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Environmental Contaminants in an Urban Fjord, 2017

CORRIGENDUM

Changes for electronic and printed version of the report “Environmental Contaminants in an Urban Fjord, 2017”, NIVA serial number 7299-2018, published 2018.

Page 4: «Dechlorane plus, et flammehemmende middel i plast og polymerer, ble inkludert i Urban fjord-programmet i 2017 og ble funnet i nevneverdige konsentrasjoner i partikkelfaser (partikkelfraksjon i overvann, kloakkslam og sediment)» changed to «Dechlorane plus, et flammehemmende middel i plast og polymerer, ble inkludert i Urban fjord-programmet i 2017 og ble detektert i partikkelfaser (partikkelfraksjon i overvann, kloakkslam og sediment)».

Page 6: «Dechlorane plus, a flame retardant in plastics and polymers, was included in the Urban fjord programme in 2017 and was found in notable concentrations in particulate phases, i.e. the particulate fraction in storm water, sewage sludge and sediment.» changed to «Dechlorane plus, a flame retardant in plastics and polymers, was included in the Urban fjord programme in 2017 and was detected in particulate phases, i.e. the particulate fraction in storm water, sewage sludge and sediment».

Page 39: «Dechlorane plus was found in notable concentrations in the sediment sample (sum of syn and anti isomers 1632 ng/g dry wt.; Figure 5). In addition, dechlorane 603 was detected in a concentration of 69 ng/g dry wt. (see electronic Appendix)» changed to «Dechlorane plus was found in the sediment sample (sum of syn and anti isomers 1.632 ng/g dry wt.; Figure 5). In addition, dechlorane 603 was detected in a concentration of 0.069 ng/g dry wt. (see electronic Appendix)».

Figure 5 (page 39): The numbers in «the table» under the figure are changed from 383 and 1249 to 0.383 and 1.249, respectively.

Table 11 (page 54): The following section of the table

Dechlorane	Mean	Min.	Max.	Detected in no. of samples
Dechlorane 602	679.7	124.6	3357.5	15
Dechlorane 603	162.1	<50	690.1	13
Dechlorane 604	n.d.	<20	<50	0
Dechlorane plus syn	47.9	<250	295.7	4
Dechlorane plus anti	108.8	<500	669.9	4

changed to:

Dechlorane	Mean	Min.	Max.	Detected in no. of samples
Dechlorane 602	0.680	0.125	3.358	15
Dechlorane 603	0.162	<0.050	0.690	13
Dechlorane 604	n.d.	<0.020	<0.050	0
Dechlorane plus syn	0.048	<0.250	0.296	4
Dechlorane plus anti	0.109	<0.500	0.670	4

Page 58: «Some dechlorane compounds (note dechlorane 602 and 603) were detected in cod liver in concentrations of several hundred ng/g (Table 11)» changed to «Some dechlorane compounds (note dechlorane 602 and 603) were detected in cod liver (Table 11)».

Page 58: The following sentence is removed from the second last paragraph: «As described in Chapter 2.2.1, dechlorane plus is used as a flame retardant in plastics and polymers, such as nylon, polyurethane, polypropylene, neoprene and silicone rubber (marketed as an alternative to deca-BDE).».

Table 12 (page 59): the following section of the table

Dechlorane	Blood Mean	Blood Min.	Blood Max.	Egg Mean	Egg Min.	Egg Max.	Det. no.
Dechlorane 602	2.70	<20	40.46	n.d.	<5	<5	14/0
Dechlorane 603	n.d.	<10	<29	14.61	<5	41.13	0/12
Dechlorane 604	n.d.	<50	<50	n.d.	<10	<10	0/0
Dechlorane plus syn	9.38	<125	140.67	107.90	<25	654.13	1/14
Dechlorane plus anti	n.d.	<250	<286	336.91	<50	1942.45	0/14

changed to:

Dechlorane	Blood Mean	Blood Min.	Blood Max.	Egg Mean	Egg Min.	Egg Max.	Det. no.
Dechlorane 602	0.003	<0.020	0.041	n.d.	<0.005	<0.005	14/0
Dechlorane 603	n.d.	<0.010	<0.029	0.015	<0.005	0.041	0/12
Dechlorane 604	n.d.	<0.050	<0.050	n.d.	<0.010	<0.010	0/0
Dechlorane plus syn	0.009	<0.125	0.141	0.108	<0.025	0.654	1/14
Dechlorane plus anti	n.d.	<0.250	<0.286	0.337	<0.050	1.943	0/14

Page 63: «Dechlorane plus was found in eggs of herring gull in concentrations of several hundred ng/g, and the variability was high (Table 12)» changed to «Dechlorane plus was found in eggs of herring gull and the variability was high (Table 12)».

Table 13 (page 70): The following section of the table

Dechlorane	Blood Mean	Blood Min.	Blood Max.	Egg Mean	Egg Min.	Egg Max.	Det. no.
Dechlorane 602	10.75	<20	161.25	n.d.	<5	<5	1/0
Dechlorane 603	n.d.	<20	<20	27.12	<5	209.31	0/12
Dechlorane 604	n.d.	<80	<80	n.d.	<20	<20	0/0
Dechlorane plus syn	16.77	<100	151.59	183.02	<33	1245.76	2/14
Dechlorane plus anti	31.57	<200	252.36	618.18	<67	3619.01	2/14

changed to:

Dechlorane	Blood Mean	Blood Min.	Blood Max.	Egg Mean	Egg Min.	Egg Max.	Det. no.
Dechlorane 602	0.011	<0.020	0.161	n.d.	<0.005	<0.005	1/0
Dechlorane 603	n.d.	<0.020	<0.020	0.027	<0.005	0.209	0/12
Dechlorane 604	n.d.	<0.080	<0.080	n.d.	<0.020	<0.020	0/0
Dechlorane plus syn	0.017	<0.100	0.152	0.183	<0.033	1.246	2/14
Dechlorane plus anti	0.032	<0.200	0.252	0.618	<0.067	3.619	2/14

Page 73: «As in eggs of herring gulls from the Inner Oslofjord, Dechlorane plus was found in eggs of herring gull from the outer Oslofjord in concentrations of several hundred ng/g, and the variability was even higher than in the inner fjord (Table 13; Figure 24)» changed to « As in eggs of herring gulls from the Inner Oslofjord, Dechlorane plus was found in eggs of herring gull from the outer Oslofjord, and the variability was even higher than in the inner fjord (Table 13; Figure 24)».

Figure 24 (page 74): The scales on the concentration axes have been changed.

Table 14 (pages 74-75): The following section of the table

Dechlorane	Blood Mean	Blood Min.	Blood Max.	Egg Mean	Egg Min.	Egg Max.	Det. no.
Dechlorane 602	3.83	<20	30.00	121.34	53.73	297.01	2/15
Dechlorane 603	n.d.	<20	<20	27.18	<20	271.60	0/5
Dechlorane 604	n.d.	<100	<100	n.d.	<100	<100	0/0
Dechlorane plus syn	7.18	<100	107.66	128.43	<100	224.22	1/13
Dechlorane plus anti	49.66	<200	262.72	245.43	<200	519.49	3/12

changed to:

Dechlorane	Blood Mean	Blood Min.	Blood Max.	Egg Mean	Egg Min.	Egg Max.	Det. no.
Dechlorane 602	0.004	<0.020	0.030	0.121	0.054	0.297	2/15
Dechlorane 603	n.d.	<0.020	<0.020	0.027	<0.020	0.272	0/5
Dechlorane 604	n.d.	<0.100	<0.100	n.d.	<0.100	<0.100	0/0
Dechlorane plus syn	0.007	<0.100	0.108	0.128	<0.100	0.224	1/13
Dechlorane plus anti	0.050	<0.200	0.263	0.245	<0.200	0.520	3/12

Page 77: «As in eggs of herring gulls, both form the Inner and Outer Oslofjord, dechlorane plus was found in eggs of Eider ducks from the Inner Oslofjord in concentrations of several hundred ng/g (Table 14). Compared to herring gull, the variability was low. Furthermore, Dechlorane 602 was detected in notable concentrations in all eider duck eggs (Table 14)» changed to «As in eggs of herring gulls, both from the Inner and Outer Oslofjord, dechlorane plus was found in eggs of Eider ducks from the Inner Oslofjord (Table 14). Compared to herring gull, the variability was low. Furthermore, Dechlorane 602 was detected in all eider duck eggs (Table 14)».

Page 78: «Dechlorane plus was found in concentrations of several µg/L, however only in the particulate fraction (Figure 25)» changed to «Dechlorane plus was found in concentrations of several ng/L, however only in the particulate fraction (Figure 25)».

Figure 25 (page 78): The numbers in the «table» under the figure are changed from 4377 and 11917 to 4.377 and 11.917, respectively.

Page 84: «Dechlorane plus were found in high concentrations in the sludge (mean concentration, sum of syn and anti isomers, n=2, 9544 ng/g dry wt.; Figure 29)» changed to «Dechlorane plus was found in the sludge (mean concentration, sum of syn and anti isomers, n=2, 9.5 ng/g dry wt.; Figure 29)».

Figure 29 (page 84): The numbers in the «table» under the figure are changed from 2140 and 7404 to 2.140 and 7.404, respectively.

Page 90: «Chlorinated paraffins apparently constitute major proportions in all species/matrices, especially in sludge from the sewage treatment plant, as well as in mussels (Figure 35)» changed to «Chlorinated paraffins apparently constitute major proportions in all species/matrices, especially in the particulate fraction of stormwater and sludge from the sewage treatment plant, as well as in mussels (Figure 35)».

Page 90: The following sentence is removed from the last paragraph: «A conspicuous result was that dechlorane plus constitute major proportions in particulate matrices (particulate phase of storm water, STP sludge and sediment from the Inner Oslofjord; Figure 35)».

Pages 90-91: «PCBs and PBDEs do not constitute major proportions of the sum of contaminants, except for PCBs in the lipid rich tissues herring muscle and cod liver (PCBs were not analysed in samples from the STP; Figure 35)» changed to «PCBs and PBDEs do not constitute very high (<5 %) proportions of the sum of contaminants, except for PCBs in the lipid rich tissues herring muscle and cod liver (PCBs were not analysed in samples from the STP; Figure 35). PBDEs constituted ~6% of the sum of the selected contaminants in sludge from the sewage treatment plant (Figure 35)».

Page 91: «Phenolic compounds constituted major proportions of the sum of contaminants in storm water (particularly the dissolved fraction), and to some degree in the samples from the STP (effluent water and sludge)» changed to «Phenolic compounds constituted major proportions of the sum of contaminants in storm water (the dissolved fraction), and to some degree in sludge from the STP (Figure 35)».

Figure 35 (pages 92-93): The figures (A. and B.) have been replaced. Corresponding changes are made in the figure legend.

Page 109: «Dechlorane plus was also included in the Urban fjord programme in 2017 and was found in notable concentrations in particulate phases (particulate fraction in storm water, sewage sludge and sediment) » changed to «Dechlorane plus was also included in the Urban fjord programme in 2017 and was detected in particulate phases (particulate fraction in storm water, sewage sludge and sediment)».

Page 110: «For instance, dechlorane plus apparently constitute a major proportion of the contaminants particulate phases, such as the particulate fraction of storm water, sediments and sewage sludge» changed to «For instance, chlorinated paraffins apparently constitute major proportions in all species/matrices examined».

The electronic appendix to the report has been changed so that the concentrations of following parameters are given as pg/g, or pg/L: Dibromaldrin, Dechlorane 602, Dechlorane 603, Dechlorane 604, Dechlorane plus syn, Dechlorane 601 and Dechlorane plus anti.

Oslo, March 2019

Anders Ruus

COLOPHON

Executive institution

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Miljøgifter i en urban fjord, 2017 ISBN: 978-82-577-7103-4
Environmental Contaminants in an Urban Fjord, 2017 Project number: 17146

Summary

This programme, “Environmental Contaminants in an Urban Fjord” has covered sampling and analyses of sediment and organisms in a marine food web of the Inner Oslofjord, in addition to samples of blood and eggs from herring gull and eider duck. The programme also included inputs of pollutants via surface water (storm water), and effluent water and sludge from a sewage treatment plant. The bioaccumulation potential of the contaminants in the Oslo fjord food web was evaluated. The exposure to/accumulation of the contaminants was also assessed in birds. A vast number of chemical parameters have been quantified, in addition to some biological effect parameters in cod, and the report serves as valuable documentation of the concentrations of these chemicals in different compartments of the Inner Oslofjord marine ecosystem.

4 emneord

Miljøgifter, urban, næringskjede, bioakkumulering

4 subject words

Contaminants, Urban areas, Food web, Bioaccumulation

Front page photo

Anders Ruus

Foreword

The programme covers sampling and analyses of organisms in a marine food web of the Inner Oslofjord in 2017 in addition to samples of blood and eggs of herring gull. Furthermore, optional samples of blood and eggs of herring gull from the Outer Oslofjord were analysed in 2017, as well as samples of blood and eggs of eider duck. The programme also includes inputs of pollutants via surface water (storm water), and sewage treatment plant discharges. This monitoring programme adds to results from other monitoring programmes such as "Contaminants in coastal areas" (MILKYS) and "Riverine inputs and direct discharges to Norwegian coastal waters" (RID). These programmes are referred to, when relevant. 2017 represents the fifth year of the Urban Fjord programme. Some changes/improvements were made in the design from 2014 to 2015 and from 2016 to 2017. In 2017, two MSc-student from the University of Oslo were affiliated with the programme to look in more detail at bird related issues.

The study was carried out by NIVA, with a majority of the chemical analyses performed by the Norwegian Institute for Air Research, NILU. Collection of herring gulls and eider duck was conducted by the University of Oslo (Morten Helberg, Centre for Ecological and Evolutionary Synthesis).

Besides the authors of this report, several persons are acknowledged for their contribution in sample collection, sample preparation and analysis: Thomas Rundberget, Ingar Johansen, Gunhild Borgersen, Alfhild Kringstad, Camilla With Fagerli, Tânia Gomes, Marthe Torunn Solhaug Jenssen, Pawel Rostowski, Mikael Harju, Hilde Uggerud, Marit Vadset, Inger-Christin Steen, Carsten Lome.

Oslo, oktober 2018

Anders Ruus
Forsker I, Marin Forurensning

Sammendrag

Dette programmet, "Miljøgifter i en Urban Fjord" har omfattet prøvetaking og analyse av sediment og organismer i en marin næringskjede i Indre Oslofjord i 2017, i tillegg til prøver av blod og egg fra gråmåke. Videre ble blod og egg fra gråmåke i Ytre Oslofjord og fra ærfugl i indre Oslofjord analysert som opsjon i 2017. Programmet omfattet også undersøkelser av tilførsler av miljøgifter via overvann, samt via kloakkrenseanlegg.

Målet med programmet var å undersøke tilførsler av miljøgifter som er tilstede i et tett befolket område og studere hvordan disse påvirker et fjordsystem. Denne undersøkelsen er ett skritt mot Miljødirektoratets generelle mål om å:

- Anslå graden av bioakkumulering av utvalgte miljøgifter på flere trofiske nivåer i marine næringskjeder.
- Koble eksponeringen av miljøgifter på marine organismer til toksiske effekter på ulike biologiske nivåer, inkludert hormonforstyrrende effekter og interaksjonseffekter ("cocktaileffekter").
- Identifisere kilder og sluk for miljøgifter i fjordsystemer ("skjebnen" til miljøgifter i en fjord), og utforme målrettede tiltak.

Intensjonen er videre at data skal brukes i internasjonale miljøgiftreguleringer, som REACH og Stockholmkonvensjonen. Dessuten skal programmet frembringe data som vil være til hjelp i å gjennomføre kravene i Vanndirektivet ("Vannforskriften") i forbindelse med statlig basisovervåking. 2017 er det femte året "Miljøgifter i en Urban Fjord" har vært gjennomført. Det er gjort noen forandringer/forbedringer i design/innhold av programmet fra starten i 2013, frem til 2017.

Bioakkumuleringspotensialet til de ulike miljøgiftene i Oslofjord-næringsnettet er undersøkt. Eksponering for/akkumulering av disse stoffene er også undersøkt i gråmåke, som representant for «urbane innbyggere». I 2017 er også gråmåke fra ytre Oslofjord analysert. Videre er utvalgte miljøgifter analysert i ærfugl fra indre Oslofjord. Konsentrasjoner av et stort antall kjemiske parametere er kvantifisert i denne undersøkelsen, i tillegg til enkelte biologisk effekt-parametere i torsk. Rapporten fungerer som verdifull dokumentasjon av konsentrasjonene av ulike kjemikalier i ulike deler («compartments») av det marine økosystemet i Indre Oslofjord.

Analyser av stabile isotoper av karbon og nitrogen viste nær identiske resultater/trofiske interaksjoner som i 2015-2016. Biomagnifiseringspotensialet til stoffene i undersøkelsen ble evaluert ved beregning av trofiske magnifiseringsfaktorer (TMF) og flere stoffer, særlig eldre miljøgifter med kjente biomagnifiserende egenskaper, viste som ventet positive sammenhenger mellom (log_{10}-) konsentrasjoner og trofisk posisjon. Dette var også tilfelle når ærfugl ble inkludert i næringsnettet (alle stoffer ble ikke analysert i ærfugl). Arsen (As), sølv (Ag) og PFOSA var stoffer som viste positive sammenhenger mellom (log_{10}-) konsentrasjoner og trofisk posisjon.

Sedimentene i Indre Oslofjord er i utgangspunktet en potensiell kilde for miljøgifter i sedimentlevende bunndyr og således den marine næringskjeden. Flere av stoffene i denne undersøkelsen ble funnet i sediment. Tilførsel til fjorden via overvann og utslippsvann fra

kloakkrenseanlegg ble også vist for flere av stoffene. Konsentrasjoner av enkelte stoffer overskred miljøkvalitetsstandarder i sediment (D5, PCB7, Cu, Zn, As, Ni, Pb, Hg og PFOS), overvann (bisfenol A, MCCP, Cu, Zn og PFOS) og utslippsvann fra kloakkrenseanlegg (D5, MCCP og PFOS).

Dechlorane plus, et flammehemmende middel i plast og polymerer, ble inkludert i Urban fjord-programmet i 2017 og ble detektert i partikkelfaser (partikkelfraksjon i overvann, kloakkslam og sediment). Det ble også funnet i polychaeter, torsk og egg fra gråmåke (fra Indre og Ytre Oslofjord) og ærfugl (fra Indre Oslofjord).

Som rapportert tidligere viste konsentrasjonene av enkelte stoffer funnet i gråmåkeegg fra Oslofjordområdet i 2017 interessante forskjeller fra konsentrasjoner funnet i gråmåkeegg fra mer fjerntliggende marine kolonier (Sklinna og Røst, 2012), som kan tyde på urban påvirkning av måkene fra Oslofjorden. I 2017 ble det også tatt prøver av gråmåke fra ytre Oslofjord. Flere PFAS-forbindelser ble funnet i høyere konsentrasjoner i måke fra ytre Oslofjord, enn i indre Oslofjord, sannsynligvis forbundet med lokal forurensning fra en tidligere flyplass i området.

En potensiell risiko (kumulativ risiko/blandingstoksisitet) for sekundær forgiftning ble påvist for fugler som kan beite på blåskjell, børstemark og sild. Relevante grenseverdier for sekundærforgiftning var ikke tilgjengelig for alle stoffer, og flere detekterte forbindelser ble derfor utelatt fra estimering av kumulativ risiko. Summen av PBDE (BDE-28, -47, -49, -100, -153 og -154) og summen av PCB7 var de viktigste risikofaktorene i alle byttedyr, i tillegg til Cd særlig i blåskjell. Grenseverdiene for sekundærforgiftning brukt for summen av PBDE og summen av PCB7 betraktes som konservative (avledet ved forskjellige metoder enn for de andre stoffene), og resultatene bør tolkes med forsiktighet.

Beregning av den kombinerte risikoen for toksiske effekter i egg fra gråmåke (Indre og Ytre Oslofjord) og ærfugl (Indre Oslofjord) viste at det er en potensiell risiko for effekter i gråmåke. I ærfugl var det indikasjon for mulige effekter om gjennomsnittskonsentrasjoner ble brukt i beregningene, men ikke om medianverdier ble anvendt. De viktigste bidragsyterne til den kumulative risikoen var SumPCB, PBDE-99 og metallene Cu, As og Hg, avhengig av fuglepopulasjon.

Summary

This programme, “Environmental Contaminants in an Urban Fjord” has covered sampling and analyses of sediment and organisms in a marine food web of the Inner Oslofjord in 2017, in addition to samples of blood and eggs from herring gull. Furthermore, optional samples of blood and eggs of herring gull from the Outer Oslofjord were analysed in 2017, as well as samples of blood and eggs of eider duck. The programme also includes inputs of pollutants via surface water (storm water), and sewage treatment plant discharges.

The objective of the programme was to monitor the inputs of chemicals present in a densely populated area and to study how this contaminant input affects a fjord system. The present study represents one step towards the Norwegian Environment Agency’s general aim to:

- Estimate the degree of bioaccumulation of selected contaminants at several trophic levels in marine food chains.
- Connect pollutant exposure of marine organisms to toxic effects at different biological levels, including endocrine disruption and contaminant interactions ("cocktail effects").
- Identify sources and sinks (i.e. the fate) of environmental contaminants in fjord systems and design targeted actions.

Furthermore, there is an intention that data will be used in international chemical regulation, such as REACH and the Stockholm Convention. The programme was also meant to provide data from governmental monitoring in Norway to comply with the requirements of The Water Framework Directive (The Water Regulation/“Vannforskriften”). 2017 represents the fifth year of the Urban Fjord programme. Some changes/improvements have been made in the design from the start in 2013 to 2017.

The bioaccumulation potential of the contaminants in the Oslo fjord food web was evaluated. The exposure to/accumulation of the contaminants was also assessed in herring gull, as an indicator of an urban fjord inhabitant. In 2017, herring gulls from the Outer Oslofjord were also analysed. In addition, selected contaminants in eider duck from the Inner Oslofjord were analysed. A vast number of chemical parameters have been quantified, in addition to some biological effect parameters in cod, and the report serves as valuable documentation of the concentrations of these chemicals in different compartments of the Inner Oslofjord marine ecosystem.

Analyses of stable isotopes of carbon and nitrogen showed nearly identical results/trophic interactions as in 2015-2016. The biomagnifying potential of contaminants was evaluated by calculation of Trophic Magnification Factors (TMFs) and several contaminants, and especially legacy contaminants with well-known biomagnifying properties, displayed a positive significant relationship between (log_{10}-)concentrations and trophic position. This was also the case when eider duck was included in the food web (all compounds were not analysed in eider duck). Arsenic (As), silver (Ag) and PFOSA were contaminants that displayed a positive significant relationship between (log_{10}-)concentrations and trophic position.

The sediments of the inner Oslofjord is a potential source of environmental contaminants to sediment dwelling organisms and the contaminants may thus enter the food chain. Several of

the target compounds of this study were detected in sediment. Inputs of several compounds to the fjord via storm water and effluent water from a sewage treatment plant (STP) is also shown. Concentrations of some compounds exceeded environmental quality standards in sediment (D5, PCB7, Cu, Zn, As, Ni, Pb, Hg and PFOS), storm water (bisfenol A, MCCP, Cu, Zn og PFOS) and STP effluent water (D5, MCCP og PFOS).

Dechlorane plus, a flame retardant in plastics and polymers, was included in the Urban fjord programme in 2017 and was detected in particulate phases, i.e. the particulate fraction in storm water, sewage sludge and sediment. Furthermore, it was found in polychaetes, cod and bird eggs (herring gulls from the Inner and Outer Oslofjord, as well as eider duck from the Inner Oslofjord).

As previously reported, concentrations of specific compounds in eggs of herring gull from the Oslo area in 2017 showed interesting differences from concentrations in herring gull eggs from more remote marine colonies (Sklinna and Røst, 2012), suggesting urban influence on the Oslo gulls. In 2017, gulls from the Outer Oslofjord were also sampled. Several PFAS compounds were found in higher concentrations in the Outer Oslofjord, compared to the Inner Oslofjord, likely associated with local contamination from an old airfield in the area.

A potential risk (cumulative risk/mixture toxicity) of secondary poisoning was identified for birds preying on blue mussels, polychaetes and herring. Proper toxicity data were not available for all substances, thus several detected compounds were excluded from the cumulative risk estimation. The sum of PBDEs (BDE-28, -47, -99, -100, -153 and -154) and the sum of PCB7 were the main risk drivers in all food sources, with the addition of Cd in particularly blue mussels. The toxicity data used for the sum of PBDE and the sum of PCB7 are considered conservative (derived by different methods than for the other substances) and the results should be interpreted with caution.

Calculations of the combined risk of effects in herring gull eggs (Inner and Outer Oslofjord) and eider duck (Inner Oslofjord) showed that there is a potential risk of effects. In eider duck, there was an indication of risk if mean concentrations were used in the calculations, but not when median values were used. The main contributors to the cumulative risk were SumPCB, PBDE-99 and the metals Cu, As and Hg, dependent on bird population.

Content

Attachments:

1. *Appendix: Support parameters (Tables A1-A10)*

   Concentrations in individual samples and composition of (calculated) pooled samples of cod are available as electronic appendix. CAS-no.and/or Chemspider ID are available as electronic appendix.

1. Introduction

"Environmental contaminants in an urban fjord" is a programme designed to monitor discharges of anthropogenic chemicals in a densely populated area and to study how this contaminant input affects a fjord system. The programme addresses inputs of pollutants from potential sources, measurements of contaminant concentrations in different marine species, assessment of bioaccumulation patterns within a food web and estimation of effect risks in organisms. The programme contributes to the Norwegian Environment Agency's ongoing monitoring activity in coastal areas and supplements two other monitoring programmes: "RID - Riverine inputs and direct discharges to Norwegian coastal waters" and "MILKYS - Environmental contaminants in coastal areas".

1.1 Objectives

The environmental monitoring activity in the present programme contributes to the Norwegian Environment Agency's general aim to:

- Estimate the bioaccumulation of selected contaminants at several trophic levels in marine food chains.
- Connect pollutant exposure of marine organisms to toxic effects at different levels of biological organisation, including endocrine disruption and contaminant interactions ("cocktail effects").
- Identify sources and sinks of environmental contaminants in fjord systems ("the fate of the contaminants in a fjord") and designing targeted actions.

The programme will also provide data that will aid to implement the requirements of The Water Framework Directive (The Water Regulation/"Vannforskriften") regarding governmental basic monitoring as well as used in international chemical regulation. The present report (2017) represents the fifth year of the Urban Fjord project. In 2017 two MSc-student from the University of Oslo were affiliated with the programme to look in more detail at bird related issues.

