure 6B. Concentrations (ng/g lw; only samples above maximum procedural blank) and congener-patterns of nona-BDEs and BDE-209 in individual seabird eggs sampled at Røst. The relative contribution of nona-BDEs and BDE-209 to ∑PBDEs is presented.

Bjørnøya

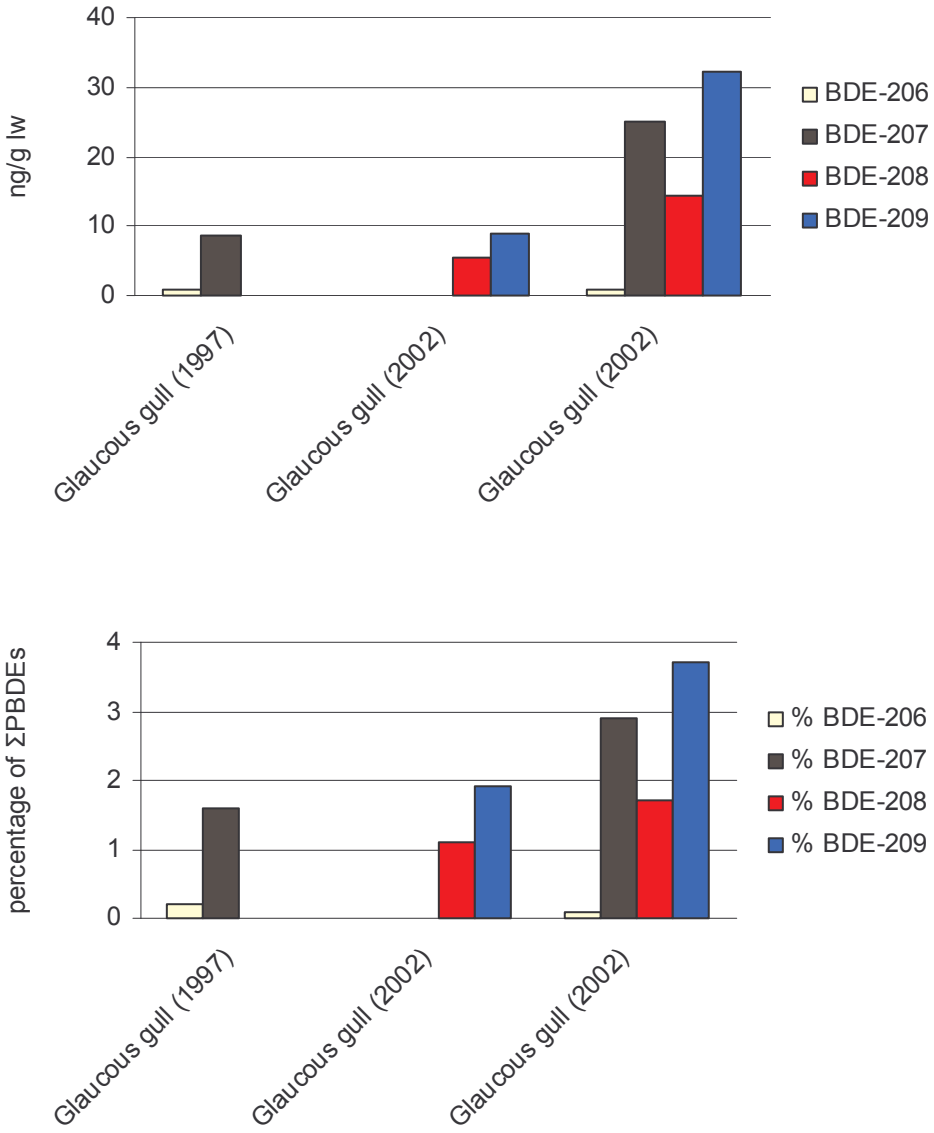

Figure 6C. Concentrations (ng/g lw; only samples above maximum procedural blank) and congener-patterns of nona-BDEs and BDE-209 in individual glaucous gull eggs sampled at Bjørnøya. The relative contribution of nona-BDEs and BDE-209 to ∑PBDEs is presented.