2. Material and Methods

2.1 Sample Collection

Polychaetes, zooplankton (krill), prawns, blue mussel, herring and cod were collected as representatives of a food chain in the inner Oslo Fjord. In addition, sediment was collected. The samples were collected in an area within 4.7 km from Steilene (Figure 1), the autumn of 2017. Herring gull samples (blood and eggs) were also collected within the programme (spring 2017), as a representative of an urban fjord inhabitant. Table 1 shows the sampling plan of the programme. The programme also included samples of storm water, and effluent water and sludge from a waste water treatment plant. Optional samples of eider duck (blood and eggs) were also collected in 2017.

2.1.1 Sediment

Sediment was collected at station Cm21 by means of a van Veen grab (0.15 m^2) from Research Vessel Trygve Braarud. Four grabs of the top layer (0-2 cm in grab samples with undisturbed surface) were prepared[1] for one sample.

2.1.2 Food web of the Inner Oslofjord

Polychaetes, zooplankton (krill), prawns, blue mussel, herring and cod were collected as representatives of a food chain in the inner Oslo Fjord.

Polychaetes were collected at station Cm21 (Figure 1) using a van Veen grab (0.15 m^2) from RV Trygve Braaarud. When possible (dependent on species and mechanical damage), the worms were held in a container of clean seawater for 6-8 hours prior to freezing and analysis. This was done in order to allow the worms to purge any residual sediment from the gut. Material for three pooled samples was collected. The samples consisted of the species listed in Table 2.

Krill (*Euphausiacea*) were collected as representatives of the zooplankton by Midtmeie, southwest of Steilene (Figure 1). A fry trawl was operated from RV Trygve Braarud for this purpose. Material for three pooled samples was collected.

Prawns (*Pandalus borealis*) were caught with benthic trawl from RV Trygve Braarud in the same area as zooplankton (krill), Midtmeie, southwest of Steilene (Figure 1). Material for three pooled samples (of 50 individuals each; size: 69-101 mm) was collected.

Mussels were collected at Steilene (Figure 1) by standard procedures (as in "Contaminants in coastal areas", MILKYS; handpicked, using rake, or snorkelling). Three pooled samples (each of 12-13 shells; shell length 59 to 74 mm) was prepared.

Herring (*Clupea harengus*) were caught with trawl from RV Trygve Braarud at Midtmeie, southwest of Steilene (Figure 1). Material for three pooled samples (of 5 individuals in each; length: 22-28.5 cm, weight: 98-234 g) was collected.

[1] According to the Norwegian Environment Agency guidelines for risk assessment of contaminated sediment (M-409/2015).

Cod (*Gadus morhua*) were caught with trawl from RV Trygve Braarud at Midtmeie, southwest of Steilene (Figure 1). Biometric data for the fish are given in Appendix.

2.1.3 Herring gull

Herring Gull (*Larus argentatus*) blood samples (from adult breeding individuals trapped at nest) and eggs (15 egg samples and 15 blood samples) were sampled at Søndre Skjælholmen (Nesodden municipality; 59.85317 N, 10.7281 E). Biometric data for the birds are given in Appendix. The blood samples were taken from adult birds trapped by walk-in trap placed at the nest, and the blood samples (~5 ml) were taken from a vein under the wing. Adult female and egg were sampled from the same nest.

In 2017, 15 additional samples of blood and eggs from herring gulls in the Outer Oslofjord (Store Revlingen; 59.3966 N, 10.635 E) were collected by the same procedures as for the Inner Oslofjord gulls.

2.1.4 Eider duck

As part of an option under the programme, samples of blood and eggs of Eider duck (*Somateria mollissima*) from the Inner Oslofjord were collected in spring 2017. The samples were from Søndre Skjælholmen (5 females), Husbergøya also in Nesodden municipality (6 females), and Raudskjæra in Asker municipality (4 females). All females were incubating birds trapped at nest late in the incubation period.

2.1.5 Storm water

Storm water samples were collected at one occasion at two specific sampling points (Bryn Ring 3/E6, and Breivoll E6, downstream terminal; Figure 1). The samples were collected from manholes by filling bottles directly in the storm water. Subsequently, the storm water samples were separated into a filtered fraction (hereafter referred to as “dissolved fraction”) and a particulate fraction by filtering (polyethylene (PE) frit, 20 µm porosity prior to analysis of per-and polyfluorinated substances (at NIVA) and Whatman Glass Microfilters GF, pore size 1.2 µm, prior to analysis of other chemical parameters (at NILU)).

2.1.6 Sewage treatment plant

Sludge and treated effleunt water were collected from Bekkelaget Sewage Treatment Plant (STP) at two occasions (June and August). Samples of effluent water were collected by the use of the STPs fixed equipment for collection of 24h-samples (according to rules for accredited sampling). Aliquots were transferred to appropriate flasks for the different analytes.

Table 1
Overview of samples collected for the "Urban Fjord" programme, including optional sampling conducted in 2017.

Species/sample	Matrix	Locality	Frequency	No. for analysis
Sediment	Whole sediment	Cm21	Once per year	1
Polychaetes	Pooled samples, whole individuals	Cm21	Once per year	3 pooled samples
Zooplankton (krill)	Pooled samples, whole individuals	Midtmeie	Once per year	3 pooled samples
Prawns	Pooled samples, soft tissue tails	Midtmeie	Once per year	3 pooled samples
Blue mussel	Pooled samples, soft body	Steilene	Once per year	3 pooled samples
Herring	Muscle	Midtmeie	Once per year	3 pooled samples
Cod	Muscle, liver, bile	Midtmeie	Once per year	15 individuals
Herring gull (blood)	Blood	Søndre skjælholmen and Revlingen *	Once per year	15 individuals
Herring gull (egg)	Egg	Søndre skjælholmen and Revlingen *	Once per year	15 eggs
Eider duck (blood) *	Blood	Søndre skjælholmen, Husbergøya and Raudskjæra	Once	15 individuals
Eider duck (egg) *	Egg	Søndre skjælholmen, Husbergøya and Raudskjæra	Once	15 eggs
Inputs storm water	Water (dissolved) and particulate fraction	See Figure 1	Once per year	4 samples (2 samples of dissolved fraction plus 2 of particulate fraction)
Inputs from Sewage Treatment Plant	Effluent water and sludge	Bekkelaget	Twice per year	4 samples (2 samples of discharge water and 2 samples of sludge)

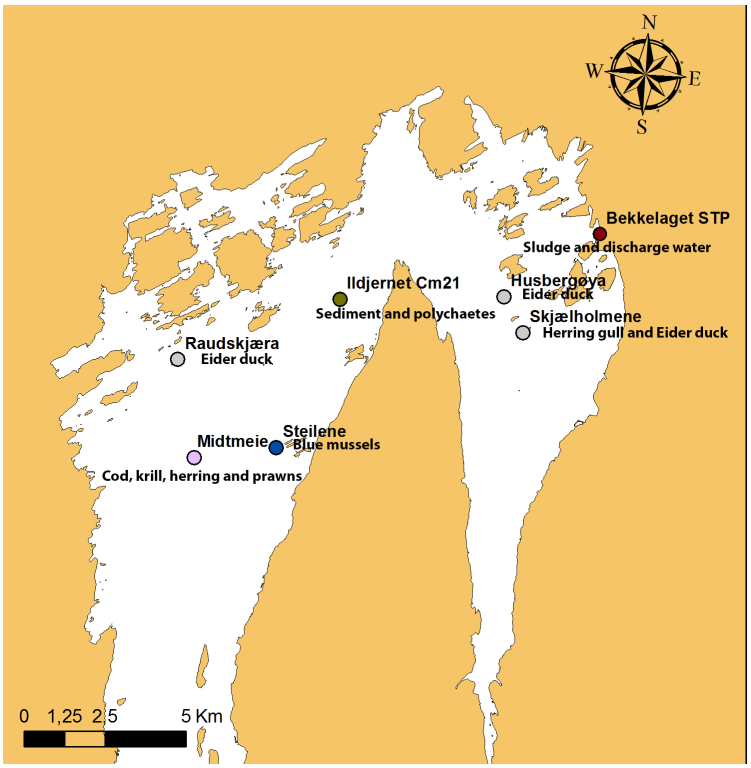
* Optional activity conducted in 2017

Table 2.
Species constituting polychaete samples (grams of each species).

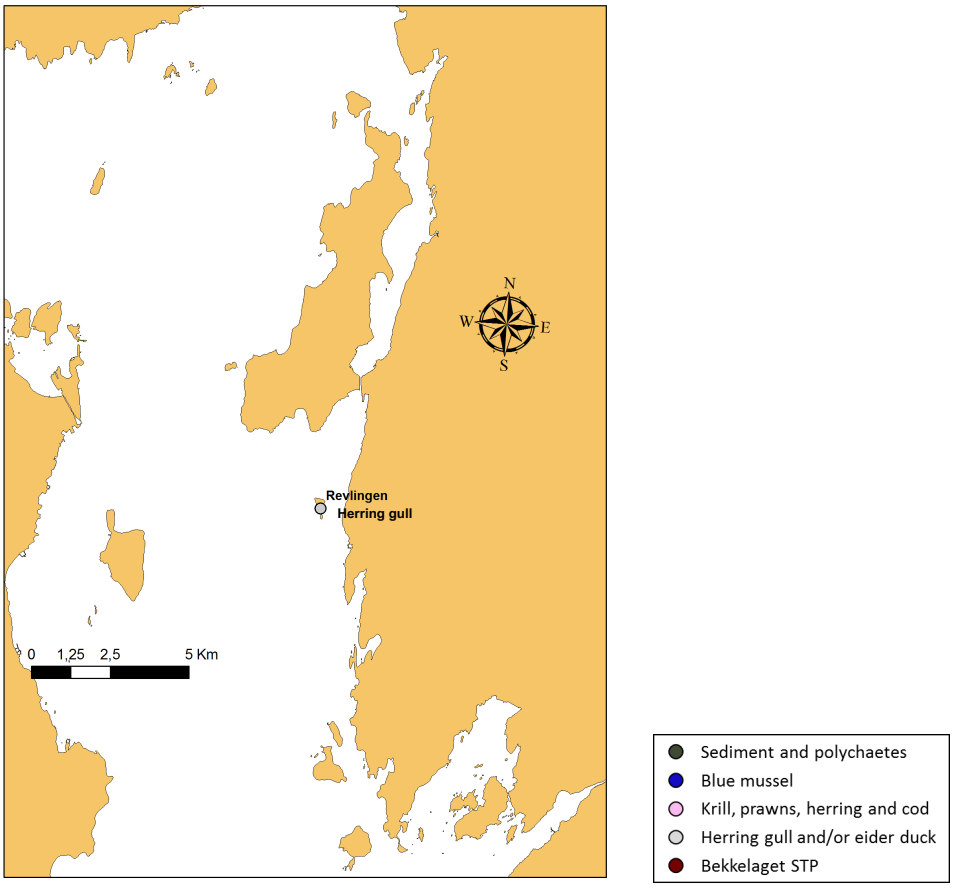
	Inner Oslofjord (Cm21)		
	Repl. 1	**Repl. 2**	**Repl. 3**
P.crassa	0	0	69.9
Lumbrineridae	107.4	0	0
Terbellidae	0	131.3	0
Aphrodita aculeata	0	0	26.9
Misc. *	0	0	74.6
Total (grams)	**107.4**	**131.3**	**171.4**

* *Inter alia*: *Nephtys, Glycera, Goniadidae, Ophelina, Ophiodromus flexuosus, Skoloplos, Spiophanes kroyeri, Scalibregma inflatum.*

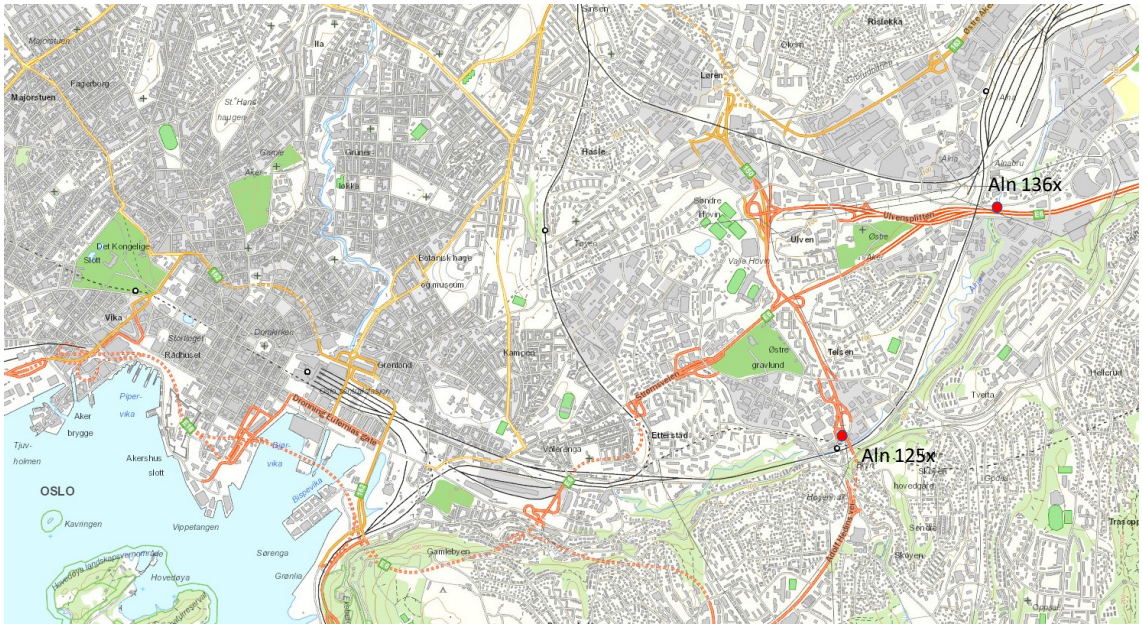
A.


N
W
E
S
Bekkelaget STP
Sludge and discharge water
Ildjernet Cm21
Sediment and polychaetes
Husbergøya
Eider duck
Skjælholmene
Herring gull and Eider duck
Raudskjæra
Eider duck
Midtmeie
Steilene
Blue mussels
Cod, krill, herring and prawns
0 1,25 2,5 5 Km

B.


N
W
E
S
Revlingen
Herring gull
0 1,25 2,5 5 Km
Sediment and polychaetes
Blue mussel
Krill, prawns, herring and cod
Herring gull and/or eider duck
Bekkelaget STP

C.



D.

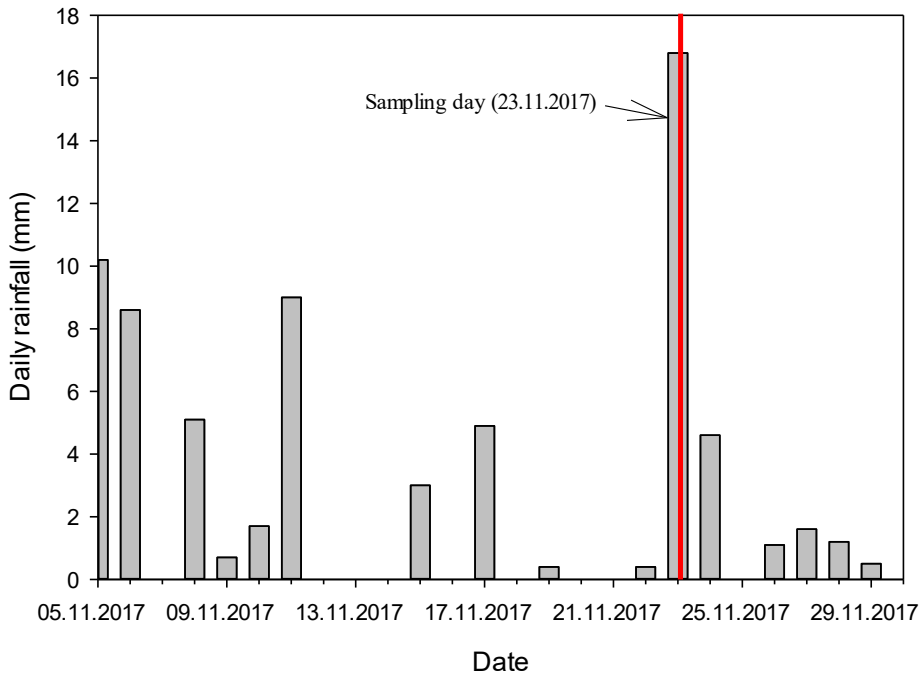


Figure 1. A.: (previous page) Map depicting stations for collection of sediment and polychaetes (green dot), blue mussel (blue dot), and krill, prawns, herring and cod (pink dot) in the Inner Oslofjord, as well as collection of herring gull and eider duck eggs and blood (grey dots) in the inner Oslofjord. The map also shows the location of Bekkelaget STP. B. Map depicting the station for collection of herring gull samples (Revlingen) in the Outer Oslofjord. C.: Map depicting sites for collection of storm water/surface water samples. D.: Overview of time of sampling of storm water/surface water in relation to rainfall (mm/d).

2.2 Chemical analysis, support parameters and biological effect parameters

Tables 3-7 provide a detailed overview of the compounds/parameters analysed in the different samples (main programme and optional in 2017). The samples were analysed at NIVA and NILU. Stable isotopes of carbon and nitrogen were analysed at IFE.

Biological effect parameters (in cod) were also included in the programme (Table 8). These were analysed at NIVA.

Table 3.
Overview: analyses in different matrices from the different localities.

Species/matrix	Locality	Analytes
Sediment	Cm21 (Inner Oslofjord)	Metals, PCB, PFAS, bisphenols, brominated flame retardants, octylphenol, nonylphenol, chloroparafins, UV-chemicals, siloxanes.
Polychaetes	Cm21 (Inner Oslofjord)	Metals, PCB, PFAS, bisphenols, brominated flame retardants, octylphenol, nonylphenol, chloroparafins, UV-chemicals, siloxanes, stable isotopes.
Zooplankton (krill)	Midtmeie	Metals, PCB, PFAS, bisphenols, brominated flame retardants, octylphenol, nonylphenol, chloroparafins, UV-chemicals, siloxanes, stable isotopes.
Prawns	Midtmeie	Metals, PCB, PFAS, bisphenols, brominated flame retardants, octylphenol, nonylphenol, chloroparafins, UV-chemicals, siloxanes, stable isotopes.
Blue mussel	Steilene	Metals, PCB, PFAS, bisphenols, brominated flame retardants, octylphenol, nonylphenol, chloroparafins, UV-chemicals, siloxanes, stable isotopes.
Herring	Midtmeie	Metals, PCB, PFAS, bisphenols, brominated flame retardants, octylphenol, nonylphenol, chloroparafins, UV-chemicals, siloxanes, stable isotopes.
Cod [1]	Midtmeie	Metals, PCB, PFAS, bisphenols, brominated flame retardants, octylphenol, nonylphenol, chloroparafins, UV-chemicals, siloxanes, stable isotopes.
Herring gull (blood)	Søndre skjælholmen and Revlingen [2]	Metals, PCB, PFAS, bisphenols, brominated flame retardants, octylphenol, nonylphenol, chloroparafins, UV-chemicals, siloxanes (incl. M3T(Ph)), antioxidant MB1, stable isotopes.
Herring gull (eggs)	Søndre skjælholmen and Revlingen [2]	Metals, PCB, PFAS, bisphenols, brominated flame retardants, octylphenol, nonylphenol, chloroparafins, UV-chemicals, siloxanes (incl. M3T(Ph)), antioxidant MB1, stable isotopes.

Eider duck [3] (blood)	Søndre skjælholmen, Raudskjæra og Husbergøya	PCB, PFAS, PBDE, Hg, stable isotopes
Eider duck [3] (egg)	Søndre skjælholmen, Raudskjæra og Husbergøya	PCB, PBDE, Hg, stable isotopes
Inputs storm water [4]	See Figure 1	Metals, PCB, PFAS, bisphenols, brominated flame retardants, octylphenol, nonylphenol, chloroparafins, UV-chemicals, siloxanes.
Sewage Treatment Plant [5]	Bekkelaget	Silver (Ag), PFAS, bisphenols, brominated flame retardants, octylphenol, nonylphenol, chloroparafins, UV-chemicals, PFR, siloxanes (incl. M3T(Ph)), antioxidant MB1.

[1] Liver. Mercury in fillet. Bisphenols, octylphenol and nonylphenol preferably in bile.
[2] Additional sampling and analysis of herring gull samples from Revlingen performed in 2017.
[3] Additional sampling and analysis of eider duck samples from Revlingen performed in 2017.
[4] Dissolved and particulate fractions.
[5] Sludge and discharge water.

Table 4.
Overview: Additional analyses performed in 2017.

Species/matrix	Analytes
Sediment, polychaetes, zooplankton (krill), prawns, blue mussel, cod	M3T(Ph), MB1, F53, F53B, monochloroPFOS, decloranplus *, behentrimonium
Herring gull (blood and egg; Inner Oslofjord), Inputs storm water	F53, F53B, monochloroPFOS, decloranplus *, behentrimonium

* In addition, dechloran plus analysed in all samples collected.

Table 5.
Analytes included in the programme. (See the electronic Appendix for CAS-no.). Additional compounds are indicated.

Parameter	Single compounds
Metals	Hg, Pb, Cd, Ni, Ag, Cu (plus Cr, Zn, Fe, As, Sb)
PCB	PCB-28, -52, -101, -118, -138, -153, -180 (plus -18, -31, -33, -37, -47, -66, -74, -99, -105, -114, -122, -123, -128, -141, -149, -156, -157, -167, -170, -183, -187, -189, -194, -206, -209)
PFAS	PFBS, PFHxS, PFOS, PFOSA, 6:2 FTS, 8:2 FTS, 4:2 FTS, PFDS, PFDoS, N-EtFOSE, N-MeFOSE, N-EtFOSA, N-MeFOSA, N-MeFOSAA, N-EtFOSAA) Perfluorinated carboxylic acids (6-15 C-atoms): PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnA, PFDoA, PFTrA, PFTeA, PFPeA (plus PFPS, PFHpS, PFNS and 10:2 FTS)
Brominated flameretardants	PBDEs: BDE-47, -99, -100, -126, -153, -154, -175, -183, -190, -196, -202, -206, -207, and -209. Tetrabromobisphenol A (TBBPA), Decabromodiphenyl ethane (DBDPE), Bis(2-ethylhexyl) tetrabromophtalate (TBPH/BEH-TBP), Hexabromobenzene (HBB), pentabromotoluene (PBT) (plus tribromoanisole, TBA)
Bisphenols	Bisphenol A, bisphenol S, bisphenol F (plus bisphenol AF, AP, B, E, FL, M, Z) (Bisphenol F is also separated in 2,2'- and 4,4'-)
Octyl-/nonylphenol	Octyl-/nonylphenol (isomer-spesifc, i.e. we separate 4- and 4-tert)
UV-chemicals	Octocrylene, benzophenone-3, ethylhexylmethoxycinnamate
Chloroparaffins	SCCP (C10-C13) and MCCP (C14-C17)
Siloxanes	Octamethylcyclotetrasiloxane (D4), decamethylcyclopentasiloxane (D5), dodecamethylcyclohexasiloxane (D6) Tris(trimethylsiloxy) Phenylsilane (M3T(Ph))
Phosphorus flame retardants (PFR)	tri-iso-butylphosphate (TIBP), tributylphosphate (TBP), tri(2-chloroethyl)phosphate (TCEP), tri(1-chloro-2-propyl)phosphate (TCPP), tri(1,3-dichloro-2-propyl)phosphate (TDCP), tri(2-butoxyethhyl)phosphate (TBEP), triphenylphosphate (TPhP), 2-ethylhexyl-di-phenylphosphate (EHDPP), dibutylphenylphosphate (DBPhP), butyldiphenylphosphate (BdPhP), tris(2-ethylhexyl)phosphate (TEHP), tris-o-cresylphosphate (ToCrP), tricresylphosphate (TCrP)
Antioxidant MB1	4,4'-methylenebis[2,6- bis (1,1 dimethylethyl)-phenol]

Table 6.
Specifics regarding compounds analysed in 2017 as an option under the programme. (See electronic Appendix for CAS-no.).

Parameter	Single compounds
M3T(Ph)	Tris(trimethylsiloxy) Phenylsilane (siloxane)
MB1	4,4'-methylenebis[2,6-bis (1,1-dimethylethyl)-phenol]
F53/F53B	F-53 (potassium 1,1,2,2-tetrafluoro-2-(perfluorohexyloxy)ethane sulfonate) F 53B (potassium 2-(6-chloro-1,1,2,2,3,3,4,4,5,5,6,6-dodecafluorohexyloxy)-1,1,2,2-tetrafluoroethane sulfonate)
Decloranplus	Decloranplus, Dec-602, -603 og -604 (plus -601)
Behentrimonium	ATAC-C20 and ATAC-C22

Table 7.
Support parameters included in the programme

Parameter	Specific single parameters	Comment
Stable isotopes	$\delta^{15}N$ and $\delta^{13}C$	In biological matrices
Lipid content (%) in biota		In biological matrices
Weight and length		Fish
Age		Cod
Grain size distribution	Fraction <63 µm	Sediment
TOC		Sediment

Table 8.
Biological effect parameters (in cod)

Parameter	Indicator of
Acetylcholin esterase (AChE)	Inhibition by contaminants such as organophosphates
Other relevant physiological parameters: **Liversomatic index** **Gonadosomatic index**	

2.2.1 Background, target compounds

The metals are naturally occurring elements, but human activities have through history led to increasing amounts of several of them in the environment. In the aquatic environment, inorganic mercury (Hg) may be transformed to methylmercury, especially by bacterial activity. In fish, the majority of the mercury is in the form of methylmercury, which is more bioaccumulative and toxic than inorganic mercury (Wolfe et al. 1998). Cadmium (Cd) has been used e.g. in various industrial processes, such as protecting steel against corrosion. Other applications have e.g. been batteries, pigments, ceramic glaze and surface treatments, but the element is also a contaminant in products, including some types of fertilizer. Cadmium can enter fish by passive diffusion across the gills or by entering the marine food chain at the plankton and microorganisms level and thereby entering fish through the diet. Cadmium is highly toxic to humans and its bioaccumulative properties prevents the reduction of the accumulated body burden (Bosch et al. 2015). Lead (Pb) has a great number of industrial applications, both in its elemental form and in the form of alloys and compounds. The major use of lead has been the manufacture of lead accumulators. Furhermore, tetralkyl lead, R_4Pb, mostly tetraethyl lead is an organic lead species used as anti-knocking agents in leaded gasoline. This application has declined dramatically due to restrictions imposed through environmental legislation. Lead infers with the biosynthesis of porphyrins and heme, eventually leading to anaemia.