Interpreting the congener pattern in wildlife animals is very complex, involving several aspects such as:

- Source of exposure
- Diet and migration
- Physical-chemical properties of the congeners
- Species- and congener- specific degradation rates

Nona-BDEs are present in octa and decaBDE technical mixtures. A direct comparison of the congener pattern of the present study with that found in technical mixtures is difficult as seabirds are exposed to several different technical mixtures of PBDEs with slightly varying congener composition at their winter and summer feeding grounds. Furthermore, the patterns of the nona-BDEs and BDE-209 in the seabirds are reflecting the congener specific half lives ($t_{1/2}$). For example, $t_{1/2}$ of the higher brominated BDEs was shown to decrease with increasing bromination in humans (Thuresson et al., 2006) (Table 8). The congener pattern may also be a result of congener-specific formation rates following BDE-209 degradation and biotransformation, in the egg itself, and/or the parents and their prey. Different formation rates following BDE-209 degradation has previously been reported in an anaerobic BDE-209 experiment, with the highest formation rates reported for BDE-207 (Gereche et al., 2005). BDE-206 was not positively identified in the anaerobic experiment, which could indicate that either the formation rate of BDE-206 was very low, or that formation and further degradation resulted in a steady state (Gereche et al., 2005).

This study does not provide conclusive evidence about either contaminant patterns in seabirds, or the source of exposure, but clearly demonstrates the complexity of interpreting pattern of nona-BDEs and BDE-209 in arctic seabird eggs due to the highly variable concentrations between and among species.

Table 8. Calculated apparent half-lives ($T_{1/2}$=days) of BDEs in humans (Thuresson et al., 2006).

BDE congener	**$T_{1/2}$**	**95 % CI**
BDE-209	15	11-18
BDE-208	28	17-39
BDE-207	39	4-73
BDE-206	18	15-20
Octa-1	72	0-150
Octa-2	85	29-140
BDE-203	37	16-59
Octa-3	91	0-280
BDE-183	94	68-120

7.1.3 RISK ASSESSEMENT OF BDES

Knowledge about effects of PBDEs in wildlife is scarce. However, laboratory reports indicate endocrine disruption (e.g., Meerts et al., 2001), neurotoxic effects (e.g., Viberg et al., 2003) and induction of biotransformation enzymes (e.g., Chen et al., 2001). Tissue-specific toxicity thresholds of BDEs have not been established. Nevertheless, the levels of PBDEs are believed to be of minor importance compared to the levels of established organochlorines, such as PCBs.

Deca-BDE exposure has been shown to cause developmental and neurotoxic effects, e.g., induction of thyroid hyperplasia, heptocellular and thyroid adenomas, and carcinomas in mice (de Wit et al., 2004). Little is known about the biological effects of nona-BDE exposure. Nonetheless, the debromination of BDE-209 and nona-BDEs into lower brominated products may pose a great threat to animals as the lower brominated congeners have been reported to be more bioavailabile, bioaccumulative, persistent and toxic than the higher brominated compounds (Birnbaum and Staskal, 2004).

8. CONCLUSIONS

In the present study the Laboratory of Environmental Toxicology developed a method for detection of nona-BDEs in biota samples using PTV injector, short column and standards of BDE-206, BDE-207 and BDE-208.

The levels of nona-BDE and deca-BDE in seabird eggs were generally low and no species-specific differences were revealed. Large differences in concentrations of both nona-BDEs and BDE-209 were reported.

There were no clear temporal trends (1983-2003) or geographical differences (Hornøya/Røst) in levels of nona-BDEs and BDE-209 in the seabird eggs studied.

This study could not provide conclusive evidence about either contaminant patterns, or the origin of the contaminant patterns and levels in seabird eggs. However, it was evident that the levels of nona-BDEs and BDE-209 in some of the eggs were well above the highest levels reported in the procedural blanks, demonstrating the presence of nona-BDEs and BDE-209 in Arctic seabird eggs.