Polychlorinated biphenyls (PCBs) are a group of industrial chemicals (209 theoretical congeners), that are also formed as byproduct in different industrial processes and combustion processes. The PCBs have unique physical and chemical properties, such as high thermal and chemical stability and high electrical resistance, hence their application in many industrial applications, such as hydraulic fluids, cooling liquids in transformers and dielectric liquids in capacitors. They have also been applied in plasticizers, lubricants, inks and paints. In Norway the production and use of PCBs was restricted since the 1970s and later banned by law. Immunosuppressive effects endocrine disrupting effects and impairment of reproduction are some toxic effects expressed by PCBs (Safe, 1994).

PFAS compounds have been applied in both industrial processes and consumer products since the 1950s. They may for instance give products water and dirt repellent properties, and they have been used to impregnate textiles and in food packaging. Some of the PFAS compounds have properties that prevent fire and evaporation of volatile compounds, and have therefore been used in firefighting. This was previously the largest source of PFOS emissions in Norway. Firefighting foam with PFOS was banned in 2007.

The brominated flame retardants have been applied in products to prevent fire. In Norway, brominated flame retardants can mainly be found in electrical/electronic products. Brominated flame retardants can also be found in cars, plastic insulation materials (polystyrene), and in textiles, such as furniture and workwear.

There are many different bisphenols available, and bisphenol A is the most known substance. It is used e.g. as raw material for plastics and paints, and may be found in imported plastic products. There is less knowledge regarding other bisphenols, such as bisphenol AF, bisphenol B, bisphenol BP, bisphenol F, bisphenol M and bisphenol S. These substances can be used as a replacement for bisphenol A. Bisphenol S is a substitute for bisphenol A in heat-sensitive paper. Furthermore, bisphenol F and bisphenol B may possibly replace bisphenol A in products made of epoxy resin and polycarbonate, such as epoxy paint and plastic cutlery.

Alkylphenols have been/are used in f.i. textiles, plastic products, paints and lubricants. Nonyl- and octylphenol ethoxylates have been widely used in products such as detergents and cosmetics. Emissions of nonyl- and octylphenols have been substantially reduced the last couple of decades. The decrease is mainly due to reduced application in detergents following regulations.

Short-chained chlorinated paraffins (SCCPs) are banned in Norway, but the compounds may still be found in several imported plastic products. Medium-chained chlorinated paraffins (MCCPs) may also be found in imported products. These substances are primarily applied as softeners and flame retardants and can be found in rubber and PVC used for the production of e.g. cables and floor coverings.

Octocrylene, benzophenone-3 and ethylhexylmethoxycinnamate are used is in sunscreens and other cosmetics to absorb UV rays from the sun, protecting the skin from damage.

Siloxanes have properties that affect the consistency of products such as shampoo and creams to facilitate their use. Siloxanes can otherwise be found in e.g. car wax, paint, insulation materials and cement. Cosmetic products such as soap, skin care products, deodorants and makeup are likely the largest source of siloxane emissions in Norway.

The phosphorus flame retardants have been applied in products to prevent fire. They are widely used in plastics as flame retardants and plasticizers. They are also used as antifoams and as additives in lubricants, hydraulic oils, floor polishers and adhesives.

4,4'-methylenebis[2,6-bis(1,1-dimethylethyl)-phenol (MB1) is used as an industrial anti-oxidant and additive to plastics.

Dechlorane plus is used as a flame retardant in plastics and polymers, such as nylon, polyurethane, polypropylene, neoprene and silicone rubber. As such, it can be found in electronic wires and cables, cars, plastic roofing materials and hard plastic couplings. It may also function as a softener. Dechlorane plus is marketed as an alternative to deca-BDE (BDE-209).

Behentrimonium (ATAC-C20 and ATAC-C22) are quaternary ammonium compounds (QACs). QACs are widely used as ingredients in industrial applications and household products, such as fabric softeners, detergents, disinfectants, preservatives, and different personal care products. Behentrimonium chloride or methosulphate, containing ATAC-C20 and ATAC-C22 are used in personal care products, especially in hair care products.

2.2.2 Analysis of metals

Metal analyses were performed by NILU.

Sample Preparation

Sediment-/sludge- and biota-samples were added supra pure acid and digested at high pressure and temperature in a microwave- based digestion unit (UltraClave). A minimum of two blanks were included with each digestion. Furthermore, reference material (traceable to NIST) was digested with the samples.

Water samples were preserved in original bottles with 1% (v/v) nitric acid.

Instrumental Analysis
Concentrations of nickel (Ni), cadmium (Cd), mercury (Hg), lead (Pb), silver (Ag) and copper (Cu) were determined using inductively coupled plasma mass spectrometer (ICP-MS). All samples, standards and blanks were added internal standard prior to analysis. In addition, Chromium (Cr), zinc (Zn), iron (Fe), arsenic (As) and antimony (Sb) were determined.

Limits of Detection
Detection limits (LoD) and Quantification limits (LoQ) were calculated from 3 times and 10 times the standard deviation of blanks, respectively.

2.2.3 Analysis of PCBs, brominated flame retardants and S/MCCP

Polychlorinated biphenyls (PCBs), brominated flame retardants (TBBPA analysed with phenolic compounds; see Chapter 2.2.5), and short- and medium chained chloroparaffins (S/MCCP) were analysed by NILU.

Extraction
Prior to extraction, the samples were added a mixture of isotope labelled PCBs for quantification purposes.

The water-, sludge-/sediment- and biota-samples were extracted with organic solvents and concentrated under nitrogen flow, followed by a clean-up procedure using concentrated sulphuric acid and a silica column to remove lipids and other interferences prior to analysis.

Analysis
The compounds were quantified on GC-HRMS (Waters Autospec) and/or BG-QToF (Agilent 7200B).

Limits of Detection
The limits of detection (LoD) and quantification (LoQ) were calculated for each sample, using the accepted standard method, i.e. the average of blanks plus 3 and 10 times the standard deviation for blanks, for LoD and LoQ, respectively.

Quality assurance and accreditation
NILU's laboratories are accredited by Norwegian Accreditation for ISO/IEC 17025. NILU is accredited for the analysis of PCBs. For the other compounds, the same quality assurance procedures (as for the accredited compounds) were applied.

2.2.4 Analysis of PFAS

Per- and polyfluorinated substances (PFAS) were analysed by NIVA

Extraction
Prior to extraction, the samples were added a mixture of isotope labelled PFAS, for quantification purposes. Sediment-/sludge-, water- and biota-samples were extracted with organic solvents and use of buffers for pH control. The extracts were cleaned using solid phase extraction (SPE) and active coal if needed (the latter for lipid rich biota samples).

Water samples were concentrated and cleaned up using an SPE column. All samples were concentrated under Nitrogen flow.

Analysis
PFAS compounds were analysed using LC-qTOF-MS.

Limits of Detection
The limits of detection (LoD) and quantification (LoQ) were calculated for each sample, using the accepted standard method; three times the signal/noise ratio (z/n) and 9 times z/n, respectively.

Quality assurance and accreditation
NIVA's laboratory is accredited by Norwegian Accreditation for ISO/IEC 17025. NIVA is not accredited for these particular compounds, but to the extent possible, documentation, preparation, analysis and calculations are performed in accordance with accredited methods.

Samples were analysed in groups with at least one additive standard sample and a blank control. To ensure repeatability, a random sample from each matrix was selected for duplicate analysis.

2.2.5 Analysis of alkylphenols and bisphenols

Alkylphenols and bisphenols (octylphenol, nonylphenol, bisphenol A, S, F, AF, AP, B, E, FL, M og Z, as well as TBBPA) were analysed by NILU.

Extraction
Prior to extraction, the samples were added a mixture of isotope labelled phenols for quantification purposes.

The sediment- and biota-samples were extracted with organic solvents and concentrated under nitrogen flow. Then they were further cleaned with an SPE column to remove interferences prior to analysis. In addition, prior to the extraction and clean-up procedure for biota, liver and bile samples were subjected to an enzyme digestion procedure in order to convert possible Phase II metabolites of phenolic compounds into their respective free forms. Water samples were concentrated and purified on a SPE column. After elution from the SPE column, the water sample extracts were further concentrated under nitrogen and subjected to instrumental analysis.

Analysis
All samples were analysed by LC-QToF (Agilent 65/50), or LC-ToF (Waters Premier).

Limits of Detection
The limits of detection (LoD) and quantification (LoQ) were calculated for each sample, using the accepted standard method, i.e. the average of blanks plus 3 and 10 times the standard deviation for blanks, for LoD and LoQ, respectively.

Quality assurance and accreditation
NILU's laboratories are accredited by Norwegian Accreditation for ISO/IEC 17025. NILU is not accredited for the analysis of alkylphenols and bisphenols, but as far as possible, the

documentation, sample preparation, analysis and calculation procedures were conducted according to the accredited methods.

2.2.6 Analysis of UV-chemicals

UV-chemicals (octocrylene, benzophenone and ethylhexylmethoxycinnamate) were analysed by NIVA. The methods are modified from earlier validated and published methods developed at NIVA (Langford et al. 2008; 2009; 2011; 2015; Thomas et al. 2014).

Extraction of UV-chemicals

Homogenized biota samples were added isotope labelled internal standards for quantification purposes. Then they were extracted twice with a combination of solvents. Extracts were concentrated under nitrogen flow and cleaned up using gel permeation chromatography (GPC) and/or SPE, dependent on complexity of matrix.

Analysis of UV-chemicals

UV-chemicals were analysed using GC-MSD (Agilent) or APGC-Vion (Waters).

Limits of Detection

The limits of detection (LoD) and quantification (LoQ) were calculated for each sample, using the accepted standard method; three times the signal/noise ratio (z/n) and 9 times z/n, respectively.

Quality assurance and accreditation

NIVA's laboratory is accredited by Norwegian Accreditation for ISO/IEC 17025. NIVA is not accredited for these particular compounds, but to the extent possible, documentation, preparation, analysis and calculations are performed in accordance with accredited methods.

2.2.7 Analysis of siloxanes

Siloxanes, i.e. octamethylcyclotetrasiloxane (D4), decamethylcyclopentasiloxane (D5), dodecamethylcyclohexasiloxane (D6) and M3T(Ph) were analysed by NILU – Norwegian Institute for Air Research. Already established methods based on liquid/liquid extraction (Warner et al. 2010, Warner et al. 2012) were used to extract and quantify siloxanes, in addition to headspace extraction techniques to analyse siloxanes in water and sediments.

Extraction

Sediment and biota tissues were extracted using solid-liquid extraction with a biphasic solvent system of acetonitrile and hexane. Extraction of water samples was performed using headspace extraction

Analysis

Collected extracts from sediment-/sludge- and biota tissues were analysed using Concurrent solvent recondensation large volume injection gas chromatography mass spectrometry (CSR-LVI-GCMS; Companioni-Damas et al. 2012). For water analysis, 2 ml of extracted headspace was directly injected onto a GCMS (Sparham et al. 2008).

Limits of Detection
The limits of detection (LoD) and quantification (LoQ) were calculated for each sample using the accepted standard method, i.e. the average of blanks plus 3 and 10 times the standard deviation for blanks, for LoD and LoQ, respectively.

Quality assurance and accreditation
NILU has extensive experience with analysis of siloxanes. The greatest risk in the analysis is background contamination, as these chemicals (D4, D5 and D6) are applied in e.g. skin care products. Using a state-of-the-art cleanroom and clean bench technologies, NILU is capable of performing trace analysis of these compounds in matrices from pristine environments, including the Arctic (Krogseth et al. 2013; Warner et al. 2013).

NILU's laboratories are accredited by Norwegian Accreditation for ISO/IEC 17025. NILU is not accredited for the analysis of siloxanes. However, to the extent possible, documentation, preparation, analysis and calculations were performed in accordance with accredited methods. NILU has previously participated in a laboratory intercalibration of siloxanes (McGoldrick et al. 2011) and has also worked closely with the industry in Artic monitoring programmes to develop methods to enhance result accuracy and limit reporting of false positives (Warner et al. 2013).

Samples were extracted and analysed in batches with a minimum of 3 procedural blanks to assess background contamination and calculate LOD and LOQ per extraction batch. As the sample matrix can contribute to the overall background response, procedural blanks were run both before and after samples to ensure results were above detection limits and not an artefact of background variation.

Field blanks were used to assess any potential contamination that occurred during sample collection and preparation. Each field blank consisted of approximately 3 grams of XAD-2 sorbent in filter bags of polypropylene/cellulose. XAD-2 sorbent was cleaned using a 1:1 mixture of hexane:dichloromethane and dried overnight in a clean cabinet equipped with a HEPA- and charcoal filter to prevent contamination from indoor air. Filter bags were cleaned by ultrasonic treatment in hexane for 30 min. Subsequently, hexane was removed and substituted with clean dichloromethane and the field blanks were sonicated once more for 30 min. After ultrasonic treatment, filter bags were placed in a clean cabinet to dry under similar conditions as the XAD-2 sorbent. Once dry, XAD-2 sorbent was transferred to filter bags and sealed in polypropylene containers to be sent for sampling purposes. Several field-blanks were stored at NILU’s laboratories (hereafter called reference blanks) and analysed to determine reference concentrations before sampling. The field blanks for sampling purposes were exposed and handled in the field during sampling and during preparation of samples. The results from the analysis of the field blanks are presented in Table 9.

Table 9.
Results of the analysis of siloxanes in (field and reference) blanks, consisting of XAD resin in filter bags of polypropylene/cellulose.

Description of sampling/purpose	D4 (ng/g)	D5 (ng/g)	D6 (ng/g)	M3T(Ph) (ng/g)
Gull eggs field blank	14.5	4	1.9	-
Gull eggs field blank control	1	0.7	1.3	-
Gull blood field blank	3.2	0.9	0.9	-
Gull blood field blank control	0.9	0.4	1	-
Misc. biota field blank	3	2.3	1.9	-
Misc. biota field blank ref.	1.5	1.5	1.8	-
Cod liver/herring field blank	1	1.1	1.4	-
Cod liver/herring field blank ref.	1.8	1.1	1.3	-

2.2.8 Analysis of PFR

Phosphorus flame retardants (PFRs) were analysed by NILU.

Extraction
Prior to extraction, the samples were added a mixture of isotope labelled PFR standards, for quantification purposes.

The water-, sediment-/sludge- and biota-samples were extracted with organic solvents and concentrated under nitrogen flow, followed by a clean-up procedure using a silica column to remove lipids and other interferences prior to analysis.

Analysis
PFR compounds were quantified on a Thermo TSQ Vantage UPLC/MS-MS.

Limits of detection
The limits of detection (LoD) and quantification (LoQ) were calculated for each sample, using the accepted standard method, i.e. the average of blanks plus 3 and 10 times the standard deviation for blanks, for LoD and LoQ, respectively.

Quality assurance and accreditation
NILU's laboratories are accredited by Norwegian Accreditation for ISO/IEC 17025. NILU is not accredited for the analysis of PFRs, but the same quality assurance procedures (as for the accredited compounds) were applied for the analyses of these compounds.

2.2.9 Analysis of antioxidant MB1

Antioxidant MB1 was analysed by NILU, with the same extraction methods as described for PCBs, brominated flame retardants and S/MCCP.

Extraction
The water-, sludge-/sediment- and biota-samples were extracted with organic solvents and concentrated under nitrogen flow, followed by a clean-up procedure using concentrated sulphuric acid and a silica column to remove lipids and other interferences prior to analysis.

Analysis
Antioxidant MB1 was analysed using GC-MS.

Limits of Detection
The limits of detection (LoD) and quantification (LoQ) were calculated for each sample, using the accepted standard method, i.e. the average of blanks plus 3 and 10 times the standard deviation for blanks, for LoD and LoQ, respectively.

Quality assurance and accreditation
NILU's laboratories are accredited by Norwegian Accreditation for ISO/IEC 17025. NILU is not accredited for the analysis of antioxidant MB1, but as far as possible, the documentation, sample preparation, analysis and calculation procedures were conducted according to the accredited methods.

2.2.10 Analysis of M3T(Ph)

M3T(Ph) was analysed by NILU – Norwegian Institute for Air Research. This compound was extracted and analysed with the siloxanes (D4, D5 and D6), as described above (Chapter 2.2.7).

Extraction
Already established methods based on liquid/liquid extraction (Warner et al. 2010, Warner et al. 2012) was used to extract M3T(Ph) with the siloxanes (see above; Chapter 2.2.7).

Analysis
Samples were analysed using Concurrent solvent recondensation large volume injection gas chromatography mass spectrometry (CSR-LVI-GCMS; Companioni-Damas et al. 2012).

Limits of Detection
The limit of detection (LoD) and quantification (LoQ) were calculated for each sample using the accepted standard method, i.e. the average of blanks plus 3 and 10 times the standard deviation for blanks, for LoD and LoQ, respectively.

Quality assurance and accreditation
NILU's laboratories are accredited by Norwegian Accreditation for ISO/IEC 17025. NILU is not accredited for the analysis of M3T(Ph). However, to the extent possible, documentation, preparation, analysis and calculations were performed in accordance with accredited methods.

2.2.11 Analysis of F53, F53B and monochloroPFOS

F53, F53B and monochloroPFOS were analysed by NIVA. Extraction and analysis were as described for PFAS, above (Chapter 2.2.4).

Extraction
Samples were extracted with organic solvents and use of buffers for pH control. The extracts were cleaned using solid phase extraction (SPE) and active coal if needed (the latter for lipid rich biota samples). Water samples were concentrated and cleaned up using an SPE column. All samples were concentrated under Nitrogen flow.

Analysis
F53, F53B and monochloroPFOS were analysed using LC-qTOF-MS.

Limits of Detection
The limits of detection (LoD) and quantification (LoQ) were calculated for each sample, using the accepted standard method; three times the signal/noise ratio (z/n) and 9 times z/n, respectively.

Quality assurance and accreditation
NIVA's laboratory is accredited by Norwegian Accreditation for ISO/IEC 17025. NIVA is not accredited for these particular compounds, but to the extent possible, documentation, preparation, analysis and calculations are performed in accordance with accredited methods.

Samples were analysed in groups with at least one additive standard sample and a blank control. To ensure repeatability, a random sample from each matrix was selected for duplicate analysis.

2.2.12 Analysis of Decloranplus and related compounds

Dechloranplus was analysed by NILU, with the same extraction methods as described for PCBs, brominated flame retardants and S/MCCP.

Extraction
The water-, sludge-/sediment- and biota-samples were extracted with organic solvents and concentrated under nitrogen flow, followed by a clean-up procedure using concentrated sulphuric acid and a silica column to remove lipids and other interferences prior to analysis.

Analysis
Antioxidant MB1 was analysed using GC-MS.

Limits of Detection
The limits of detection (LoD) and quantification (LoQ) were calculated for each sample, using the accepted standard method, i.e. the average of blanks plus 3 and 10 times the standard deviation for blanks, for LoD and LoQ, respectively.

Quality assurance and accreditation
NILU's laboratories are accredited by Norwegian Accreditation for ISO/IEC 17025. NILU is not accredited for the analysis of Dechloranplus, but as far as possible, the documentation, sample preparation, analysis and calculation procedures were conducted according to the accredited methods.

2.2.13 Analysis of Behentrimonium

Behentrimonium was analysed by NIVA.

Extraction
Sediment-/sludge- and biological samples were freeze dried and added internal standard (EADAC-C12) prior to extraction with methanol and hydrochloric acid (HCl) in an ultrasonic bath. The extraction was repeated twice. The extract was evaporated to dryness and dissolved in 50:50 vol/vol methanol and water. Water samples were extracted by use of SPE Strata X cartridge, followed by the following steps: (1.) Conditioning, (2.) washing (water) and (3.) elution (ACN, acetic acid and water).

Analysis
The extracts were injected and analysed using UPLC-HRMS with RP-column (Luna C18; 150 mm, 2mm, 5 µm).

Limits of Detection
The limits of detection (LoD) and quantification (LoQ) were calculated for each sample, using the accepted standard method; three times the signal/noise ratio (z/n) and 9 times z/n, respectively.

Quality assurance and accreditation
NIVA's laboratory is accredited by Norwegian Accreditation for ISO/IEC 17025. NIVA is not accredited for behentrimonium, but to the extent possible, documentation, preparation, analysis and calculations are performed in accordance with accredited methods.

2.2.14 Support parameters

Stable isotopes of nitrogen and carbon were analysed by IFE. Analysis of nitrogen and carbon isotopes was done by combustion in an element analyser, reduction of NOx in Cu-oven, separation of N_2 and CO_2 on a GC-column and determination of $\delta^{13}C$ and $\delta^{15}N$ at IRMS (Isotope Ratio Mass Spectrometer).

Trophic level was calculated as follows (assuming a 3.8 increase per full trophic level; Hobson and Welch, 1992; and that blue mussel inhabit trophic level 2, filtrating algal particles on trophic level 1):

$$TL_{consumer} = 2 + (\delta^{15}N_{consumer} - \delta^{15}N_{blue\ mussel})/3.8$$

Captive-rearing studies on piscivorous birds indicate that the $\delta^{15}N$ isotopic fractionation factor between bird diet and tissue is less than that derived for other trophic steps, most likely linked to the fact that birds produce uric acid (Mizutani et al. 1991). According to Mizutani et al (1991) an isotopic fractionation factor of +2.4 ‰ is appropriate. Thus, the following equation was used to calculate the trophic level of herring gulls and eider ducks:

$$TL_{herring\ gull} = 3 + (\delta^{15}N_{herring\ gull} - (\delta^{15}N_{blue\ mussel} + 2.4))/3.8$$

Lipid content in biological samples was determined gravimetrically during extraction for chemical analyses.

Weight and length of fish were determined before dissection.

The age of the cod was read from otoliths. The age was read by counting the number of opaque zones (summer zones) and hyaline zones (winter zones).

Grain size distribution (fraction of particles <63 µm) in sediment was determined according to procedures described by Krumbein and Petttijohn (1938).

Total organic carbon content (TOC) in sediment was determined by catalytic combustion in an element analyser.

2.2.15 Biological effect parameters (cod)

Acetylcholinesterase (AChE)
Inhibition of Acetylcholinesterase (AChE) was measured in the microsomal fraction of muscle samples of cod, using methods described by Bocquené and Galgani (1998).

In addition to AChE, the following physiological parameters were measured/calculated: liversomatic index (LSI) and gonadosomatic index (GSI). These are measured of liver weight and gonad weight, respectively, relative to body mass:

Liversomatic and gonadosomatic indices

$$\text{Liversomatic index (LSI)} = \frac{[\text{liver weight (g)} \times 100]}{\text{body mass (g)}}$$

$$\text{Gonadosomatic index (GSI)} = \frac{[\text{gonad weight (g)} \times 100]}{\text{body mass (g)}}$$

2.3 Data treatment

Statistical analyses (linear regressions; general linear models) were performed with the use of Statistica software (Ver 13.1; Statsoft/Dell). A significance level of $\alpha = 0.05$ was chosen. When appropriate, data were $\log_{10}$-transformed.

When results are below LoD (especially when this occurs in many samples), the value of the information is reduced, and there are challenges regarding presentations and statistical evaluation. For the purpose of calculating mean concentrations, we have assigned these samples/parameters a value of zero. In regression models, we have omitted samples with non-detects from processing ("case-wise deletion").

It has earlier been pointed out (Ruus et al. 2015; The Norwegian Environment Agency M-375) that there was a need for a more balanced design, in terms of the number of individual samples from each species in the food web (when possible biomagnification of compounds in the Inner Oslofjord food web was evaluated). Therefore, pooled samples of cod (3 samples constituted of 5 individuals each) are constructed mathematically (mean of the 5 individuals) to obtain 3 samples of each species in the food web (in the same manner as in the 2015- and 2016-programmes; Ruus et al. 2016; Ruus et al. 2017; The Norwegian Environment Agency M-601 and M-812). The individuals were assigned to the different "pooled" samples according to

their length (the five smallest fish in one “pooled” sample, the five largest fish in one “pooled” sample, and the remaining five fish in one “pooled” sample).

When exploring correlations between contaminant concentrations and trophic position, concentrations of the following contaminants were expressed on a wet weight basis: Metals, PFASs (including F53, F53B and monochloroPFOS) and phenolic compounds. The concentrations of the following contaminants were expressed on a lipid weight basis: PCBs and other organochlorine compounds, chlorinated paraffins, brominated flame retardants, siloxanes (including M3T(Ph)), UV-filters, antioxidant MB1 and Decloranplus. Behentrimonium was expressed at both wet weight and lipid weight basis when exploring correlations between contaminant concentrations and trophic position.

When exploring correlations between contaminant concentrations and biochemical response parameters (such AChE activity), concentrations were expressed on a wet weight basis.

Trophic Magnification Factors (TMFs) were calculated from statistically significant relationships: Log_{10}[Contaminant] = a + b(Trophic position)
as TMF = 10^b.

2.3.1 Mixture toxicity / cumulative risk

A conceptual framework for environmental risk assessment of chemical mixtures has been proposed based on an approximation to concentration addition (CA) (Backhaus and Faust, 2012). In the proposed framework, the environmental risk of chemical mixtures is assessed through a tiered approach using available effect data (NOEC and EC50 values) and predicted or measured exposure concentrations (PEC or MEC). In the first tier a risk quotient (RQ) is calculated by summing up the ratios between exposure concentrations (MEC or PEC) and predicted no effect concentrations (PNEC) for all chemicals in the mixture. Backhaus and Faust (2012) showed that summation of PEC/PNEC ratios can serve as a justifiable, conservative, first-tier approach to CA. If the resulting RQ is ≥ 1, there is a potential environmental risk and the next tier should be initiated. In tier 2, the environmental risk of the chemical mixture is assessed for each species group (e.g. algae, crustaceans, fish) by summing up the toxic units (TU = MEC/EC50) for all chemicals in the mixture. The RQ is obtained by application of an appropriate assessment factor on the highest sum of TUs (STU), and a value ≥ 1 is indicative of an environmental risk. Concentration Addition as well as Independent Action can be applied to external (aqueous) or internal (in-biota) concentrations, as long as exposure and hazard estimates relate to the same compartment.

This (or similar) approach(es) has been used in several studies to assess the environmental risk of chemical mixtures detected in the aquatic environment (Backhaus and Karlsson, 2014; Bundschuh et al. 2014; Finizio et al. 2005; Moschet et al. 2014; Petersen et al. 2013), and in biota (Herzke et al. 2014, 2015; The Norwegian Environment Agency M-261 and M-354).