9. ACKNOWLEDGEMENTS

We would like to thank Hallvard Strøm (Norwegian Polar Institute), Tycho Anker-Nilssen and Geir Helge Systad (Norwegian Institute for Nature Research), and Statskog for their assistance with collection of the eggs. We would also like to thank Vidar Berg, Synne Authén Andresen and Katharina Bjarnar Løken (Norwegian School of Veterinary Science) for carrying out the analyses, and Kine Bæk for valuable comments on the manuscript (Norwegian School of Veterinary Science). This research was funded by the Norwegian College of Veterinary Science and the Norwegian Pollution Control Authority.

10. REFERENCES

Anker-Nilssen T, Bakken V, Strøm H, Golovkin AN, Bianki VV, Tatarinkova IP. 2000. The status of marine birds breeding in the Barents Sea region. Rapport nr. 113, Norsk Polarinstitutt, Norway, pp. 94-96

Barrett RT, Skaare JU, Norheim G, Vader W, Frøslie A. 1985. Persistent organochlorines and mercury in eggs of Norwegian seabirds 1983. Environmental Pollution (Ser. A) 39: 79-93

Barrett RT, Skaare JU, Gabrielsen GW. 1996. Recent changes in levels of persistent organochlorines and mercury in eggs of seabirds from the Barents Sea. Environmental Pollution 92: 13-18

Birnbaum LS, Staskal DF. 2004. Brominated flame retardants: Cause for concern? Environmental Health Perspectives 112: 9-17

Borgå K, Wolkers H, Skaare JU, Hop H, Muir DCG, Gabrielsen GW. 2005. Bioaccumulation of PCBs in Arctic seabirds: influence of dietary exposure and congener biotransformation. Environmental Pollution 134: 397-409

Braune BM, Donaldson GM, Hobson KA. 2001. Contaminant residues in seabird eggs from the Canadian Arctic. Part I. Temporal trends 1975-1998. Environmental Pollution 114: 39-54

BSEF. 2006. http//www.bsef-site.com/bromine/our_industry/index.php

Burreau S, Zebuhr Y, Broman D, Ishaq R. 2004. Biomagnification of polychlorinated biphenyls (PCBs) and polybrominated diphenyl ethers (PBDEs) studied in pike (Esox lucius), perch (Perca fluviatilis) and roach (Rutilus rutilus) from the Baltic Sea. Chemosphere 55: 1043-1052

Bustnes JO, Erikstad KE, Bakken V, Mehlum F, Skaare JU. 2000. Feeding ecology and the concentration of organochlorines in glaucous gulls. Ecotoxicology 9: 179-186

Chen G, Konstantinov AD, Chittim BG, Joyce EM, Bols NC, Bunce NJ. 2001. Synthesis of polybrominated diphenyl ethers and their capacity to induce CYP1A by the Ah receptor mediated pathway. Environmental Science & Technology 35: 3749-3756

Covaci A, Voorspoels S, de Boer J. 2003. Determination of brominated flame retardants, with emphasis on polybrominated diphenyl ethers (PBDEs) in environmental and human samples - a review. Environ. Int. 29: 735-56

Cox P, Efthymiou P. 2003. Directive 2003/11/EC of the European parliament and of the

council of February 6, 2003 amending for the 24th time Council Directive 76/669/EEC relating to restrictions on the marketing and use of certain dangerous substances and preparations (pentabromodiphenyl ether, octabromodiphenyl ether). Official Journal of European Union OJ L 42: 45-46

de Wit C. 2002. An overview of brominated flame retardants in the environment. Chemosphere 46: 583-624

de Wit C, Fisk, A, Hobbs, K, Muir, D, Gabrielsen, G, Kallenborn, R, Krahn, M, Norstrom, R, Skaare, J. 2004. AMAP Assessment 2002. Persistent Organic Pollutants in the Arctic. Arctic Monitoring and Assessment Program report. Oslo, Norway, 310 pp

EU, 2005. ADDENDUM to the May 2004 Environmental Risk Assessment Report for DECABROMODIPHENYL ETHER. CAS Number: 1163-19-5

Fimreite N, Brun E, Frøslie A, Frederichsen P, Gundersen N. 1974. Mercury in Eggs of Norwegian Seabirds. Astarte 7: 71-76