In order to assess whether the mixture of contaminants measured in the organisms pose a risk to their predators, measured concentrations (MEC) in blue mussels, polychaetes, and herring, and available predicted no effect concentrations for secondary poisoning ($PNEC_{pred}$, $PNEC_{oral}$, or $(E)QS_{biota,\ secpois}$) or human health ($(E)QS_{biota,\ hh}$) were used to calculate the sum of $MEC/PNEC_{pred}$ ratios. The average of three measured concentrations was used as MEC for blue mussels, polychaetes and herring. It should be noted that $(E)QS_{biota,hh}$ values are calculated in a different way than the values for secondary poisoning as the tolerable daily intake (TDI) or acceptable daily intake (ADI) for humans are used instead of PNEC values, potentially making this value lower and thus more conservative than the $PNEC_{pred}$, $PNEC_{oral}$ and $EQS_{biota,\ sec.pois.}$values. $PNEC_{pred}$, $PNEC_{oral}$ and $(E)QS_{biota,\ secpois}$ values also have different protection goals than the $(E)QS_{biota,\ hh}$. The $(E)QS_{biota,\ hh}$ values are set to protect humans from adverse effects resulting from the consumption of chemical-contaminated food (fish, molluscs, crustaceans, etc), whereas the protection goal of $QS_{biota,\ secpois}$ is to protect top predators, such as birds and mammals, from risks of secondary poisoning brought about by consuming toxic chemicals in their prey. Therefore, $PNEC_{pred,}$ $PNEC_{oral}$ and $(E)QS_{biota,\ secpois}$ values were used as far as possible to avoid overestimation of the risk and $(E)QS_{biota,\ hh}$ values were only used for substances or substance groups where no other values were found. In cases where several PNECs for secondary poisoning were found, the lowest one was used. Only the compounds listed in Table 17 (see Chapter 3.6) could be included in the cumulative risk assessment for secondary poisoning. The $MEC/PNEC_{pred}$ ratios were summed and a potential risk was identified by a sum ≥ 1.

The potential risk of effects on gulls and eider ducks brought about by the level of measured contaminants in their eggs were assessed. Available effect data for exposure in eggs compiled

and assessed by Andersen et al. (2014) were used in the assessment. The median value of 15 egg concentrations was used as MEC. The sum of MEC/effect data for all possible compounds was calculated and a sum ≥1 was indicative of a potential risk to the birds.

As $PNEC_{pred}$ values and effect data were only available for a few of the detected compounds, the mixture risk assessment performed in this study is not considered complete but is thought to give an indication of which food source pose the highest risk for predators, and an indication of the potential risk drivers.

3. Results and Discussion

The results of the chemical analyses (and lipid content of biological samples) are given in the electronic Appendix, where also analyses falling below LoD are indicated together with the values of the LoDs.

3.1 Stable isotopes

The results of the individual stable isotope analysis are given in Appendix (Tables A3-A10).

Stable isotopes of carbon and nitrogen are useful indicators of food origin and trophic levels. $\delta^{13}C$ gives an indication of carbon source in the diet or a food web. For instance, it is in principle possible to detect differences in the importance of autochthonous (native marine) and allochthonous (watershed/origin on land) carbon sources in the food web, since the $\delta^{13}C$ signature of the land-based energy sources is lower (greater negative number). Also $\delta^{15}N$ (although to a lesser extent than $\delta^{13}C$) may be lower in allochthonous as compared to autochthonous organic matter (Helland et al. 2002), but more important, it increases in organisms with higher trophic level because of a greater retention of the heavier isotope (^{15}N). The relative increase of ^{15}N over ^{14}N is 3-5‰ per trophic level (Layman et al. 2012; Post 2002), and provides a continuous descriptor of trophic position. It is also the basis for Trophic Magnification Factors (TMFs) that give the factor of increase in concentrations of contaminants, and have been amended to Annex XIII of the European Community Regulation on chemicals and their safe use (REACH) for possible use in weight of evidence assessments of the bioaccumulative potential of chemicals as contaminants of concern.

In the present report, the stable isotope data have been reviewed partly to indicate possible different energy sources for the organisms/individuals in question. Secondly, trophic level is calculated from $\delta^{15}N$ for the organisms to assess possible biomagnification of the compounds/contaminants in question in the Inner Oslofjord food web.

It has previously been noted (Ruus et al. 2014; Ruus et al. 2015; Ruus et al. 2016; Ruus et al. 2017; The Norwegian Environment Agency M-205, M-375, M-601 and M-812) that herring gull sampled in the Inner Oslofjord display low $\delta^{15}N$ and low $\delta^{13}C$, relative to the marine species sampled in the programme. This indicates that important food items for the gull are not related to the marine food web sampled. Herring gull is therefore treated separately (not as part of the food web) in the present study (as in the “Urban fjord” programme in 2015 and

2016; Ruus et al. 2016; Ruus et al. 2017; The Norwegian Environment Agency M-601 and M-812).

Some changes were made to the programmme from 2016 to 2017, such as inclusion of eider duck (inner Oslofjord) and additional herring gulls from the Outer Oslofjord (Reference), However, the aquatic food web sampled was identical to that in 2015-2016. The results of the stable isotope analysis (Figure 2 A) suggest that the species sampled in 2015-2017 well represent members of the marine food web of the Inner Oslofjord, as the differences in $\delta^{15}N$ seem to reflect expected trophic relationships; blue mussel (filters particulate organic matter from the water) < zooplankton (herbivore) = polychaetes (different modes of living, largely detritivorous) < herring (pelagic fish feeding on zooplankton) = prawns (some scavenging behaviour) < cod (mesopelagic fish, predator on fish and benthic organisms). The food web spans over 2 to 3 (~2.3) trophic levels with blue mussel defined at trophic level 2 (see Chapter 2.2.14), polychaetes and zooplankton (krill) at trophic level 3.0 and 3.4, respectively, prawns and herring at trophic level 3.8 and 3.5, respectively, and cod at trophic level 4.3 in average (assuming an increase in $\delta^{15}N$ of 3.8‰ per integer trophic level). As such the isotopic signatures of the species in the food web were nearly identical to those observed in 2015-2016 (Ruus et al. 2016; The Norwegian Environment Agency M-601; Ruus et al. 2017; The Norwegian Environment Agency M-812), although with one cod sample with low $\delta^{13}C$ and $\delta^{15}N$ ratios (for unknown reasons).

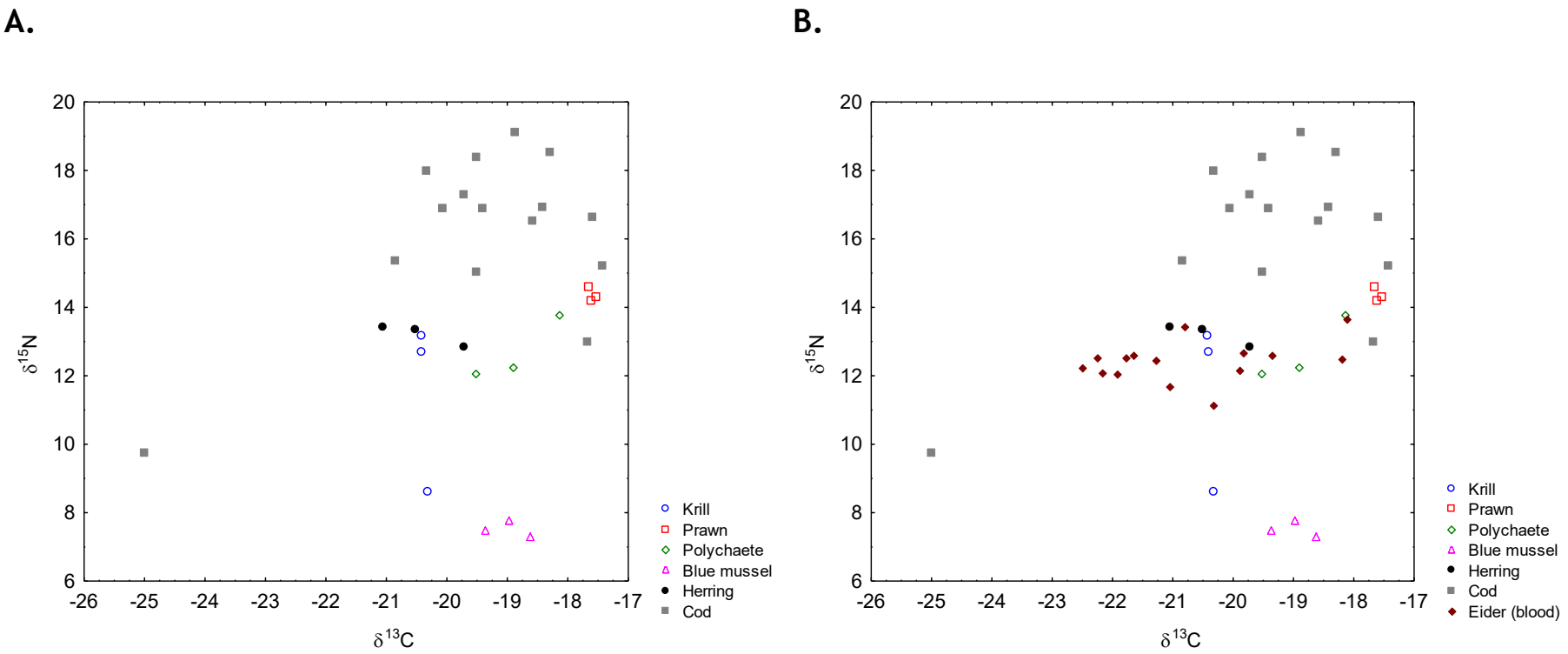


Figure 2. $\delta^{13}C$ plotted against $\delta^{15}N$ in organisms from the inner Oslofjord marine food web (A.), also with eider duck (blood) included (B.).

The isotopic signatures of the herring gulls showed the same patterns as in 2015-2016 (Ruus et al. 2016; The Norwegian Environment Agency M-601; Ruus et al. 2017; The Norwegian Environment Agency M-812). When herring gull matrices (blood and eggs) are evaluated (Figure 3), it can be seen that the matrices show similar $\delta^{15}N$. Herring gull would therefore be placed on approximately the same average trophic level regardless of matrix. The $\delta^{13}C$ ratio is, however, higher in blood than in eggs likely related to different lipid content. It should be noted that samples were not treated to remove carbonates or lipid before stable isotope analysis. The C:N ratio was measured (Appendix, Tables A3-A6) and a C:N ratio of >3.5 implies the presence of lipids, which may somewhat confound $\delta^{13}C$ interpretation, since lipids are ^{13}C -depleted relative to proteins (Sweeting et al. 2006). Eggs showed a higher C:N ratio than blood (Appendix, Tables A3-A6). Figure 3 also displays the isotopic signatures of eider duck (blood and egg), and the same applies: the matrices show similar $\delta^{15}N$, while the $\delta^{13}C$ ratio appear somewhat higher in blood than in eggs, likely related to different lipid content.

In 2017, Herring gull samples (blood and egg) were also collected in the Outer Oslofjord (Revlingen). Figure 3 also suggests somewhat higher $\delta^{15}N$ and $\delta^{13}C$ ratios in the Outer Oslofjord gulls, than in the Inner Oslofjord gulls. (no statistical differences in $\delta^{13}C$; p=0,0512 and p=0,0619 in egg and blood, respectively, but significant differences in $\delta^{15}N$; p=0,0004 and p=0,0001 in egg and blood, respectively; Mann-Whitney U). This could be related to a different baseline in the signatures, or different feeding preferences, if the Outer Oslofjord gulls including more diet items of marine origin than the inner Oslofjord gulls. Analyses of stable isotopes in blue mussels from both the Inner and Outer Oslofjord (Green et al. 2017) suggest no large differences in baseline between the two areas.

Analysis of samples (blood and egg) from eider duck from the Inner Oslofjord was also an addition to the programme in 2017. As can be seen from Figure 3, $\delta^{15}N$ and $\delta^{13}C$ appear higher in the eider duck, than in the herring gull from the Inner Oslofjord (statistical significant differences for both $\delta^{15}N$ and $\delta^{13}C$ in both blood and eggs; p=0,000003; Mann-Whitney U). As such, the isotopic signatures of the eider duck correspond much better with a member of the Inner Oslofjord Marine food web (Figure 2 B.).

Regarding the birds (herring gulls and eider duck), adult female and egg were sampled from the same nest (i.e. mother and future offspring). This is reflected in the isotopic signatures, as significant relationships were found between egg and blood ($\delta^{13}C$ herring gull: R^2=0.49; p=0.00002; $\delta^{13}C$ eider duck: R^2=0.38; p=0.014; $\delta^{15}N$ herring gull: R^2=0.66; p=0.00000; $\delta^{15}N$ eider duck: R^2=0.57; p=0.0011; Figure 4).

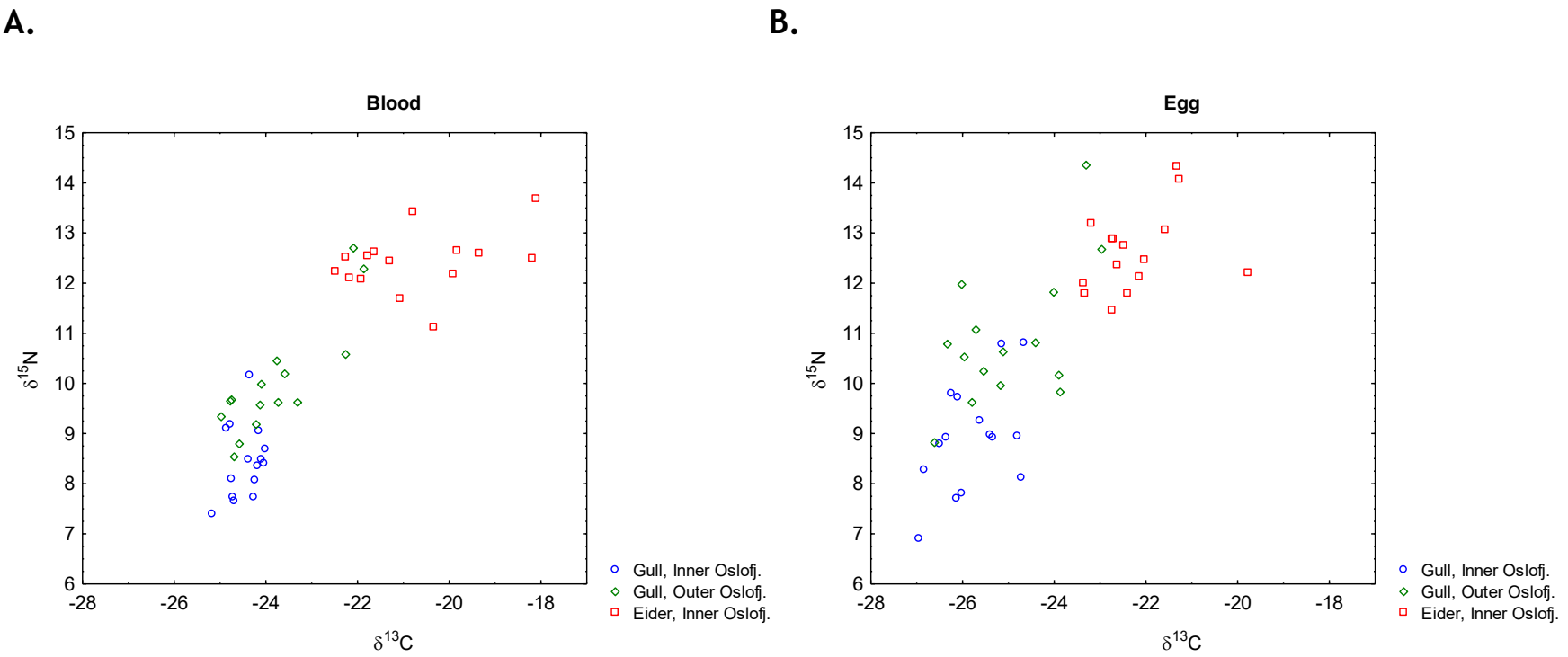


Figure 3. $\delta^{15}N$ plotted against $\delta^{13}C$ in blood (A.) and eggs (B.) of herring gull (Inner and Outer Oslofjord, respectively) and eider duck (Inner Oslofjord).

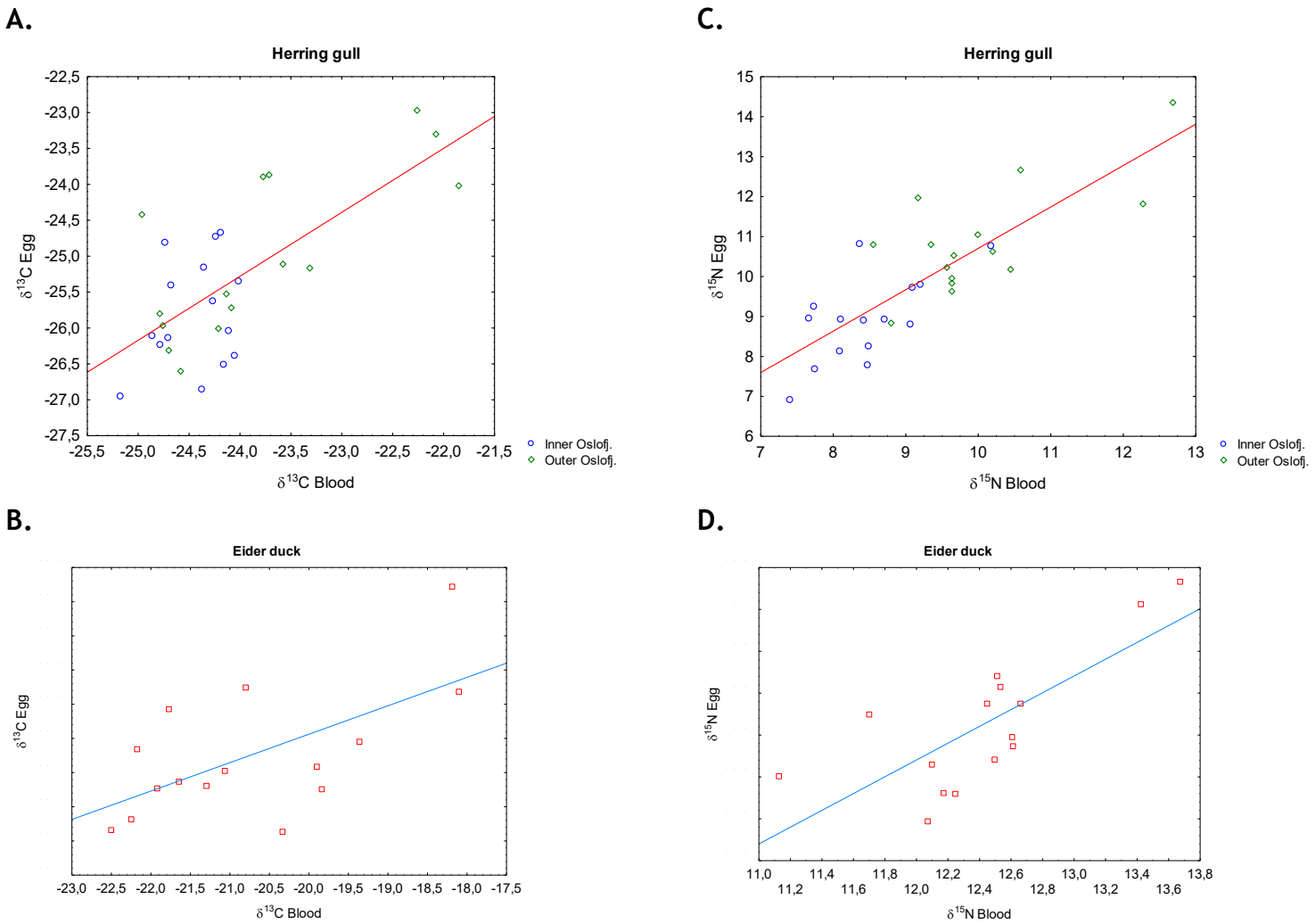


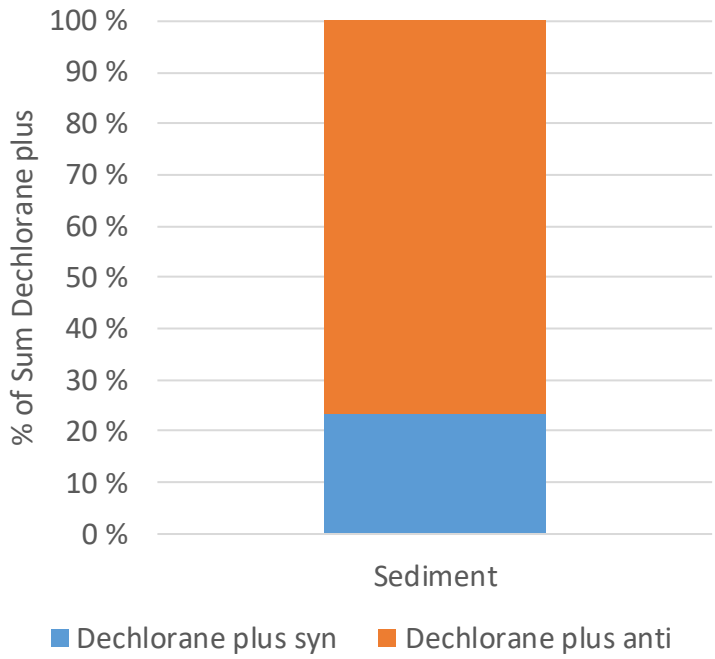
Figure 4. Isotopic ratios of carbon ($\delta^{13}C$; A. and B.) and nitrogen ($\delta^{15}N$, C. and D.) in herring gull (A. and C.) and eider duck (B. and D.) eggs plotted against isotopic ratios inn blood sampled at the same nest.

3.2 Environmental contaminants

3.2.1 Sediment

The sediments of the inner Oslofjord is a potential source of environmental contaminants to sediment dwelling organisms and the contaminants may thus enter the food chain. Several of the target compounds of this study were detected in the sediment sample. Inputs to the fjord via storm water and effluent water from a sewage treatment plant (see Chapters 3.2.6 and 3.2.7) for several of the compounds are also shown.

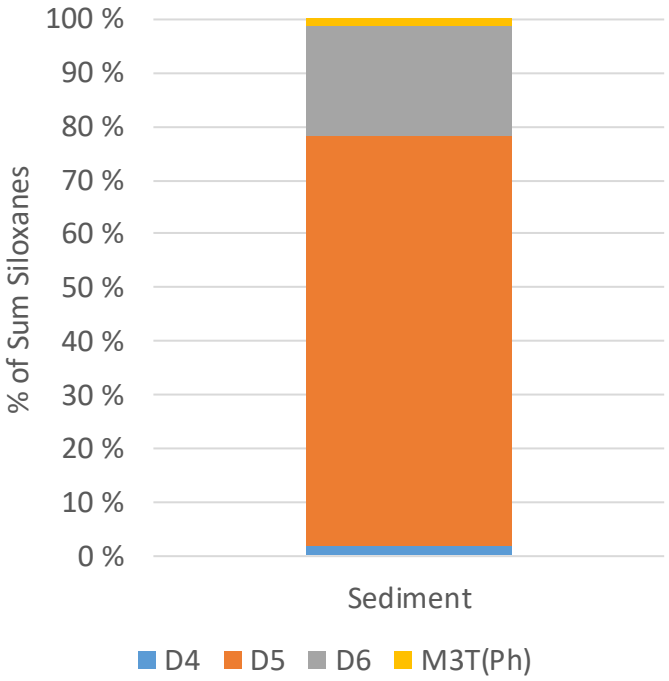
Dechlorane plus was found in the sediment sample (sum of *syn* and *anti* isomers 1.632 ng/g dry wt.; Figure 5). In addition, dechlorane 603 was detected in a concentration of 0.069 ng/g dry wt. (see electronic Appendix).



	Dechlorane plus syn	Dechlorane plus anti
ng/g (dry wt.)	0.383	1.249

Figure 5. Relative contribution (%) of dechlorane plus syn and anti isomers to the sum of dechlorane plus in sediment from the Inner Oslofjord (station Cm21). Concentrations (ng/g dry wt.) are given in the associated table.

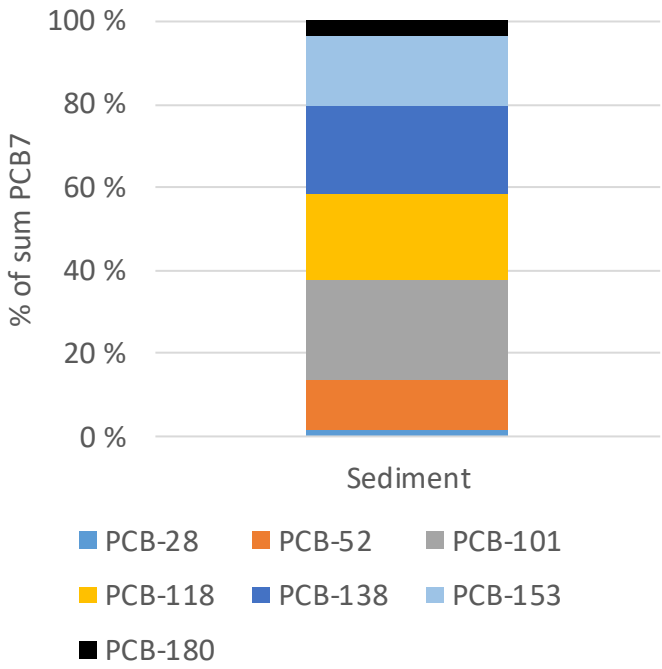
Of the siloxanes, D5 constituted the highest percentage of the sum in sediment (Figure 6).



	D4	D5	D6	M3T(Ph)
ng/g (dry wt.)	2.154	89.69	23.74	1.692

Figure 6. Relative contribution (%) of Siloxanes to the sum of Siloxanes in sediment from the Inner Oslofjord (station Cm21). Concentrations (ng/g dry wt.) are given in the associated table.

The concentration of PCB7 in the sediment appeared a factor 6-7 higher than in 2016 (Ruus et al. 2017; The Norwegian Environment Agency M-812). The relative contribution (%) of PCB-congeners to the sum of PCB7 is presented in Figure 7. PCB-101 -118 -138 and -153 constituted the highest percentages. No polybrominated diphenyl ethers (PBDEs) were detected in sediment.



	PCB-28	PCB-52	PCB-101	PCB-118	PCB-138	PCB-153	PCB-180
ng/g (dry wt.)	1.61	10.30	20.20	17.80	17.90	14.30	3.17

Figure 7. Relative contribution (%) of PCB-congeners to the sum of PCB7 in sediment from the Inner Oslofjord (station Cm21). Concentrations (ng/g dry wt.) are given in the associated table.

For several compounds, environmental quality standards (EQS) for sediment are given through Norwegian law (The Water Regulation/"Vannforskriften"), according to the requirements of the Water Framework Directive. Furthermore, quality standards are given for even more compounds (The Norwegian Environment Agency M-608). For the target compounds of this study of which quality standards exist, the sediment concentrations and quality standards are compared in Table 10. D5, PCB7, Zn, As, Pb, Ni, Hg and PFOS exceeded the quality standards. Regarding inputs to the fjord (apart from the storm water and STP effluent; Chapter 3.2.6), according to Skarbøvik et al. (2017; The Norwegian Environment Agency M-862), River Alna also brought some contaminants to the fjord (see Chapter 3.2.6).

Table 10.
Concentrations of contaminants (mg/kg dry wt) of which Norwegian quality standards (from the Norwegian Environment Agency; M-608) exist in sediment from the inner Oslofjord. Red numbers indicate excess of the quality standard (annual average, AA-EQS).

River basin specific compounds	EQS (mg/kg dry wt.)	Sediment conc. (mg/kg dry wt.)
Bisphenol A	0.0011	<0.080 ***
Decamethylcyclopentasiloxane (D5)	0.044	0.090
Medium chained chloroparafins (MCCPs)	4.6	0.14
Copper (Cu)	84	102
PCB7	0.0041	0.0853
PFOA	0.071	<0.0005
Zinc (Zn)	139	378
TBBPA	0.108	<0.020
Arsenic (As)	18	59
Chromium (Cr)	660	162
EU priority substances		
Cadmium (Cd)	2.5	0.2
Lead (Pb)	150	180
Nickel (Ni)	42	74
Mercury (Hg)	0.52	1.12
Brominated diphenyl ethers *	0.062	<0.002
Hexachlorobenzene	0.017	<0.001
C10-13 chloroalkanes **	0.8	0.39
Pentachlorobenzene	0.4	<0.0006
Nonylphenol (4-)	0.016	<0.005
Oktylphenol (4-*tert*-)	0.0003	<0.6 ***
PFOS	0.00023	0.00041

* Sum of BDE-28, -47, -99, -100, -153 and -154.
** Short chained chloroparaffins (SCCPs)
*** Too high limit of detection to evaluate

3.2.2 Inner Oslofjord Food Web

Several legacy contaminants with well-known biomagnifying properties displayed a positive significant relationship between (log_{10}-)concentrations and trophic position (deduced from the $\delta^{15}N$ isotopic ratio) in the studied Inner Oslofjord marine food web. Of the 32 analysed PCB congeners, 25 showed significant biomagnification, including the seven congeners constituting PCB7 (PCB-153 and 180 shown in Figure 8; TMFs of PCB-28, -52, -101, -118 and -138 were 1.66, 1.86, 2.6, 3.42 and 4.13, respectively). These findings correspond well with the findings from previous years of the "Urban fjord" programme (Ruus et al. 2016; The Norwegian Environment Agency M-601; Ruus et al. 2017; The Norwegian Environment Agency M-812), as well as with previous observations from marine systems (Hallanger et al. 2011; Fisk et al. 2001). Thus, PCBs display expected behaviour in the Inner Oslofjord food web, suggesting again that the studied food web is appropriate for assessing biomagnifying behaviour of contaminants (where PCBs may serve as "benchmark").

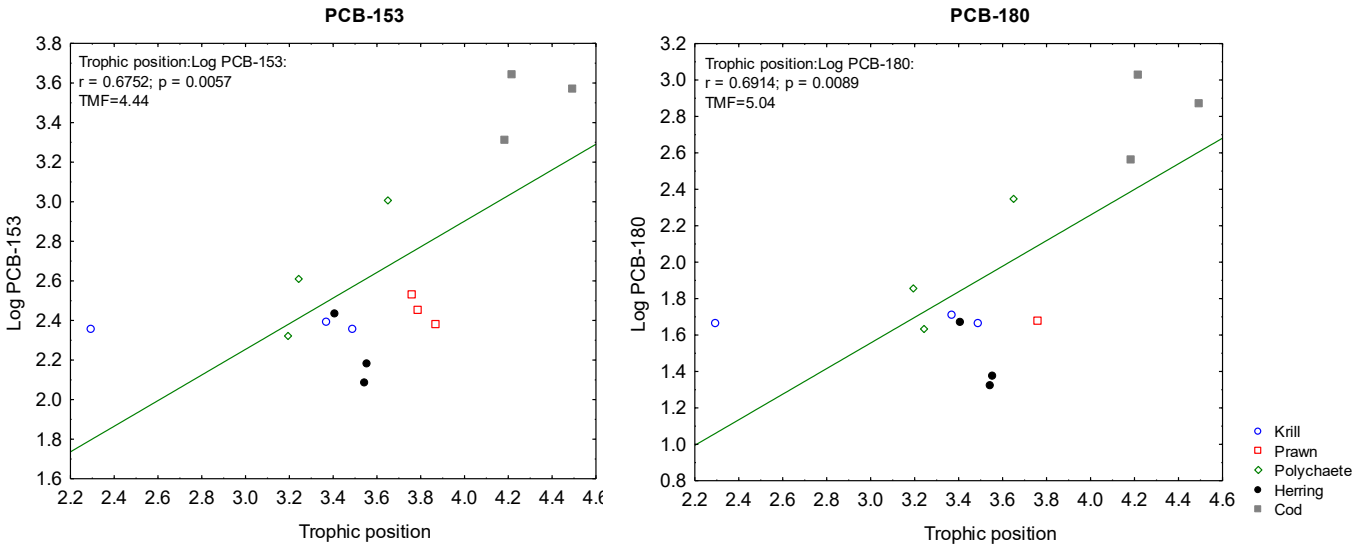


Figure 8. Trophic position against concentrations (ng/g lipid wt.; log-transformed) of PCB-153 and PCB-180 in the studied Inner Oslofjord food web. Note different scales on axes.

Furthermore, if eider duck (here blood) is included in the food web, most of these relationships prevail (some congeners not detected in eider duck blood), and the TMFs are largely unchanged (PCB-28 TMF=1.82; PCB-118 TMF=3.69; PCB-138 TMF=4.35; PCB-153 TMF=4.67; PCB-180 TMF=5.33; PCB-153 and -180 shown in Figure 9).

The relative contribution (%) of PCB-congeners to the sum of PCB7 was similar among the species of the Inner Oslofjord food web, with PCB-153 constituting the highest percentage (this congener was, however, not detected in blue mussel, Figure 10).

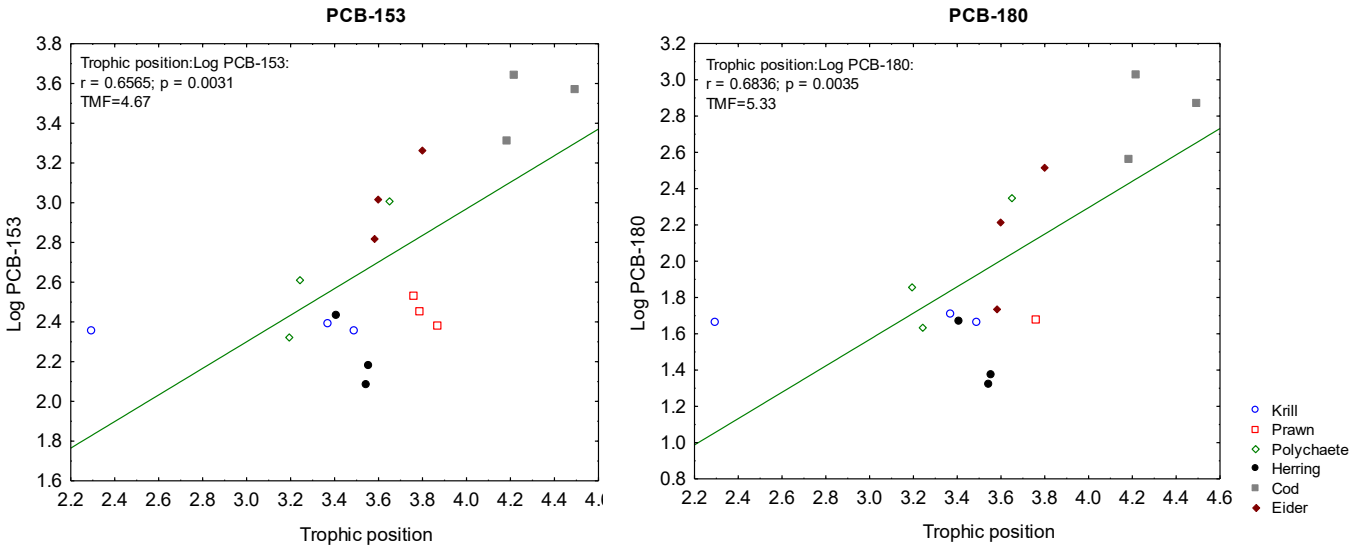
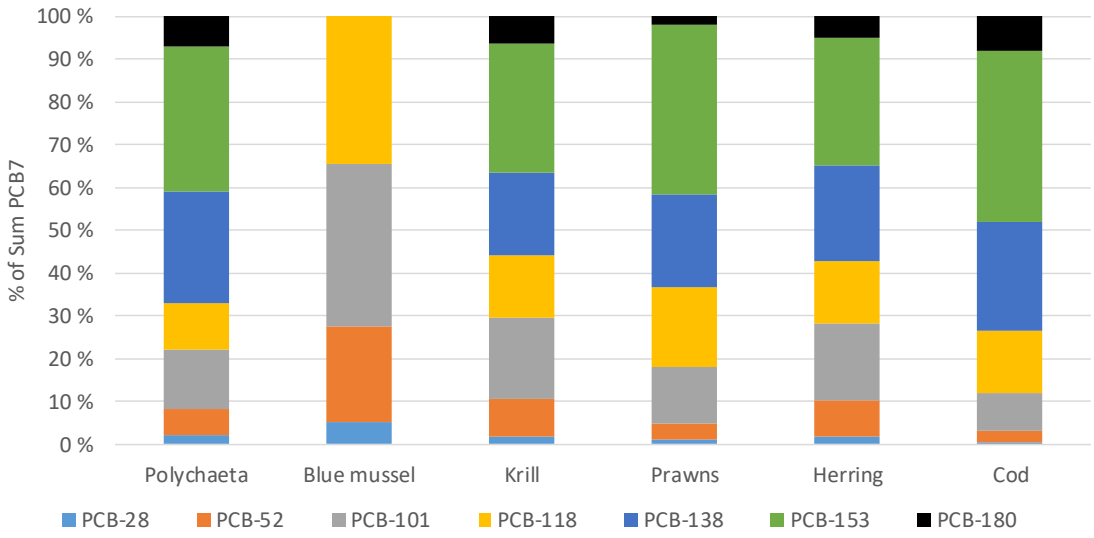


Figure 9. Trophic position against concentrations (ng/g lipid wt.; log-transformed) of PCB-153 and PCB-180 in the studied Inner Oslofjord food web when eider duck (blood) is included. Note different scales on axes.



	PCB-28	PCB-52	PCB-101	PCB-118	PCB-138	PCB-153	PCB-180
Polychaete	0.267	0.799	1.906	1.474	3.377	4.417	0.982
Blue mussel	0.038	0.165	0.280	0.254	n.d.	n.d.	n.d.
Krill	0.076	0.416	0.881	0.683	0.897	1.413	0.289
Prawn	0.030	0.112	0.396	0.547	0.643	1.183	0.063
Herring	0.438	2.250	4.663	3.893	5.900	7.917	1.322
Cod	11.5	74.5	254.4	420.4	729.9	1124.4	227.2

Figure 10. Relative contribution (%) of PCB-congeners to the sum of PCB7 in the species of the Inner Oslofjord food web. Concentrations (ng/g wet wt.; mean) are given in the associated table.

Among the brominated compounds, tribromanisole (TBA) showed statistically significant trophic dilution (TMF=0.21), as previously observed in (Ruus et al. 2016; The Norwegian Environment Agency M-601; Ruus et al. 2017; The Norwegian Environment Agency M-812). However, this compound was not detected in krill and prawn. The following polybrominated diphenyl ethers showed statistically significant biomagnification: BDE-47 (TMF=3.71; Figure 11), BDE-49 (TMF=3.44) and BDE-100 (TMF=3.35; Figure 11). This corresponds to previous observations in the "Urban fjord" programme (Ruus et al. 2016; The Norwegian Environment Agency M-601 Ruus et al. 2017; The Norwegian Environment Agency M-812). Furthermore, biomagnification of PBDEs has previously been shown in marine systems (e.g. Hallanger et al. 2011).

Again, if eider duck (here blood) is included in the food web, there is still a significant TMF for BDE-100, although somewhat lower (BDE-100 TMF=2.90; some congeners not detected in eider duck blood, see Table 14).

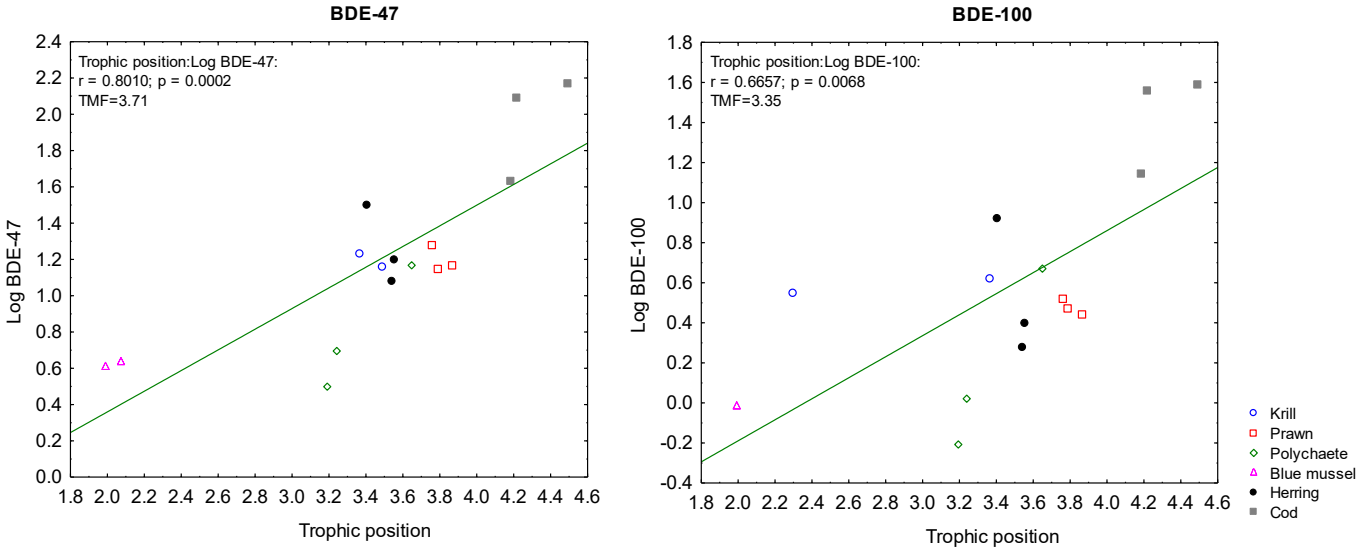
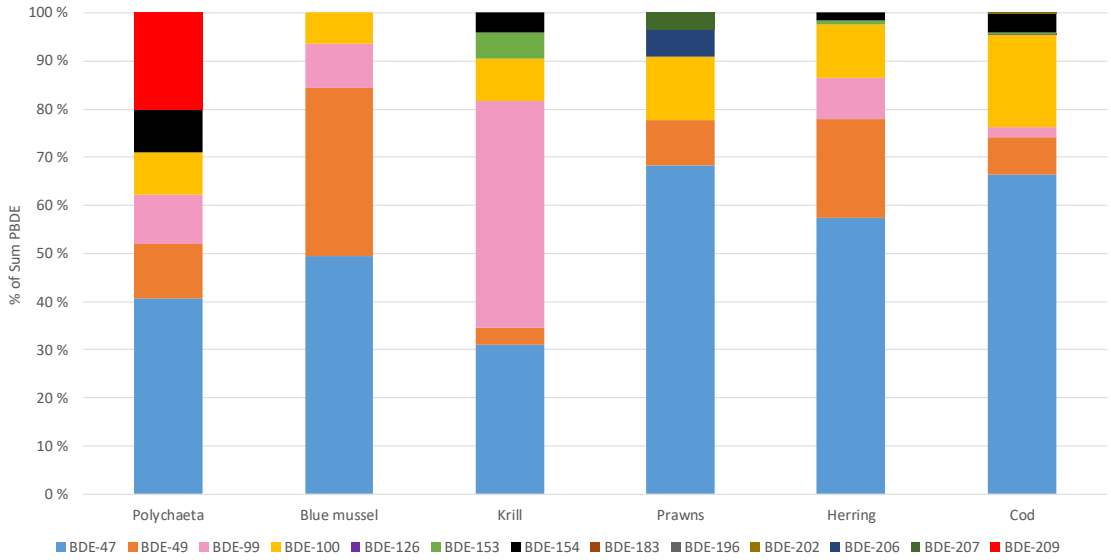


Figure 11. Trophic position against concentrations (ng/g lipid wt.; log-transformed) of BDE-47 and -100 in the studied Inner Oslofjord food web. Note different scales on axes.

The relative contribution (%) of BDE-congeners to the sum of PBDEs appeared somewhat different among the species of the Inner Oslofjord food web (Figure 12). BDE-47 constituted the highest percentage in most species (Figure 12). BDE-99 was detected in all species, except prawn and constituted 37% in krill (Figure 12).



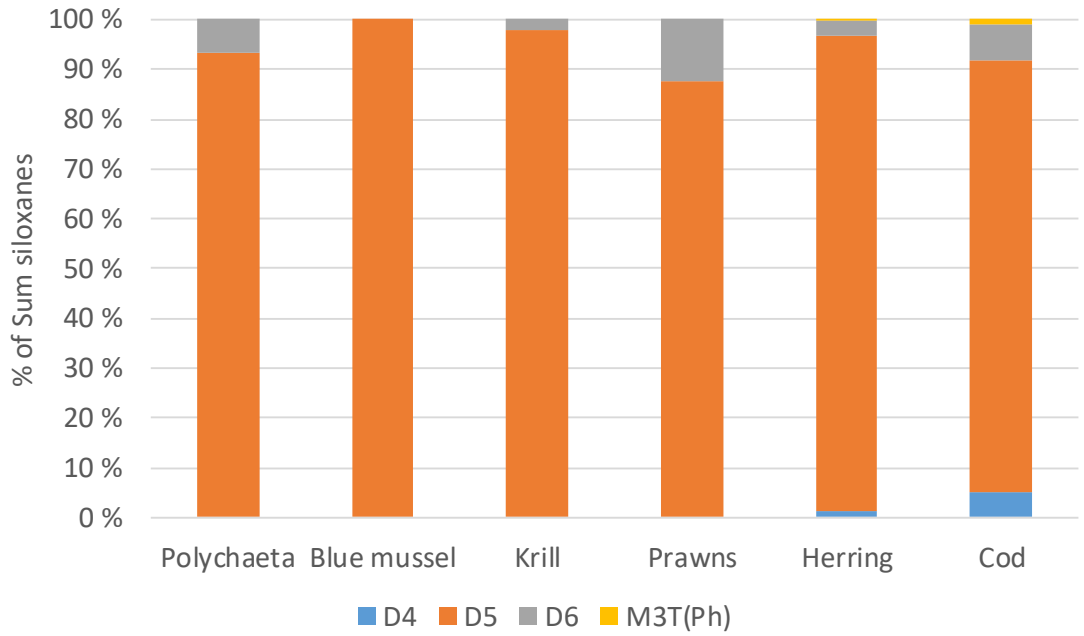
	Polychaete	Blue mussel	Krill	Prawn	Herring	Cod
BDE-47	0.062	0.024	0.062	0.065	0.854	42.00
BDE-49	0.017	0.002	0.006	0.009	0.305	6.11
BDE-99	0.024	0.005	0.081	n.d.	0.118	0.923
BDE-100	0.016	0.003	0.014	0.012	0.174	11.34
BDE-126	n.d.	n.d.	n.d.	n.d.	n.d.	0.042
BDE-153	n.d.	n.d.	0.009	n.d.	0.012	0.134
BDE-154	0.013	n.d.	0.007	n.d.	0.020	1.94
BDE-183	n.d.	n.d.	n.d.	n.d.	n.d.	0.012
BDE-196	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
BDE-202	n.d.	n.d.	n.d.	n.d.	n.d.	0.105
BDE-206	n.d.	n.d.	n.d.	0.008	n.d.	0.009
BDE-207	n.d.	n.d.	n.d.	0.005	n.d.	n.d.
BDE-209	0.106	n.d.	n.d.	n.d.	n.d.	n.d.

Figure 12. Relative contribution (%) of selected BDE-congeners (see Table 5) to the sum of those PBDEs in the species of the Inner Oslofjord food web (previous page). Concentrations (ng/g wet wt.; mean; non-detected components were assigned a value of zero) are given in the associated table. Components that were not detected in any replicate samples of a species are noted n.d.

The concentrations of siloxanes (D4, D5, D6 and M3T(Ph)) displayed no significant relationship with trophic position. Mor precisely, concentrations of D4 did show a positive relationship with trophic position (TMF=10.33), but was only detected in herring and cod. A biomagnification factor (BMF) calculated from herring to cod was 10.4. Calculating BMF from

herring to cod for D5 and D6 gives 2.0 and 6.9, respectively. There have previously been some divergences in reports of the biomagnifying properties of siloxanes in different systems (e.g. Borgå et al. 2012 and references therein). By compiling data from different surveys from the period 2010-2016, Fjeld et al. (2017; The Norwegian Environment Agency M-807) demonstrated biomagnification of D5 in the lakes Mjøsa and Randsfjorden with a common TMF of 2.34, and biomagnification of D6 with a common TMF of 1.92. D5 appeared in the highest concentrations (Fjeld et al. 2017; The Norwegian Environment Agency M-807). On the other hand, Powel et al (2018) found no biomagnification of D4, D5 and D6 across demersal and pelagic food webs in the Oslofjord.

Of the siloxanes analysed in the present study, D5 also appeared in the highest concentrations in all species of the food web (Figure 13).



	D4	D5	D6	M3T(Ph)
Blue mussel	n.d.	16.73	n.d.	n.d.
Prawn	n.d.	7.51	1.27	n.d.
Krill	n.d.	182.94	4.25	n.d.
Polychaete	n.d.	107.89	7.00	n.d.
Herring	2.35	162.33	5.18	0.40
Cod	175.85	2518.27	274.09	39.02

Figure 13. Relative contribution (%) of D4, D5, D6 and M3T(Ph) to the sum of siloxanes in the species of the Inner Oslofjord food web. Concentrations (ng/g wet wt.; mean) are given in the associated table. Components that were not detected in any replicate samples of a species are noted n.d.

Mercury displayed statistically significant biomagnification (TMF=3.80; Figure 14) in the Inner Oslofjord food web, as previously observed in the "Urban fjord" programme (Ruus et al. 2016;

Ruus et al. 2017; The Norwegian Environment Agency M-601 and M-812). The biomagnifying properties of Hg (particularly methylmercury, MeHg) are well known (e.g. Jaeger et al. 2009; Ruus et al. 2015). Again, if eider duck (here blood) is included in the food web, there is still a significant, and similar, TMF for Hg (Hg TMF=3.99). It should be noted that the proportion of total Hg that is MeHg in the different organism is not known and could differ.

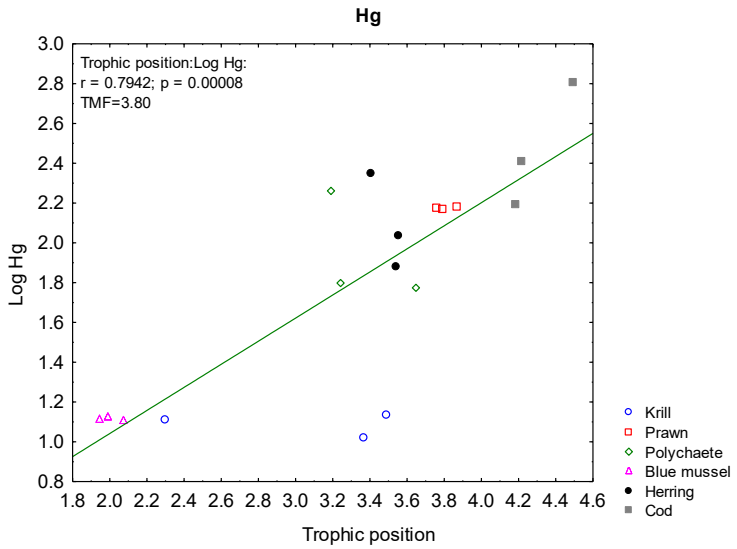


Figure 14. Trophic position against concentrations (ng/g wet wt.; log-transformed) of mercury (Hg) in the studied Inner Oslofjord food web.

Furthermore, also the elements As (TMF=2.29; Figure 15) and Ag (TMF=10.26; Figure 16) again displayed statistically significant positive relationships between (log) concentrations and trophic position (as in 2015 and 2016). It should be mentioned again that in this programme, total As was measured (not only inorganic As), and most of the arsenic found in fish, and marine animals in general, is present as arsenical arsenobetaine, which is regarded as non-toxic (Amlund, 2005 and references therein). Arsenobetaine is rapidly absorbed over the gastrointestinal tract (Amlund, 2005 and references therein). There is little evidence of biomagnification of Ag in marine systems, and according to a review by Fisher and Wang (1998), trophic transfer of Ag has been shown to be insignificant in several aquatic animals but more important in others. Maneekarn et al. (2014) studied bioaccumulation and biomagnification of nano Ag^0 particles (AgNPs) in a model food chain containing green algae (*Chlorella sp.*), water flea (*Moina macroscopa*), blood worm (*Chironomus spp.*) and silver barb *(Barbonys gonionotus*). They found that food chain transfer of AgNPs occurred only from *Chlorella sp.* to *M. macroscopa*. Both As and Ag were detected in sediment from the Inner Oslofjord, as well as in storm water (Ag and Hg only in the particle phase) entering the fjord (see electronic Appendix), while Ag (the only element analysed) was not detected in effluent water from Bekkelaget STP (<0.007 ng/ml). Silver nanoparticles (AgNP) are used in several consumer products (*inter alia* textiles) for their antimicrobial properties, however, their possible influence on the observed results is unknown. Wang et al (2014) showed that the marine polychaete *Nereis virens* accumulated Ag in the forms of AgNP-citrate, AgNP-polyvinylpyrrolidone and as a salt ($AgNO_3$).