Gerecke AC, Hartmann PC, Heeb NV, Kohler H-PE, Giger W, Schmid P, Zennegg M, Kohler. M. 2005. Anaerobic degradation of decabromodiphenyl ether. Environmental Science & Technology 39: 1078-1083

Great Lakes Chemical Corporation 2005. http://www.e1.greatlakes.com/corp/news/jsp/current_news_detail.jsp?contentfile=01182005_FR_phase_out.htm

Hardy ML. 2002. A comparison of the properties of the major commercial PBDPO/PBDE product to those of major PBB and PCB products. Chemosphere 46: 757-777

Herzke D, Berger U, Kallenborg R, Nygård T, Vetter W. 2005. Brominated flame retardants and other organobromines in Norwegian predatory bird eggs. Chemosphere 61: 441-449

Hobson KA, Fisk A, Karnovsky N, Holst M, Gagnon JM, Fortier M. 2002. A stable isotope (δ13C, δ15N) model for the North Water food web: implications for evaluating trophodynamics and the flow of energy and contaminants. Deep-Sea Research 49: 5131-5150

Jakobsson K, Thuresson K, Rylander L, Sjodin A, Hagmar L, Bergman Å. 2002. Exposure to polybrominated diphenyl ethers and tetrabromobisphenol A among computer technicians. Chemosphere 46: 709-716

Jaspers V, Covaci A, Maervoet J, Dauwe T, Schepens P, Eens M. 2004. Brominated flame retardants in Belgian little owl (*Athene noctua*) eggs. Organohalogen Compounds 66: 3856-3860

Knudsen LB, Gabrielsen GW, Verreult J, Barrett R, Polder A, Skaare JU. 2005. Levels and temporal trends (1983-2003) of brominated flame retardants in eggs of herring gulls (*Larus argentatus*), Atlantic puffins (*Fratercula arctica*) and black-legged kittiwakes (*Rissa tridactyla*) from Northern Norway. Report, The Norwegian Pollution Control Authority (SFT). ISBN: 82-7655-497-0

Kierkegaard A, Balk L, Tjarnlund U, de Wit CA, Jansson B. 1999. Dietary uptake and biological effects of decabromodiphenyl ether in rainbow trout (*Oncorhynchus mykiss*). Environmental Science & Technology 33: 1612-1617

Lindberg P, Sellström U, Häggberg L, de Wit CA. 2004. Higher brominated diphenyl ethers and hexabromocyclododecane found in eggs of peregrine falcons (*Falco peregrinus*) breeding in Sweeden. Environmental Science & Technology 38: 93-96

Meerts IATM, Letcher RJ, Hoving S, Marsh G, Bergman A, Lemmen JG, van der Burg B, Brouwer A. 2001. In vitro estrogenicity of polybrominated diphenyl ethers, hydroxylated PBDEs, and polybrominated bisphenol A compounds. Environmental Health Perspectives 109: 399-407

Murvoll KM, Jenssen BM, Skaare JU. 2005. Effects of Pentabrominated Diphenyl Ether (Pbde-99) on Vitamin Status. Journal of Toxicology and Environmental Health Part A 68: 515-533

Mörck A, Hakk H, Orn U, Klasson-Wehler E. 2003. Decabromodiphenyl ether in the rat: absorption, distribution, metabolism, and excretion. Drug Metab Dispos. 31: 900-907

Sanderson JT, Kennedy SW, Giesy JP. 1998. In vitro induction of ethoxyresorufin-O-deethylase and porphyrins by halogenated aromatic hydrocarbons in avian primary hepatocytes. Environmental Toxicology and Chemistry 17: 2006-2018

Sørmo EG, Salmer MP, Jenssen BM, Hop H, Bæk K, Kovacs KM, Lydersen C, Falk-Petersen S, Gabrielsen GW, Lie E, Skaare JU. Biomagnification of polybrominated diphenyl ether and hexabromocyclododecane flame retardants in the polar bear food chain in Svalabard, Norway. Environmental Toxicology and Chemistry (in press)

Thuresson K, Bergman A, Jakobsson K. 2005. Occupational Exposure to Commercial decabromodiphenyl ether in workers manufacturing or handling flame-retarded rubber. Environmental Science & Technology 39: 1980-1986

Thuresson K, Höglund P, Hagmar L, Sjödin A, Bergman Å, Jakobsson K. 2006. Apparent half-lives of hepta- to decabrominated diphenyl ethers in human serum

as determined in occupationally exposed workers. Environmental Health Perspectives 114: 176-181

Thomsen C, Leknes H, Lundanes E, Becher G. 2001. Brominated flame retardants in laboratory air. Journal of Chromatography A 923: 299-304

U.S. Department of Commence. 2004. Work summary and data report for the collection of eggs from American peregrine falcon, Hudson river, New York. Hudson river natural resource damage assessment. Hudson river natural resource trustees. State of New York. pp. 1-52

Verreault J, Gabrielsen GW, Letcher RJ, Muir DCG, Chu S. 2004. New and established organohalogen contaminants and their metabolites in plasma and eggs of glaucous gull from Bear Island. SFT rapport prosjektnummer 6003089, The Norwegian Pollution Control Authority, Oslo, Norway. pp. 2-19

Verreault J, Gabrielsen GW, Chu S, Muir DCG, Andersen M, Hamaed A, Letcher RJ. 2005. Flame retardants and methoxylated and hydroxylated polybrominated diphenyl ethers in two Norwegian Arctic Top predators: Glaucous gulls and Polar Bears. Environmental Science & Technology 39: 6021-6028

Viberg H, Fredriksson A, Jakobsson E, Örn U, Eriksson P. 2003. Neurobehavioral derangements in adult mice receiving decabrominated diphenyl ether (PBDE 209) during a defined period of neonatal brain development. Toxicological Sciences 76: 112-120

Vorkamp K, Thomson M, Falk K, Leslie H, Møller S, Sørensen PB. 2005. Temporal development of brominated flame retardants in peregrine falcon (*Falco peregrinus*) eggs from South Greenland (1986-2003). Environmental Science & Technology 39: 8199-8206

Walker CH. 1990. Persistent pollutants in fish-eating sea birds – bioaccumulation, metabolism and effects. Aquatic Toxicology 17: 293-324

Watanabe I, Sakai S. 2003. Environmental release and behavior of brominated flame retardants. Environment International 29: 665-682

Statens forurensningstilsyn (SFT)
Postboks 8100 Dep, 0032 Oslo
Besøksadresse: Strømsveien 96

Telefon: 22 57 34 00
Telefaks: 22 67 67 06
E-post: postmottak@sft.no
Internett: www.sft.no

Utførende institusjon Norges veterinærhøgskole	Kontaktperson SFT Jon L. Fuglestad	ISBN-nummer 82-7655-289-7

	Avdeling i SFT Seksjon for miljødata	TA-nummer 2175/2006

Oppdragstakers prosjektansvarlig Janneche Utne Skåre	År 2006	Sidetall 29	SFTs kontraktnummer 952/2006

Utgiver Norges Veterinærhøgskole	Prosjektet er finansiert av SFT

Forfatter(e)
Lisa Bjørnsdatter Knudsen, Anuschka Polder, Siri Føreid, Elisabeth Lie, Geir Wing Gabrielsen, Rob Barrett, Janneche Utne Skåre

Tittel - norsk og engelsk
Nona- og deka-bromerte difenyletere i sjøfuglegg fra Nord-Norge og Svalbard
Nona- and deca-brominated diphenylethers in seabird eggs from Northern Norway and Svalbard

Sammendrag – summary
The aim of the present study was to determine levels of nona-brominated diphenylethers (BDEs) and BDE-209 in seabird eggs. Eggs were collected from herring gulls (*Larus argentatus*), Atlantic puffins (*Fratercula arctica*) and black-legged kittiwakes (*Rissa tridactyla*) in 1983, 1993 and 2003 at Røst and Hornøya (Northern Norway). Furthermore, eggs of glaucous gulls (*Larus hyperboreus*) were collected in 1997 and 2002 at Bjørnøya (Svalbard).

4 emneord Sjøfuglegg, nona-BDEer, BDE-209, kongener mønster	4 subject words Seabird eggs, nona-BDEer, BDE-209, congener pattern