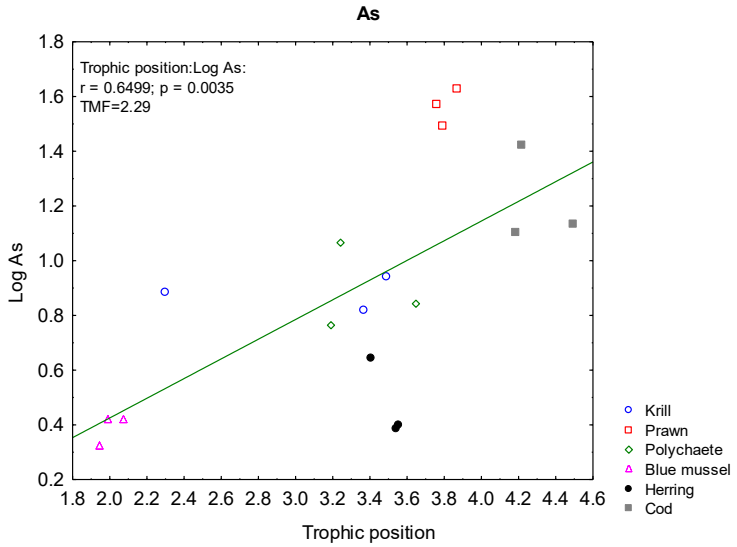


Figure 15. Trophic position against concentrations (µg/g wet wt.; log-transformed) of arsenic (As) in the studied Inner Oslofjord food web.

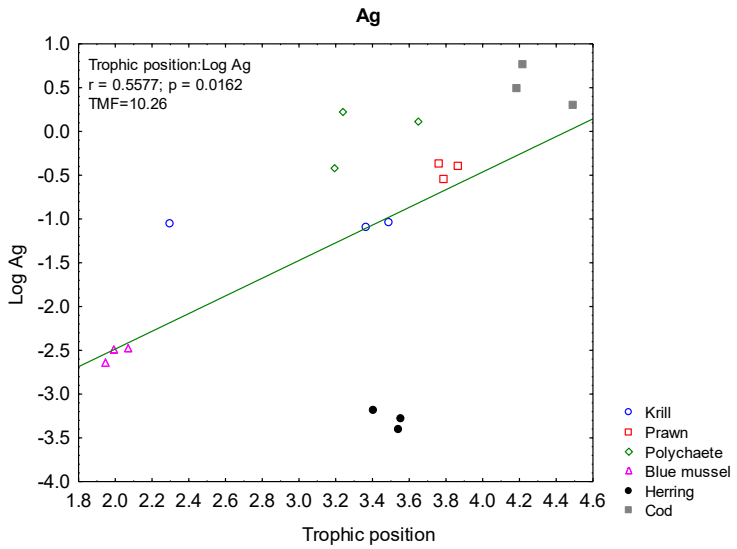


Figure 16. Trophic position against concentrations (µg/g wet wt.; log-transformed) of silver (Ag) in the studied Inner Oslofjord food web.

Regarding PFAS compounds, there were many non-detects for most compounds. PFOSA, however, was detected in all samples but one, and displayed a significant positive relationship between (log) concentration and trophic position (TMF= 2.71; Figure 17). Again, if eider duck (here blood) is included in the food web, there is still a significant, and similar, TMF for PFOSA (PFOSA TMF=2.64). Previously, PFOS also showed significant biomagnification in the Inner Oslofjord marine food web (Ruus et al. 2017; The Norwegian Environment Agency M-812). In 2017, however, PFOS was not detected in blue mussel and krill. Biomagnification of PFOSA and PFOS has previously been shown in marine food webs (e.g. Kelly et al. 2009; Houde et al. 2011), However, Franklin (2015), points to the great variability in Field derived biomagnification estimates of PFAS compounds.

PFOSA constituted the highest percentage (of sum PFAS) in blue mussel, krill, herring and cod (Figure 18), as previously observed (Ruus et al. 2017; The Norwegian Environment Agency M-812). PFOS was also an important constituent in herring and cod (constituting ~20-30% of sum PFAS; Figure 18).

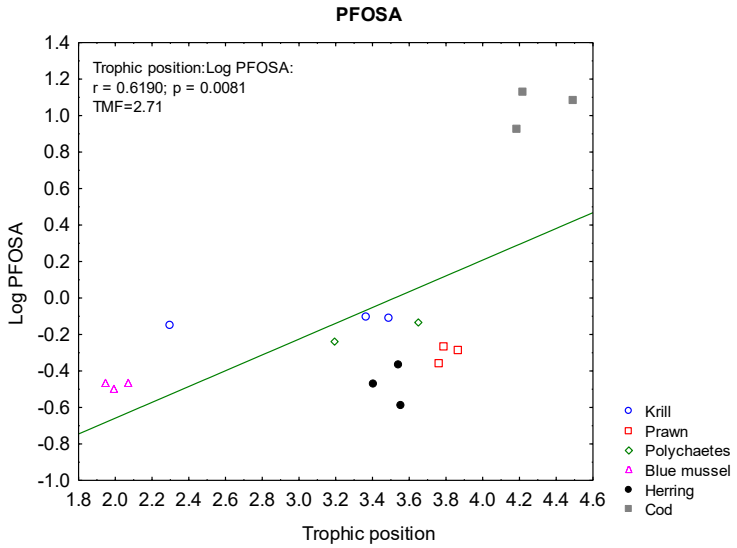
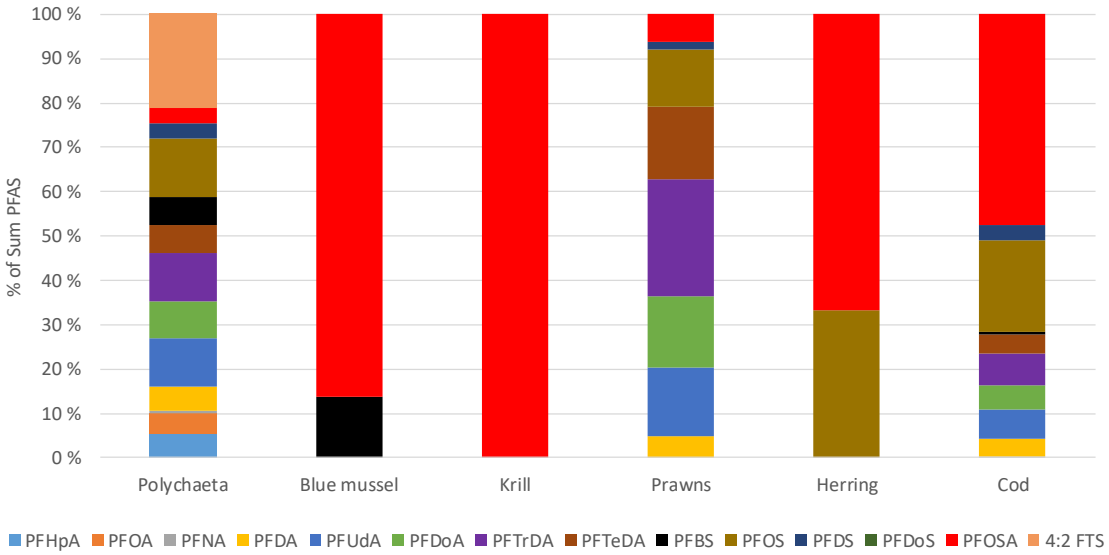


Figure 17. Trophic position against concentration (ng/g wet wt.; log-transformed) of PFOSA in the studied Inner Oslofjord food web.



	Polychaete	Blue mussel	Krill	Prawn	Herring	Cod
PFHpA	0.45	n.d.	n.d.	n.d.	n.d.	n.d.
PFOA	0.71	n.d.	n.d.	n.d.	n.d.	n.d.
PFNA	0.19	n.d.	n.d.	n.d.	n.d.	0.09
PFDA	1.59	n.d.	n.d.	0.38	n.d.	1.02
PFUdA	2.82	n.d.	n.d.	1.29	n.d.	1.46
PFDoA	1.53	n.d.	n.d.	1.32	n.d.	1.21
PFTrDA	1.68	n.d.	n.d.	2.18	n.d.	1.55
PFTeDA	0.90	n.d.	n.d.	1.36	n.d.	0.91
PFBS	0.53	0.08	n.d.	n.d.	n.d.	0.17
PFOS	2.55	n.d.	n.d.	1.06	0.17	4.24
PFDS	0.30	n.d.	n.d.	0.17	n.d.	0.70
PFOSA	0.44	0.34	0.76	0.50	0.34	11.41
4:2 FTS	1.95	n.d.	n.d.	n.d.	n.d.	n.d.

Figure 18. Relative contribution (%) of PFAS compounds-to the sum of (detected) PFASs in the species of the Inner Oslofjord food web (previous page). Concentrations (ng/g wet wt.; mean; non-detected components were assigned a value of zero) of detected components are given in the associated table. Components that were not detected in any replicate samples of a species are noted n.d.

UV chemicals were only detected in samples of herring in the Inner Oslofjord marine food web (see electronic Appendix).

Behentrimonium (ATAC-C20 and -C22) was detected in all biota samples of the Inner Oslofjord food web (see electronic Appendix). Furthermore, it showed significant bioaccumulation on a wet weight basis (Figure 19 A), with high concentrations in cod liver. In a recent Nordic survey (Nordic cooperation on screening; Kaj et al. 2014), these compounds were also found in fish liver and muscle, as well as in effluents and sludges from STPs and in sediments. As described in Chapter 3.2.6, ATAC-C20 and -C22 was also detected in storm water, with the highest concentrations in the particulate phase. According to Kaj et al. (2014), data on K_{OW} and BCF is limited and lacking for ATAC-C20 and -C22. High concentrations in lipid rich cod liver and affinity for particles might suggest that it is most appropriate to express the concentrations on a lipid weight basis. If this is the case, it would render the TMF not significant (Figure 19).

As in 2016 (Ruus et al. 2017; The Norwegian Environment Agency M-812), no phenolic compounds were detected in more than three samples of the Inner Oslofjord food web. The limit of detection was high for some of the compounds, due to blank issues.

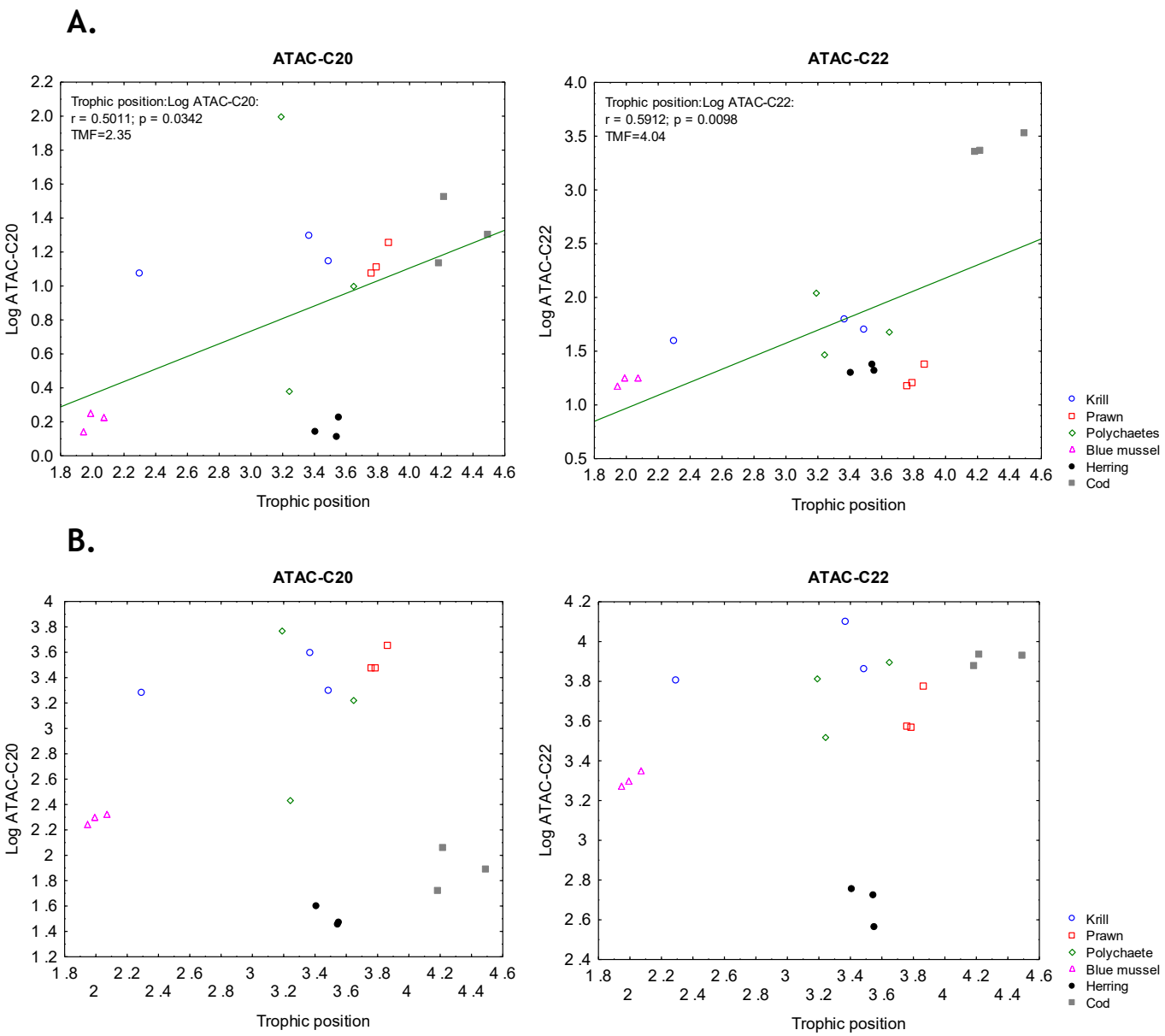


Figure 19. Trophic position against concentrations (A: ng/g wet wt. and B: ng/g lipid wt.; log-transformed) of ATAC-C20 (left) and ATAC-C22 (right) in the studied Inner Oslofjord food web. Note different scales on axes.

3.2.3 Cod

Environmental contaminants were analysed in 15 cod individuals. Pooled samples of cod, 3 samples constituted of 5 individuals each sorted by their length, were constructed mathematically to obtain 3 samples of each species, for evaluation of biomagnifying behaviour in the Inner Oslofjord food web.

Biological effect parameters were also measured in cod, and these are dealt with in Chapter 3.5.

Concentrations (mean and range) for all compounds and elements analysed in cod are presented Table 11, as well as in Appendix.

Table 11.
Lipid content (%) and concentrations of the different analytes (see Table 5) in cod liver from the Inner Oslofjord. Concentrations are ng/g wet wt., except for concentrations of Ni, Cu, Ag, Cd, Pb, Cr, Fe, Zn, As and Sb, which are expressed as µg/g wet wt. Arithmetic mean and range (minimum and maximum) are presented (n=15). In calculations of mean, non-detected components were assigned a value of zero (0). No phenolic compounds were detected, hence not included in the table.

Analyte	Mean	Min.	Max.	Detected in no. of samples
Lipid content (%), liver	37.4	12.7	75.1	15
PeCB	0.8	0.3	2.2	15
HCB	8.6	2.0	28.4	15
MB1	<10	<10	<10	0
Dechlorane	**Mean**	**Min.**	**Max.**	**Detected in no. of samples**
Dechlorane 602	0.680	0.125	3.358	15
Dechlorane 603	0.162	<0.050	0.690	13
Dechlorane 604	n.d.	<0.020	<0.050	0
Dechlorane plus syn	0.048	<0.250	0.296	4
Dechlorane plus anti	0.109	<0.500	0.670	4
PCBs (PCB7)	**Mean**	**Min.**	**Max.**	**Detected in no. of samples**
PCB-28	11.5	2.1	39.7	15
PCB-52	74.5	16.1	253.0	15
PCB-101	254.4	77.2	636.0	15
PCB-118	420.4	124.0	1400.0	15
PCB-138	729.9	216.0	2120.0	15
PCB-153	1124.4	332.0	3280.0	15
PCB-180	227.2	64.8	655.0	15
Sum-PCB_7	2842.2	850.4	8259.2	15
Brominated comp.	**Mean**	**Min.**	**Max.**	**Detected in no. of samples**
BDE-47	42.001	8.140	98.400	14
BDE-49	6.109	0.552	23.500	14
BDE-99	0.923	0.197	2.510	14
BDE-100	11.334	1.910	30.100	14
BDE-126	0.042	<0.005	0.172	8
BDE-153	0.134	<0.015	0.295	13
BDE-154	1.945	0.843	6.030	14

BDE-183	0.012	<0.009	0.059	4
BDE-196	n.d.	<0.02	<0.05	0
BDE-202	0.105	<0.048	0.390	10
BDE-206	0.008	<0.029	0.076	2
BDE-207	n.d.	<0.02	<0.05	0
BDE-209	n.d.	<0.581	<1.45	0
ATE (TBP-AE)	n.d.	<0.038	<0.094	0
a-TBECH	n.d.	<0.093	<0.233	0
b-TBECH	n.d.	<0.067	<0.167	0
g/d-TBECH	0.126	<0.024	0.383	13
BATE	n.d.	<0.012	<0.031	0
PBT	n.d.	<0.08	<0.2	0
PBEB	n.d.	<0.083	<0.208	0
HBB	0.011	<0.047	0.158	2
DPTE	n.d.	<0.013	<0.032	0
EHTBB	n.d.	<0.029	<0.374	0
BTBPE	n.d.	<0.048	<0.121	0
TBPH (BEH /TBP)	n.d.	<0.093	<0.232	0
DBDPE	6.264	<2.64	56.600	4
Chloroparaffins	**Mean**	**Min.**	**Max.**	**Detected in no. of samples**
SCCP	738.5	46.0	2170.0	15
MCCP	216.0	51.0	1050.0	15
Siloxanes	**Mean**	**Min.**	**Max.**	**Detected in no. of samples**
D4	175.8	29.6	1334.6	14
D5	2518.3	550.5	8558.1	15
D6	274.1	45.4	2067.6	15
M3T(Ph)	39.0	4.9	238.0	15
Metals	**Mean**	**Min.**	**Max.**	**Detected in no. of samples**
Cr	0.318	0.116	0.808	15
Fe	30.129	13.007	58.512	15
Ni	0.244	0.112	0.555	15
Cu	4.077	1.342	7.724	15

Zn	18.526	7.652	29.433	15
As	17.636	3.204	56.147	15
Ag	3.640	0.349	10.595	15
Cd	0.054	0.020	0.193	15
Sb	0.004	0.000	0.009	15
Pb	0.063	0.005	0.226	15
Hg	350.901	45.478	2297.881	15
PFAS compounds	**Mean**	**Min.**	**Max.**	**Detected in no. of samples**
PFPA	n.d.	<0.5	<0.5	0
PFHxA	n.d.	<0.5	<0.5	0
PFHpA	n.d.	<0.5	<0.5	0
PFOA	n.d.	<0.5	<0.5	0
PFNA	0.088	<0.5	0.815	2
PFDA	1.016	<0.5	2.608	11
PFUdA	1.458	<0.4	3.661	13
PFDoA	1.214	<0.4	2.678	13
PFTrDA	1.554	<0.4	3.638	14
PFTeDA	0.915	0.22	1.999	15
PFBS	0.167	<0.2	2.096	3
PFPS	n.d.	<0.2	<0.2	0
PFHxS	n.d.	<0.1	<0.1	0
PFHpS	n.d.	<0.2	<0.2	0
PFOS	4.242	1.036	11.668	15
8Cl-PFOS	n.d.	<0.2	<0.2	0
PFNS	n.d.	<0.2	<0.2	0
PFDS	0.650	<0.2	1.145	14
PFDoS	n.d.	<0.2	<0.2	0
PFOSA	11.410	2.331	35.665	15
me-FOSA	n.d.	<0.3	<0.3	0
et-FOSA	n.d.	<0.3	<0.3	0
me-FOSE	n.d.	<5	<5	0
et-FOSE	n.d.	<5	<5	0

4:2 FTS	n.d.	<0.3	<0.3	0
6:2 FTS	n.d.	<0.3	<0.3	0
8:2 FTS	n.d.	<0.3	<0.3	0
me-FOSAA	n.d.	<0.3	<0.3	0
Et-FOSAA	n.d.	<0.3	<0.3	0
UV-chemicals	**Mean**	**Min.**	**Max.**	**Detected in no. of samples**
BP3	n.d.	<1	<7	0
EHMC	n.d.	<7	<50	0
OC	n.d.	<10	<60	0
Behentrimonium	**Mean**	**Min.**	**Max.**	**Detected in no. of samples**
ATAC-C20	22.8	8.2	47.0	14
ATAC-C22	2635.7	1300.0	4700.0	14

Of the substances analysed for which (biota) quality standards exist (for EU priority substances or Norwegian river basin specific substances; The Norwegian Environment Agency; M-608), mean concentrations of Hg, PBDEs, PCB7 and MCCPs exceeded the quality standards. Note that the biota quality standards relate to (whole) fish, but that an alternative biota taxon, or another matrix, may be monitored instead, as long as the quality standard applied provides an equivalent level of protection.

No individual D5 concentration exceeded the quality standard of 15217 ng/g (The Norwegian Environment Agency; M-608). This was also the result for cod liver collected in the Inner Oslofjord in 2017, in a parallel study (Green et al. 2018, The Norwegian Environment Agency M-1120). In that study, the median D5 concentration in cod liver was 1117.6 ng/g wet wt. In the present study, the mean D5 concentration in the cod liver on a lipid weight basis (6677 ng/g ± 3985 standard deviation) was higher than that in trout from Lake Mjøsa in 2016 (1312 ± 585; Fjeld et al. 2017). Furthermore, the mean D5 concentration was apparently (not statistically tested) approximately 20 % higher in 2017, compared to 2016 (Ruus et al. 2017; The Norwegian Environment Agency M-812). M3T(Ph) was found in cod liver, however, not in equally high concentrations as D4, D5 and D6 (Table 11). The mean concentration (39.0 ng/g wet wt.) appeared higher than that previously observed in trout from lake Mjøsa (1.2 ng/g wet. wt.; van Bavel et al. 2016; The Norwegian Environment Agency M-596), but note that basis for comparison is only on a wet weight basis.

In previous studies of cod from the Inner Oslofjord (e.g. Powell et al. 2018; Schlabach et al. 2007), D5 was, as in the present study, detected as the dominating siloxane compound.

Co-variation between fish length and Hg-concentrations is well known (e.g. Eikenberry et al. 2015; Green and Knutzen, 2003; Jones et al. 2013; Julshamn et al. 2013; Sackett et al. 2013), and previously a positive relationship was found between Hg concentrations in cod and the length of cod (Ruus et al. 2016; Ruus et al. 2015). Jones et al. (2013) have also argued that detecting the influence of changes in Hg exposure will depend on how well fish biotmetrics (length, age and growth rates) are considered. In 2017, there was no statistically significant

relationship (p=0.056) between Hg in cod and the length of cod (Figure 20), and one individual cod displayed markedly higher Hg-concentration (2298 ng/g wet wt.) than the other specimens.

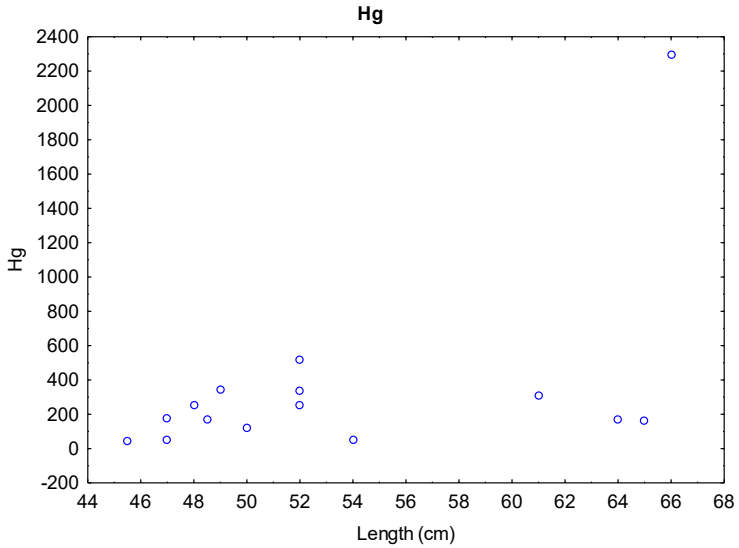


Figure 20. Concentrations (ng/g wet wt.) of mercury (Hg) in muscle of cod against length (cm) in cod from the Inner Oslofjord.

As in 2016 (Ruus et al. 2017; The Norwegian Environment Agency M-812), the flame retardant decabromodiphenyl ethane (DBDPE) was found in elevated concentrations in some individuals (See appendix). DBDPE is a flame retardant substance for various applications, such as plastic and rubber materials, electrical and electronic equipment, adhesives and sealants (an alternative to deca-BDE).

UV chemicals were not detected in any cod liver samples.

Some dechlorane compounds (note dechlorane 602 and 603) were detected in cod liver (Table 11).

Behentrimonium was found in all analysed samples of cod liver (see Appendix). ATAC-C22 was found in concentrations a factor of ~50-500 higher than ATAC-20. As mentioned, in a recent Nordic survey (Nordic cooperation on screening; Kaj et al. 2014), these compounds were also found in fish liver and muscle. In that study, 3 liver samples of cod collected in vicinity of the VEAS STP discharge pointwas analysed. The concentrations of ATAC-C20 were 11, 23 and 160 ng/g, while the concentrations of ATAC-C22 were 250, 460 and 5400 ng/g, in these samples respectively. Mean concentrations of ATAC-C20 and ATAC-C22 in the present study were 22.8 and 2635.7 ng/g, respectively (Table 11). As described in Chapter 2.2.1, behentrimonium chloride or methosulphate, containing ATAC-C20 and ATAC-C22 are used in personal care products, especially in hair care products.

Phenolic compounds were not detected in any cod liver samples (n=15). The limit of detection was high for some of the compounds, due to blank issues. Note that the phenolic compounds in Table 11 were also analysed in bile of cod (n=7). Also in these samples, no

concentrations of phenolic compounds were detected, except for 4,4-bisphenol F in one sample (12 ng/g; see Appendix).

3.2.4 Herring gull

Inner Oslofjord

Both blood and egg were sampled from herring gull. Adult female blood and egg was sampled from the same nest (i.e. mother and future offspring). Herring gulls were also collected from the Outer Oslofjord. Results are presented and compared under a separate heading ("*Outer Oslofjord*"), below.

Concentrations (mean and range; wet wt. basis) for all compounds and elements analysed in herring gull (blood and egg) are presented in Table 12. The number of samples in which the substance was detected is also shown in Table 12.

Table 12.
Lipid content (%) and concentrations of the different analytes in herring gull blood and egg from the Inner Oslofjord. Concentrations are ng/g wet wt., except for concentrations of Ni, Cu, Ag, Cd, Pb, Cr, Fe, Zn, As and Sb, which are expressed as µg/g wet wt. Arithmetic mean and range (minimum and maximum) are presented (n=15). In calculations of mean, non-detected components were assigned a value of zero (0). Det. no. is the number of samples in which the substance was detected (blood/egg).

Analyte	Blood Mean	Blood Min.	Blood Max.	Egg Mean	Egg Min.	Egg Max.	Det. no.
Lipid content (%)	2.28	0.60	5.30	8.01	5.92	9.97	15/15
PeCB	n.d.	<0.103	<0.118	0.179	0.041	0.392	15/15
HCB	0.415	<0.131	1.270	3.655	0.720	9.350	15/15
MB1	<2	<2	<2				0
Dechlorane	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
Dechlorane 602	0.003	<0.020	0.041	n.d.	<0.005	<0.005	14/0
Dechlorane 603	n.d.	<0.010	<0.029	0.015	<0.005	0.041	0/12
Dechlorane 604	n.d.	<0.050	<0.050	n.d.	<0.010	<0.010	0/0
Dechlorane plus syn	0.009	<0.125	0.141	0.108	<0.025	0.654	1/14
Dechlorane plus anti	n.d.	<0.250	<0.286	0.337	<0.050	1.943	0/14
PCBs (PCB7)	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
PCB-28	0.057	<0.059	0.239	1.047	0.127	5.750	7/15
PCB-52	0.069	<0.064	0.365	1.707	0.077	8.830	7/15
PCB-101	0.052	<0.523	0.781	2.171	0.114	10.200	1/15
PCB-118	1.821	<0.506	5.260	26.004	1.740	103.000	14/15

PCB-138	3.653	<1.060	14.500	46.559	5.690	142.000	14/15
PCB-153	5.633	<1.860	23.000	73.309	9.530	204.000	14/15
PCB-180	1.369	0.498	4.430	19.840	3.310	43.900	15/15
Sum-PCB_7	12.650	0.710	48.450	170.640	21.130	511.470	15/15
Brominated comp.	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
BDE-47	0.330	0.086	1.000	6.143	0.355	34.300	15/15
BDE-49	0.006	<0.007	0.045	0.072	<0.004	0.255	4/13
BDE-99	0.256	0.043	1.810	6.752	0.330	53.900	15/15
BDE-100	0.096	0.023	0.314	1.710	0.121	9.800	15/15
BDE-126	n.d.	<0.005	<0.021	0.006	<0.002	0.024	0/9
BDE-153	0.062	<0.011	0.348	1.841	0.104	14.000	11/15
BDE-154	0.030	<0.009	0.130	0.618	0.058	2.820	9/15
BDE-183	0.015	<0.012	0.093	0.516	0.035	4.620	8/15
BDE-196	0.007	<0.025	0.098	0.306	<0.005	1.990	1/13
BDE-202	0.002	<0.024	0.035	0.076	<0.012	0.296	1/12
BDE-206	0.021	<0.036	0.094	0.173	<0.007	1.000	5/10
BDE-207	0.090	<0.025	0.568	1.950	0.072	14.000	10/15
BDE-209	0.474	<0.726	3.620	7.309	<0.145	49.700	4/14
ATE (TBP-AE)	n.d.	<0.038	<0.282	n.d.	<0.047	<0.047	0/0
a-TBECH	n.d.	<0.093	<0.235	n.d.	<0.116	<0.116	0/0
b-TBECH	n.d.	<0.067	<0.173	n.d.	<0.083	<0.083	0/0
g/d-TBECH	n.d.	<0.024	<0.086	n.d.	<0.03	<0.03	0/0
BATE	n.d.	<0.012	<0.042	n.d.	<0.015	<0.015	0/0
PBT	n.d.	<0.08	<2.28	n.d.	<0.146	<0.146	0/0
PBEB	n.d.	<0.083	<2.79	n.d.	<0.141	<0.141	0/0
HBB	0.003	<0.047	<0.066	n.d.	<0.058	<0.058	1/0
DPTE	n.d.	<0.013	<0.036	n.d.	<0.016	<0.016	0/0
EHTBB	n.d.	<0.029	<0.137	n.d.	<0.036	<0.036	0/0
BTBPE	n.d.	<0.048	<0.069	n.d.	<0.06	<0.06	0/0
TBPH (BEH /TBP)	n.d.	<0.093	<0.913	n.d.	<0.201	<0.201	0/0
DBDPE	0.921	<2.64	9.850	n.d.	<3.3	<3.3	13/0

Chloroparaffins	Blood Mean	Blood Min.	Blood Max.	Egg Mean	Egg Min.	Egg Max.	Det. no.
SCCP	50.27	14.00	108.00	35.00	13.00	91.00	15/15
MCCP	28.23	8.20	76.00	29.14	6.10	68.00	15/15
Siloxanes	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
D4	0.54	<0.17	1.75	1.12	<0.52	6.83	11/6
D5	1.01	<0.40	2.42	56.12	<0.17	205.92	11/12
D6	0.55	<0.17	2.01	11.85	<0.17	65.46	10/12
M3T(Ph)	n.d.	<0.17	<0.17	0.81	<0.17	8.02	0/10
Phenolic compounds	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
Bisphenol A	5.47	<10	82	n.d.	<20	<20	1/0
TBBPA	n.d.	<6	<6	n.d.	<10	<10	0/0
4,4-bisphenol F	n.d.	<7	<7	6.85	<60	103	0/1
2,2-bisphenol F	n.d.	<1	<1	n.d.	<5	<5	0/0
Bisphenol AF	n.d.	<1	<1	n.d.	<2	<2	0/0
Bisphenol S	n.d.	<2	<2	n.d.	<2	<2	0/0
4-nonylphenol	n.d.	<1	<1	n.d.	<2	<2	0/0
4-tert-octylphenol	n.d.	<600	<600	n.d.	<250	<250	0/0
Bisphenol B	n.d.	<1	<1	n.d.	<2	<2	0/0
Bisphenol Z	n.d.	<5	<5	n.d.	<5	<5	0/0
Bisphenol AP	n.d.	<1	<1	n.d.	<1	<1	0/0
Bisphenol E	n.d.	<20	<20	n.d.	<20	<20	0/0
Bisphenol FL	n.d.	<1	<1	n.d.	<5	<5	0/0
Bisphenol M	n.d.	<1	<1	n.d.	<1	<1	0/0
Dodekylphenol	n.d.	<1	<1	n.d.	<2	<2	0/0
Metals	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
Cr	0.007	<0.002	0.064	0.027	0.002	0.190	5/15
Fe	571.988	438.091	716.533	35.096	24.272	45.592	15/15
Ni	0.057	<0.027	0.109	0.031	0.009	0.144	14/15
Cu	0.503	0.387	0.676	0.816	0.607	0.967	15/15

Zn	6.265	4.954	8.223	15.212	11.926	20.130	15/15
As	0.080	0.011	0.328	0.054	0.006	0.131	15/15
Ag	0.000	<0.0001	0.003	0.001	0.000	0.001	11/15
Cd	0.001	0.001	0.002	0.000	0.000	0.000	15/15
Sb	0.000	<0.0001	0.000	0.000	<0.0001	0.000	4/2
Pb	0.098	0.025	0.193	0.012	0.002	0.035	15/15
Hg	88.595	17.051	288.577	62.708	9.790	166.890	15/15
PFAS compounds	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
PFPA	n.d.	<5	<5	n.d.	<0.5	<0.5	0/0
PFHxA	n.d.	<5	<5	n.d.	<0.5	<0.5	0/0
PFHpA	n.d.	<5	<5	n.d.	<0.5	<0.5	0/0
PFOA	n.d.	<5	<5	0.39	<0.5	1.60	0/4
PFNA	n.d.	<5	<5	0.19	<0.5	0.86	0/4
PFDA	0.48	<0.5	0.94	0.31	<0.5	1.87	10/4
PFUdA	0.24	<0.4	0.66	0.49	<0.4	1.51	7/12
PFDoA	0.78	<0.4	3.01	0.82	<0.4	4.42	12/12
PFTrDA	0.69	<0.4	1.14	1.01	<0.4	2.77	14/12
PFTeDA	0.52	<0.4	2.16	0.95	<0.4	3.61	8/13
PFBS	n.d.	<0.2	<0.2	n.d.	<0.2	<0.2	0/0
PFPS	n.d.	<0.2	<0.2	n.d.	<0.2	<0.2	0/0
PFHxS	0.37	0.03	0.72	0.07	<0.1	0.36	15/5
PFHpS	0.01	<0.2	0.20	0.07	<0.2	0.60	1/12
PFOS	11.20	2.72	26.48	25.55	4.17	172.29	15/15
8Cl-PFOS	n.d.	<0.2	<0.2	n.d.	<0.2	<0.2	0/0
PFNS	n.d.	<0.2	<0.2	n.d.	<0.2	<0.2	0/0
PFDS	n.d.	<0.2	<0.2	0.23	<0.2	1.65	0/5
PFDoS	n.d.	<0.2	<0.2	n.d.	<0.2	<0.2	0/0
PFOSA	n.d.	<0.1	<0.1	n.d.	<0.1	<0.1	0/0
me-FOSA	n.d.	<0.3	<0.3	n.d.	<0.3	<0.3	0/0
et-FOSA	n.d.	<0.3	<0.3	n.d.	<0.3	<0.3	0/0
me-FOSE	n.d.	<5	<5	n.d.	<5	<5	0/0

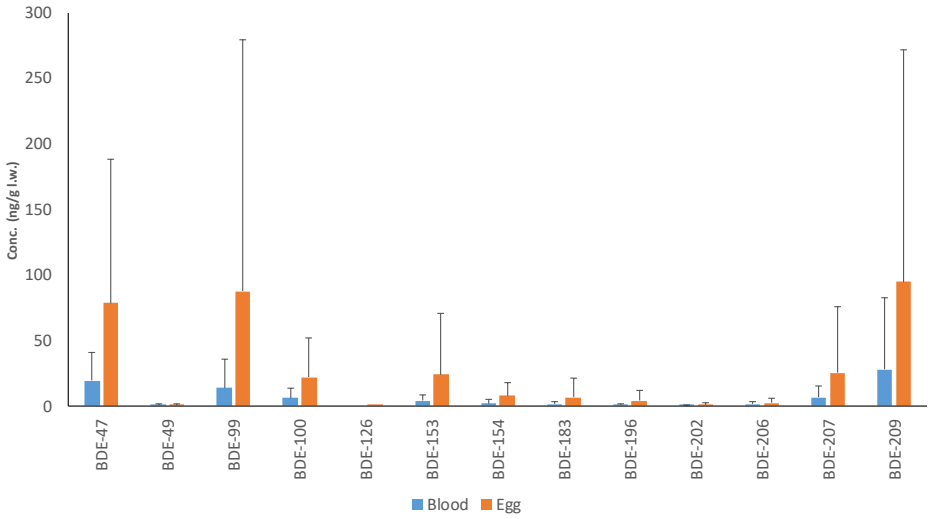
et-FOSE	n.d.	<5	<5	n.d.	<5	<5	0/0
4:2 FTS	n.d.	<0.3	<0.3	n.d.	<0.3	<0.3	0/0
6:2 FTS	0.17	<0.3	2.54	n.d.	<0.3	<0.3	1/0
8:2 FTS	0.88	<0.3	12.16	1.51	<0.3	18.51	2/7
me-FOSAA	n.d.	<0.3	<0.3	n.d.	<0.3	<0.3	0/0
Et-FOSAA	n.d.	<0.3	<0.3	n.d.	<0.3	<0.3	0/0
UV-chemicals	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
BP3	n.d.	<1	<1	n.d.	<2	<6	0/0
EHMC	n.d.	<7	<7	n.d.	<5	<5	0/0
OC	2.14	<8	32.12	2.40	<15	36.00	1/1
Behentrimonium	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
ATAC-C20	n.d.	<1	<1	n.d.	<1	<1	0/0
ATAC-C22	1.16	<1	3.20	0.79	<1	2.80	9/7

Concentrations of selected contaminants, specifically PBDEs (lipid wt. basis), siloxanes (lipid wt. basis) and PFAS compounds (wet wt. basis) in herring gull (blood and egg) are also presented in Figure 21 to Figure 23. The figures include tables with concentrations (on relevant basis: wet wt. or lipid wt.).

Dechlorane plus was found in eggs of herring gull and the variability was high (Table 12). Dechlorane plus is marketed as a flame retardant alternative to deca-BDE.

The PBDE congeners displaying the highest concentrations in herring gull from the Inner Oslofjord (both blood and eggs) were BDE-209, -47 and -99, although variability was high (Figure 21). This corresponds with previous observations from the Urban fjord programme (Ruus et al. 2017; Ruus et al. 2016; Ruus et al. 2015; Ruus et al. 2014; The Norwegian Environment Agency M-812, M-601, M-375 and M-205). In blood, concentrations of DBDPE were even higher than the above mentioned PBDE congeners (Table 12). DBDPE is a substitute for BDE-209 in the market. The same was observed in 2016 (Ruus et al. 2017; The Norwegian Environment Agency M-812) and future monitoring will indicate potential temporal trends. As observed/mentioned earlier (Ruus et al. 2015; Ruus et al. 2016; Ruus et al. 2017; The Norwegian Environment Agency M-375, M-601 and M-812), the concentrations of PBDEs (e.g. BDE-47 and -209) in herring gull eggs from the present study displayed concentrations that were higher than those observed in herring gull eggs from remote colonies in Norway (Sklinna and Røst; Huber et al. 2015) a few years ago, indicating urban influence. It can also be mentioned that according to Gentes et al. (2015), intraspecific forage strategies have strong influence on the PBDE accumulation in gulls, and that foraging on waste management facilities particularly results in higher BDE-209 exposure. As mentioned earlier, some PBDE congeners, such as BDE-209 in the herring gull eggs appeared somewhat higher than what was observed in eggs of sparrow hawk (a small bird of prey feeding on small to medium sized birds) from the Oslo area (Heimstad et al. 2017; The Norwegian Environment Agency M-1076).

A.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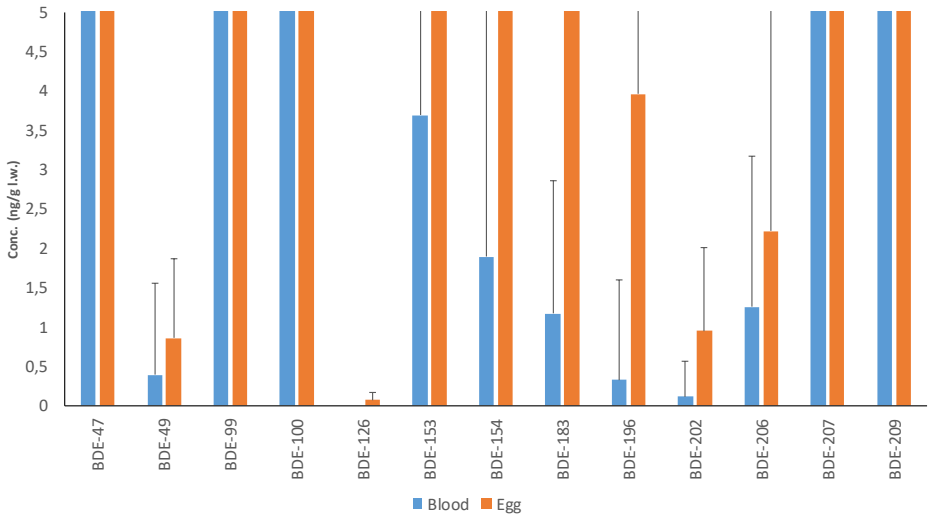
BDE-49
BDE-99
BDE-100
BDE-126
BDE-153
BDE-154
BDE-183
BDE-196
BDE-202
BDE-206
BDE-207
BDE-209
Blood
Egg

B.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Conc. (ng/g l.w.)
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BDE-49
BDE-99
BDE-100
BDE-126
BDE-153
BDE-154
BDE-183
BDE-196
BDE-202
BDE-206
BDE-207
BDE-209
Blood
Egg

C.

Analyte	Blood Mean	Blood Min.	Blood Max.	Egg Mean	Egg Min.	Egg Max.	Det. no.
Lipid content (%)	2.280	0.600	5.300	8.013	5.920	9.970	15/15
PBDEs							
BDE-47	19.116	4.245	84.400	78.566	5.182	423.457	15/15
BDE-49	0.393	n.d.	4.540	0.861	n.d.	3.312	4/13
BDE-99	13.441	1.558	90.500	87.427	4.839	665.432	15/15
BDE-100	5.806	0.730	31.400	21.617	1.766	120.988	15/15
BDE-126	n.d.	n.d.	n.d.	0.071	n.d.	0.312	0/9
BDE-153	3.695	n.d.	17.400	23.937	1.161	172.840	11/15
BDE-154	1.892	n.d.	13.000	7.813	0.642	34.815	9/15
BDE-183	1.173	n.d.	4.640	6.613	0.353	57.037	8/15
BDE-196	0.327	n.d.	4.910	3.967	n.d.	24.568	1/13
BDE-197	1.887	n.d.	9.717	7.651	0.505	52.716	6/15
BDE-202	0.116	n.d.	1.735	0.950	n.d.	3.654	1/12
BDE-206	1.258	n.d.	4.700	2.223	n.d.	12.346	5/10
BDE-207	6.292	n.d.	28.400	25.262	1.057	172.840	10/15
BDE-209	27.705	n.d.	181.000	94.633	n.d.	613.580	4/14

Figure 21. A. Concentrations of PBDEs (ng/g lipid wt.) in herring gull (blood and eggs) from the Inner Oslofjord (mean and standard deviation; n=15; non-detects are assigned values of zero). B. Magnification of the lower part (0-5) of the concentration axis in A. C. Lipid content (%) and concentrations of PBDEs in herring gull blood and egg from the Inner Oslofjord (ng/g lipid wt.) presented in a table. Arithmetic mean and range (minimum and maximum) are presented (n=15). In calculations of mean, non-detected components were assigned a value of zero (0). Det. no. is the number of samples in which the substance was detected (blood/egg).

Siloxanes were detected in eggs and blood of herring gull from the Inner Oslofjord (Figure 22). D5 displayed the highest concentrations but the variability was high. This corresponds with previous observations from the Urban fjord programme (Ruus et al. 2017; Ruus et al. 2016; Ruus et al. 2015; Ruus et al. 2014; The Norwegian Environment Agency M-812, M-601, M-375 nad M-205). In 2017, M3T(Ph) was also analysed and was detected only in eggs (10 of 15 samples; Figure 22).

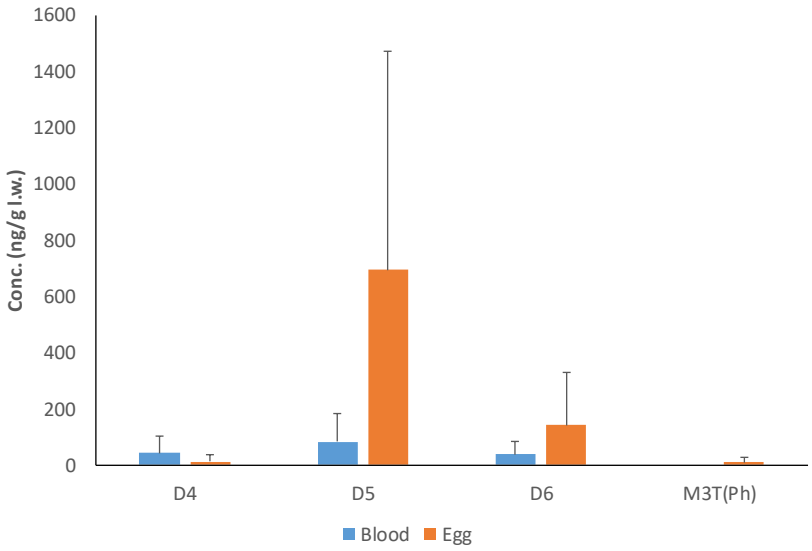
Mean D5 concentration in eggs from the Oslofjord area (present study) was a factor of ~47 higher than those observed in herring gull eggs from remote colonies in Norway (Sklinna and Røst; Huber et al. 2015) a few years ago, indicating urban influence. As such, the D5 concentrations in herring gull eggs from the Inner Oslofjord in 2017 appeared somewhat lower

than in 2016 (Ruus et al. 2017; The Norwegian Environment Agency M-812; not tested statistically, and variability was high).As earlier observed (Ruus et al. 2017; The Norwegian Environment Agency M-812), the mean concentration of siloxanes in the herring gull eggs from the Oslofjord area also appeared higher than in eggs of sparrow hawk (*Accipiter nisus*) from the Oslo area (Heimstad et al. 2018; The Norwegian Environment Agency M-1076). This may also reflect that while the sparrow hawk feeds mostly on birds, the herring gull might feed on human waste and leftovers.

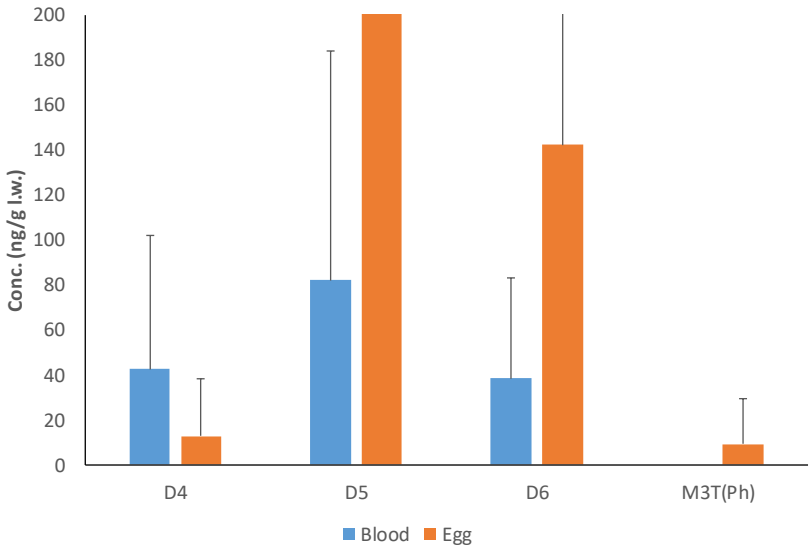
As previously observed (Ruus et al. 2017; The Norwegian Environment Agency M-812), concentrations of "legacy" contaminants, such as PCB-153 and SumPCB7 appeared lower in the eggs from Oslofjorden, than those observed in herring gull eggs from remote colonies in Norway (Sklinna and Røst; Huber et al. 2015). This suggests that these contaminants (associated with diffuse pollution) accumulate to somewhat higher concentrations in gulls foraging to a larger degree on marine prey organisms. However, the concentrations of PCBs in the sparrow hawk eggs from the Oslo area (Heimstad et al. 2018; The Norwegian Environment Agency M-1076) appeared higher than in the herring gull eggs from the Oslofjord area (Table 12). This was also observed in 2016 (Ruus et al. 2017; The Norwegian Environment Agency M-812).

The consistent herring gull results between years in the "Urban fjord" programme, suggest the suitability of this species to study urban influence. In this regard, it is important to acknowledge that with the opportunistic feeding habits of herring gull, urbanisation implies a shift towards less marine diet items and more diet items of terrestrial/anthropogenic origin.

A.



B.



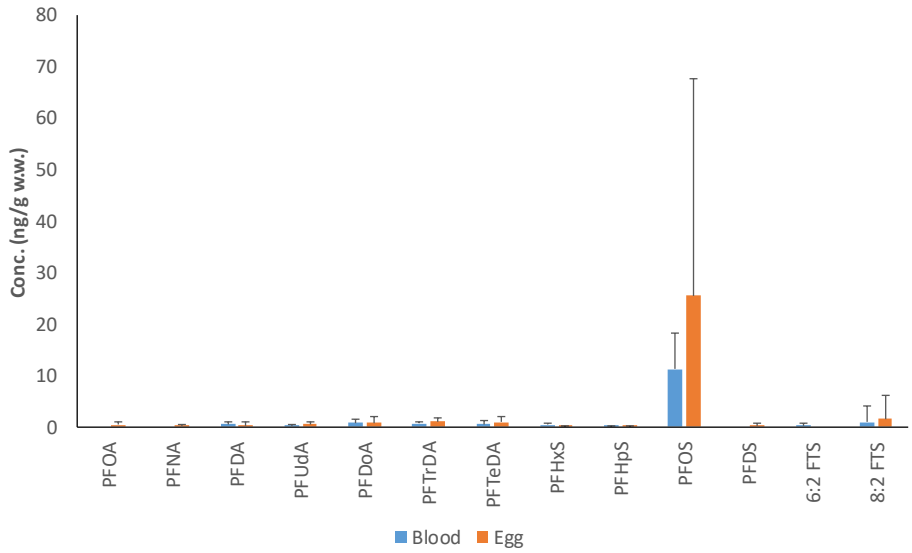
C.

Analyte	Blood Mean	Blood Min.	Blood Max.	Egg Mean	Egg Min.	Egg Max.	Det. no.
Lipid content (%)	2.280	0.600	5.300	8.013	5.920	9.970	15/15
Siloxanes							
D4	42.55	n.d.	234.65	12.81	n.d	84.25	11/6
D5	81.94	n.d.	326.22	694.13	n.d.	2340.05	11/12
D6	38.51	n.d.	134.02	142.15	n.d.	656.58	10/12
M3T(Ph)	n.d.	n.d.	n.d.	9.04	n.d.	80.41	0/10

Figure 22. A. Concentrations of siloxanes (ng/g lipid wt.) in herring gull (blood and eggs) from the Inner Oslofjord (mean and standard deviation; n=15; non-detects are assigned values of zero). B. Magnification of the lower part (0-200) of the concentration axis in A. C. Lipid content (%) and concentrations of siloxanes in herring gull blood and egg from the Inner Oslofjord (ng/g lipid wt.) presented in a table. Arithmetic mean and range (minimum and maximum) are presented (n=15). In calculations of mean, non-detected components were assigned a value of zero (0). Det. no. is the number of samples in which the substance was detected (blood/egg).

PFAS compounds were also detected in eggs and blood of herring gull from the Inner Oslofjord (Figure 23). PFOS constituted the highest concentrations in both matrices. The variability was high. This corresponds with previous observations from the Urban fjord programme (Ruus et al. 2017; Ruus et al. 2016; Ruus et al. 2015; Ruus et al. 2014; The Norwegian Environment Agency M-812, M-601, M-375 and M-205). PFOS was also the dominating PFAS compound in sparrow hawk eggs from the Oslo area (Heimstad et al. 2018; The Norwegian Environment Agency M-1076), and as in 2016 (Ruus et al. 2017; The Norwegian Environment Agency M-812) the PFOS concentrations appeared higher than in the herring gull eggs (Table 12)

A.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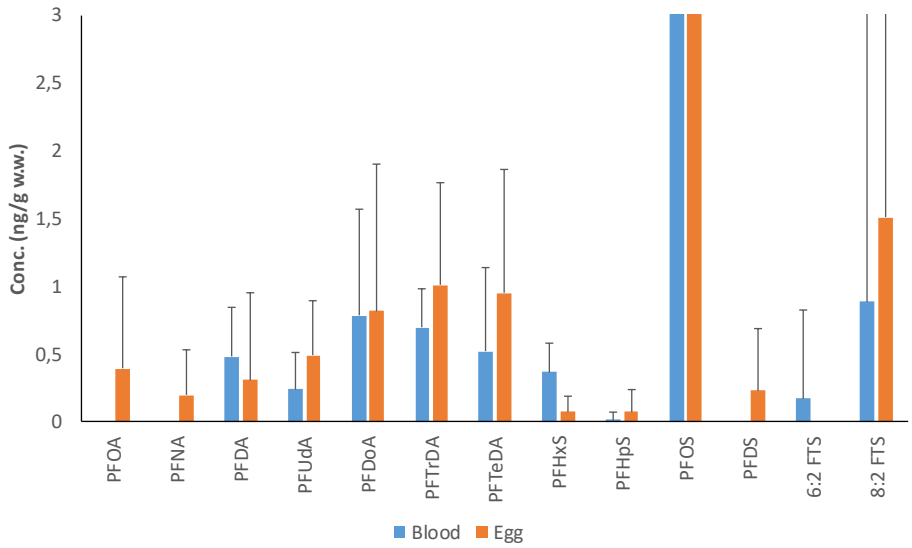
PFNA
PFDA
PFUdA
PFDoA
PFTrDA
PFTeDA
PFHxS
PFHpS
PFOS
PFDS
6:2 FTS
8:2 FTS
Blood
Egg

B.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Conc. (ng/g w.w.)
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PFDA
PFUdA
PFDoA
PFTrDA
PFTeDA
PFHxS
PFHpS
PFOS
PFDS
6:2 FTS
8:2 FTS
Blood
Egg

C.

Analyte	Blood Mean	Blood Min.	Blood Max.	Egg Mean	Egg Min.	Egg Max.	Det. no.
Lipid content (%)	2.23	0.40	14.80	9.02	5.80	13.50	15/15
PFAS compounds							
PFOA	n.d.	<5	<5	0.39	<0.5	1.60	0/4
PFNA	n.d.	<5	<5	0.19	<0.5	0.86	0/4
PFDA	0.48	<0.5	0.94	0.31	<0.5	1.87	10/4
PFUdA	0.24	<0.4	0.66	0.49	<0.4	1.51	7/12
PFDoA	0.78	<0.4	3.01	0.82	<0.4	4.42	12/12
PFTrDA	0.69	<0.4	1.14	1.01	<0.4	2.77	14/12
PFTeDA	0.52	<0.4	2.16	0.95	<0.4	3.61	8/13
PFHxS	0.37	0.03	0.72	0.07	<0.1	0.36	15/5
PFHpS	0.01	<0.2	0.20	0.07	<0.2	0.60	1/12
PFOS	11.20	2.72	26.48	25.55	4.17	172.29	15/15
PFDS	n.d.	<0.2	<0.2	0.23	<0.2	1.65	0/5
6:2 FTS	0.17	<0.3	2.54	n.d.	<0.3	<0.3	1/0
8:2 FTS	0.88	<0.3	12.16	1.51	<0.3	18.51	2/7

Figure 23. A. Concentrations (ng/g wet wt.) of PFAS in herring gull (blood and eggs) from the Inner Oslofjord (mean and standard deviation; n=15; non-detects are assigned values of zero). B. Magnification of the lower part (0-3) of the concentration axis in A. C. Lipid content (%) and concentrations of PFAS in herring gull blood and egg from the Inner Oslofjord (ng/g wet wt.) presented in a table. Arithmetic mean and range (minimum and maximum) are presented (n=15). In calculations of mean, non-detected components were assigned a value of zero (0). Det. no. is the number of samples in which the substance was detected (blood/egg). The following compounds were detected in neither blood, nor egg: PFPA, PFHxA, PFHpA, PFBS, PFPS, 8Cl-PFOS, PFNS, PFDoS, PFOSA, me-FOSA, et-FOSA, me-FOSE, et-FOSE, 4:2FTS, me-FOSAA, et-FOSAA.

Outer Oslofjord

Both blood and egg were sampled from herring gull also in the Outer Oslofjord. Adult female blood and egg was sampled from the same nest (i.e. mother and future offspring).

Concentrations (mean and range; wet wt. basis) for all compounds and elements analysed in herring gull (blood and egg) are presented in Table 13. The number of samples in which the substance was detected is also shown in Table 13.

Table 13.
Lipid content (%) and concentrations of the different analytes in herring gull blood and egg from the Outer Oslofjord. Concentrations are ng/g wet wt., except for concentrations of Ni, Cu, Ag, Cd, Pb, Cr, Fe, Zn, As and Sb, which are expressed as µg/g wet wt. Arithmetic mean and range (minimum and maximum) are presented (n=15). In calculations of mean, non-detected components were assigned a value of zero (0). Det. no. is the number of samples in which the substance was detected (blood/egg). No phenolic compounds were detected, hence not included in the table.

Analyte	Blood Mean	Blood Min.	Blood Max.	Egg Mean	Egg Min.	Egg Max.	Det. no.
Lipid content (%)	2.27	1.10	4.30	7.09	3.98	9.30	15/15
PeCB	0.044	<0.082	0.656	0.201	0.091	0.400	1/15
HCB	0.794	0.106	8.490	3.033	0.848	6.660	15/15
MB1	n.d.	<2	<2	n.d.	<1	<1	0/0
Dechlorane	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
Dechlorane 602	0.011	<0.020	0.161	n.d.	<0.005	<0.005	1/0
Dechlorane 603	n.d.	<0.020	<0.020	0.027	<0.005	0.209	0/12
Dechlorane 604	n.d.	<0.080	<0.080	n.d.	<0.020	<0.020	0/0
Dechlorane plus syn	0.017	<0.100	0.152	0.183	<0.033	1.246	2/14
Dechlorane plus anti	0.032	<0.200	0.252	0.618	<0.067	3.619	2/14
PCBs (PCB7)	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
PCB-28	0.260	<0.047	3.620	0.668	0.138	2.250	4/15
PCB-52	1.081	<0.051	15.600	1.481	0.191	7.950	4/15
PCB-101	3.302	<0.418	48.800	3.013	0.263	19.000	2/15
PCB-118	4.959	<0.354	59.700	17.919	3.490	53.600	13/15
PCB-138	8.315	<0.745	88.000	39.608	8.400	111.000	13/15
PCB-153	11.179	<1.300	113.000	64.927	15.200	177.000	13/15
PCB-180	4.645	<0.333	57.000	15.261	4.130	40.000	13/15
Sum-PCB_7	33.740	n.d.	385.720	142.877	31.938	399.170	15/15
Brominated comp.	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
BDE-47	0.501	<0.095	4.830	3.132	0.514	7.860	14/15
BDE-49	0.061	<0.005	0.584	0.072	<0.003	0.220	6/14
BDE-99	0.183	0.036	1.140	1.555	0.463	3.660	15/15

BDE-100	0.106	<0.028	0.931	0.906	0.146	2.350	12/15
BDE-126	n.d.	<0.004	<0.054	0.002	<0.003	0.009	0/4
BDE-153	0.094	<0.009	1.130	0.460	0.169	1.150	9/15
BDE-154	0.087	<0.007	1.170	0.310	0.067	0.906	7/15
BDE-183	0.014	<0.009	0.127	0.158	0.030	0.877	3/15
BDE-196	0.004	<0.02	0.061	0.076	<0.02	0.314	1/14
BDE-202	n.d.	<0.019	<0.19	0.055	0.014	0.117	0/15
BDE-206	0.066	<0.029	0.708	0.065	<0.007	0.243	6/9
BDE-207	0.088	<0.02	0.635	0.407	0.026	1.490	8/15
BDE-209	0.549	<0.581	4.070	2.436	<0.145	8.990	3/13
ATE (TBP-AE)	n.d.	<0.038	<0.054	n.d.	<0.047	<0.047	0/0
a-TBECH	n.d.	<0.093	<0.116	n.d.	<0.116	<0.116	0/0
b-TBECH	n.d.	<0.067	<0.111	n.d.	<0.083	<0.083	0/0
g/d-TBECH	n.d.	<0.024	<0.034	n.d.	<0.03	<0.03	0/0
BATE	n.d.	<0.012	<0.021	n.d.	<0.015	<0.015	0/0
PBT	n.d.	<0.08	<0.133	n.d.	<0.146	<0.146	0/0
PBEB	n.d.	<0.083	<0.125	n.d.	<0.141	<0.141	0/0
HBB	n.d.	<0.047	<0.066	n.d.	<0.058	<0.058	0/0
DPTE	n.d.	<0.013	<0.021	n.d.	<0.016	<0.016	0/0
EHTBB	n.d.	<0.029	<0.041	n.d.	<0.016	<0.016	0/0
BTBPE	n.d.	<0.048	<0.069	n.d.	<0.06	<0.06	0/0
TBPH (BEH /TBP)	n.d.	<0.093	<0.133	n.d.	<0.201	<0.201	0/0
DBDPE	5.730	3.980	7.770	n.d.	<3.3	<3.3	4/0
Chloroparaffins	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
SCCP	30.29	5.00	200.00	42.00	18.00	178.00	15/15
MCCP	38.87	5.80	200.00	69.58	3.10	630.00	15/15
Siloxanes	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
D4	n.d.	<1.31	<1.31	n.d.	<3.90	<3.90	0/0
D5	0.26	<0.62	1.21	111.40	13.04	695.01	8/15
D6	1.72	0.71	3.56	8.96	3.89	19.68	15/15
M3T(Ph)	n.d.	<0.57	<0.57	n.d.	<0.75	<0.75	0/0

Metals	Blood Mean	Blood Min.	Blood Max.	Egg Mean	Egg Min.	Egg Max.	Det. no.
Cr	0.009	<0.002	0.139	0.028	0.003	0.173	2/15
Fe	566.187	402.873	883.694	33.274	23.528	44.278	15/15
Ni	0.030	<0.032	0.066	0.040	0.001	0.127	9/15
Cu	0.501	0.416	0.638	0.786	0.642	1.068	15/15
Zn	5.665	4.101	7.346	15.666	10.035	25.646	15/15
As	0.121	0.015	0.550	0.124	0.034	0.251	15/15
Ag	0.000	<0.0002	0.002	0.001	0.000	0.002	8/15
Cd	0.001	0.000	0.003	0.000	0.000	0.000	15/15
Sb	0.000	<0.0001	0.000	0.000	<0.0001	0.001	2/11
Pb	0.054	0.022	0.120	0.007	0.003	0.012	15/15
Hg	111.789	25.742	287.840	84.527	25.153	225.872	15/15
PFAS compounds	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
PFPA	n.d.	<5	<5	n.d.	<0.5	<0.5	0/0
PFHxA	n.d.	<5	<5	n.d.	<0.5	<0.5	0/0
PFHpA	n.d.	<5	<5	0.11	<0.5	1.72	0/1
PFOA	n.d.	<5	<5	1.04	<0.5	10.39	0/6
PFNA	n.d.	<5	<5	0.50	<0.5	4.33	0/6
PFDA	1.34	<0.5	3.16	1.13	<0.5	8.88	14/12
PFUdA	1.15	0.47	2.10	0.96	<0.4	2.46	15/13
PFDoA	1.41	0.59	2.89	1.06	<0.4	6.26	15/13
PFTrDA	1.62	<0.4	3.33	1.76	<0.4	5.51	15/15
PFTeDA	1.00	<0.4	2.52	1.18	<0.4	6.49	15/12
PFBS	n.d.	<0.2	<0.2	n.d.	<0.2	<0.2	0/0
PFPS	n.d.	<0.2	<0.2	n.d.	<0.2	<0.2	0/0
PFHxS	0.96	0.42	3.36	0.23	<0.1	1.76	15/7
PFHpS	0.12	<0.2	0.55	0.08	<0.2	0.97	5/13
PFOS	18.68	6.45	51.36	38.47	4.39	126.11	15/15
8Cl-PFOS	n.d.	<0.2	<0.2	n.d.	<0.2	<0.2	0/0
PFNS	n.d.	<0.2	<0.2	n.d.	<0.2	<0.2	0/0
PFDS	n.d.	<0.2	<0.2	0.37	<0.2	1.64	0/6

PFDoS	n.d.	<0.2	<0.2	n.d.	<0.2	<0.2	0/0
PFOSA	n.d.	<0.1	<0.1	0.01	<0.1	0.16	0/1
me-FOSA	n.d.	<0.3	<0.3	n.d.	<0.3	<0.3	0/0
et-FOSA	n.d.	<0.3	<0.3	n.d.	<0.3	<0.3	0/0
me-FOSE	n.d.	<5	<5	n.d.	<5	<5	0/0
et-FOSE	n.d.	<5	<5	n.d.	<5	<5	0/0
4:2 FTS	n.d.	<0.3	<0.3	n.d.	<0.3	<0.3	0/0
6:2 FTS	n.d.	<0.3	<0.3	n.d.	<0.3	<0.3	0/0
8:2 FTS	0.32	<0.3	4.76	0.66	<0.3	5.14	1/6
me-FOSAA	n.d.	<0.3	<0.3	n.d.	<0.3	<0.3	0/0
Et-FOSAA	n.d.	<0.3	<0.3	n.d.	<0.3	<0.3	0/0
UV-chemicals	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
BP3	n.d.	<1	<1	n.d.	<2	<2	0/0
EHMC	n.d.	<7	<7	n.d.	<3	<10	0/0
OC	7.26	<8	36.55	n.d.	<15	<15	6/0

As indicated in Chapter 3.1, according to the results of the stable isotope analysis, the Outer Oslofjord gulls may include more diet items of marine origin in their diet, than the inner Oslofjord gulls. There were also some differences in concentrations of contaminants between the colonies, although many appeared similar (Mann-Whitney U on wet wt. concentrations; $p<0.05$). The most conspicuous was as follows (Knudtzon in prep.; Thorstensen in prep.): D4 was only detected in the Inner Oslofjord (in both blood and eggs). D5 was also higher in the Inner Oslofjord (blood), while D6 was higher in the Outer Oslofjord (blood). M3T(Ph) was only detected in the inner fjord (eggs). Several of the PFAS compounds (e.g. PFOS) was found in higher concentrations in the gulls of the Outer Oslofjord (both blood and eggs), possibly related to contamination in the area because of an earlier airport in proximity of the colony.

As in eggs of herring gulls from the Inner Oslofjord, Dechlorane plus was found in eggs of herring gull from the outer Oslofjord, and the variability was even higher than in the inner fjord (Table 13; Figure 24).

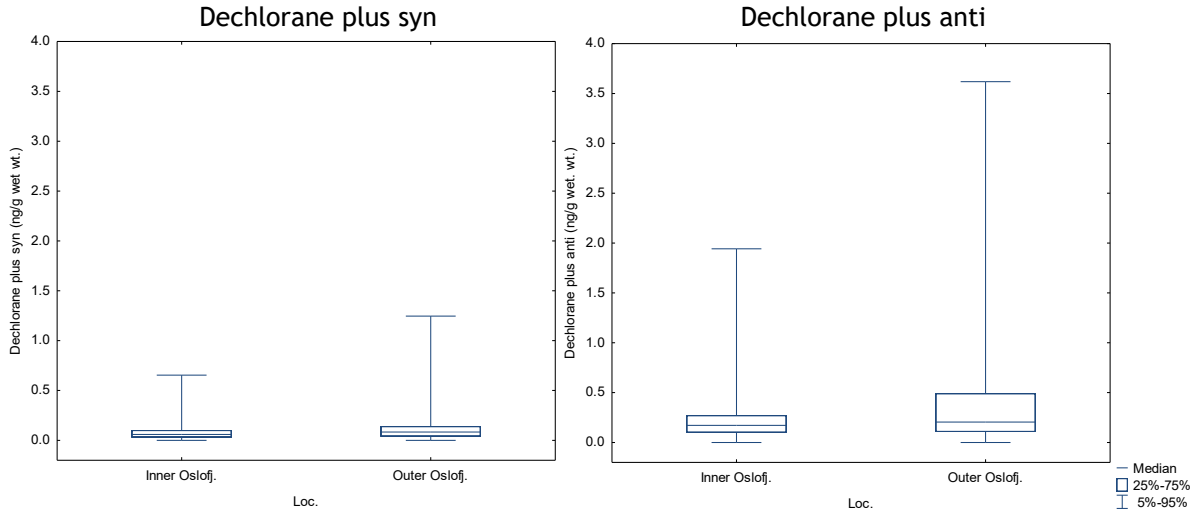


Figure 24. Concentrations (ng/g wet wt.) of dechlorane plus in eggs of Herring gull from the Inner and Outer Oslofjord, syn (left) and anti (right) isomers are shown. Median, 25-75 percentiles and 5-95 percientiles are given.

3.2.5 Eider duck

Both blood and egg were sampled also from eider duck in the Inner Oslofjord. Adult female blood and egg was sampled from the same nest (i.e. mother and future offspring).

Concentrations (mean and range; wet wt. basis) for all compounds and elements analysed in eider duck (blood and egg) are presented in Table 14. The number of samples in which the substance was detected is also shown in Table 14. Note that Eider duck was analysed as an addition to the programme in 2017. Because of a limited budget for these additional analyses, all analytes in the programme were not analysed, and PFAS compounds were only analysed in blood.

Table 14.
Lipid content (%) and concentrations of the different analytes in eider duck blood and egg from the Inner Oslofjord. Concentrations are ng/g wet wt. Arithmetic mean and range (minimum and maximum) are presented (n=15). In calculations of mean, non-detected components were assigned a value of zero (0). Det. no. is the number of samples in which the substance was detected (blood/egg).

Analyte	Blood Mean	Blood Min.	Blood Max.	Egg Mean	Egg Min.	Egg Max.	Det. no.
Lipid content (%)	0.39	0.14	1.25	18.43	16.90	21.00	15/15
PeCB	n.d.	<0.082	<0.082	0.398	0.254	0.513	0/15
HCB	0.245	0.101	0.419	2.919	1.800	4.170	15/15
Dechlorane	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
Dechlorane 602	0.004	<0.020	0.030	0.121	0.054	0.297	2/15
Dechlorane 603	n.d.	<0.020	<0.020	0.027	<0.020	0.272	0/5

Dechlorane 604	n.d.	<0.100	<0.100	n.d.	<0.100	<0.100	0/0
Dechlorane plus syn	0.007	<0.100	0.108	0.128	<0.100	0.224	1/13
Dechlorane plus anti	0.050	<0.200	0.263	0.245	<0.200	0.520	3/12
PCBs (PCB7)	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
PCB-28	0.156	<0.047	0.484	1.892	0.883	4.110	14/15
PCB-52	0.008	<0.051	0.060	1.579	0.404	3.310	2/15
PCB-101	n.d.	<0.418	0.000	3.096	1.260	6.420	0/15
PCB-118	2.106	0.415	9.270	27.920	11.700	66.400	15/15
PCB-138	2.772	<0.745	13.500	33.407	15.400	86.500	13/15
PCB-153	4.697	<1.300	20.000	60.193	28.700	150.000	14/15
PCB-180	0.781	<0.333	4.460	10.225	3.570	33.500	9/15
Sum-PCB_7	10.519	0.529	47.714	138.312	62.753	350.020	15/15
Brominated comp.	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
BDE-47	0.005	<0.057	0.072	0.385	0.224	1.130	1/15
BDE-49	0.003	<0.005	0.048	0.020	0.008	0.078	1/15
BDE-99	0.003	<0.027	0.046	0.245	0.147	0.444	1/15
BDE-100	0.005	<0.012	0.038	0.227	0.117	0.791	3/15
BDE-126	0.001	<0.004	0.018	0.001	<0.004	0.009	1/3
BDE-153	0.016	<0.009	0.153	0.138	0.053	0.813	6/15
BDE-154	0.006	<0.007	0.037	0.115	0.070	0.493	6/15
BDE-183	0.001	<0.009	0.011	0.028	<0.009	0.053	2/14
BDE-196	n.d.	<0.02	<0.05	n.d.	<0.02	<0.02	0/0
BDE-197	n.d.	<0.015	<0.041	0.005	<0.015	0.028	0/4
BDE-202	n.d.	<0.019	<0.062	0.012	<0.019	0.044	0/6
BDE-206	0.003	<0.029	0.045	n.d.	<0.029	<0.029	1/0
BDE-207	n.d.	<0.02	<0.02	0.007	<0.02	0.048	0/3
BDE-209	n.d.	<0.581	<0.581	n.d.	<0.581	<0.581	0/0
Metals	**Blood Mean**	**Blood Min.**	**Blood Max.**	**Egg Mean**	**Egg Min.**	**Egg Max.**	**Det. no.**
Hg	187.159	<106.381	390.670	153.798	72.788	327.363	14/15

PFAS compounds	Blood Mean	Blood Min.	Blood Max.	Egg Mean	Egg Min.	Egg Max.	Det. no.
PFPA	n.d.	<5	<5	-	-	-	0/-
PFHxA	n.d.	<5	<5	-	-	-	0/-
PFHpA	n.d.	<5	<5	-	-	-	0/-
PFOA	n.d.	<5	<5	-	-	-	0/-
PFNA	n.d.	<5	<5	-	-	-	0/-
PFDA	1.44	<0.5	2.87	-	-	-	14/-
PFUdA	0.95	0.42	1.69	-	-	-	15/-
PFDoA	0.73	<0.4	1.32	-	-	-	14/-
PFTrDA	0.37	<0.4	0.94	-	-	-	9/-
PFTeDA	0.21	<0.4	1.35	-	-	-	4/-
PFBS	n.d.	<0.2	<0.2	-	-	-	0/-
PFPS	n.d.	<0.2	<0.2	-	-	-	0/-
PFHxS	1.98	0.48	5.22	-	-	-	15/-
PFHpS	0.22	<0.2	0.60	-	-	-	8/-
PFOS	14.37	5.64	35.21	-	-	-	15/-
8Cl-PFOS	n.d.	<0.2	<0.2	-	-	-	0/-
PFNS	n.d.	<0.2	<0.2	-	-	-	0/-
PFDS	n.d.	<0.2	<0.2	-	-	-	0/-
PFDoS	n.d.	<0.2	<0.2	-	-	-	0/-
PFOSA	0.40	<0.1	1.82	-	-	-	5/-
me-FOSA	n.d.	<0.3	<0.3	-	-	-	0/-
et-FOSA	n.d.	<0.3	<0.3	-	-	-	0/-
me-FOSE	n.d.	<5	<5	-	-	-	0/-
et-FOSE	n.d.	<5	<5	-	-	-	0/-
4:2 FTS	n.d.	<0.3	<0.3	-	-	-	0/-
6:2 FTS	n.d.	<0.3	<0.3	-	-	-	0/-
8:2 FTS	0.56	<0.3	4.23	-	-	-	3/-
me-FOSAA	n.d.	<0.3	<0.3	-	-	-	0/-
Et-FOSAA	n.d.	<0.3	<0.3	-	-	-	0/-

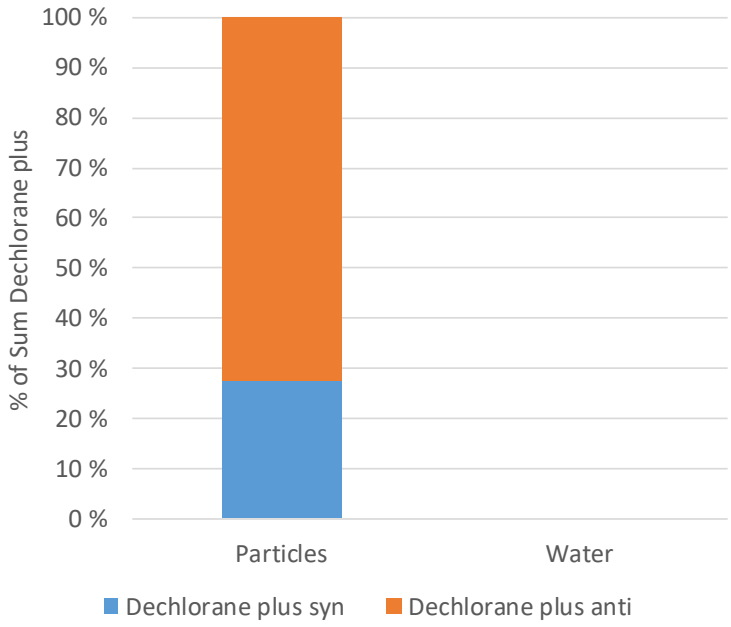
As expected and indicated in Chapter 3.1, according to the results of the stable isotope analysis, the Inner Oslofjord eider ducks have a diet consisting of more marine items, compared to the diet of the herring gulls sampled in the Inner Oslofjord. There were also some differences in concentrations of contaminants between the species, although many appeared similar (Mann-Whitney U on wet wt. concentrations; $p<0.05$). The most conspicuous was as follows (Knudtzon in prep.; Thorstensen in prep.): Several of the higher chlorinated PCBs showed higher concentrations in the Inner Oslofjord herring gull than in the eider duck (blood and eggs), despite significantly higher lipid content in the eider duck eggs. The same applies to a large number of PBDE congeners. Mercury, on the other hand showed higher concentrations in eider duck (both blood end eggs), possibly related to the marine diet of eider duck. Furthermore, there was great variability in the concentrations of PFAS compounds in eider duck blood (PFAS not analysed in eggs), and for some compounds, the concentrations were statistically significantly higher in eider duck, than in herring gull from the Inner Oslofjord (such as PFHxS).

As in eggs of herring gulls, both from the Inner and Outer Oslofjord, dechlorane plus was found in eggs of Eider ducks from the Inner Oslofjord (Table 14). Compared to herring gull, the variability was low. Furthermore, Dechlorane 602 was detected in all eider duck eggs (Table 14).

See Chapter 3.2.2 for insight in how concentrations in the eider duck relates to other species of the Inner Oslofjord marine food web.

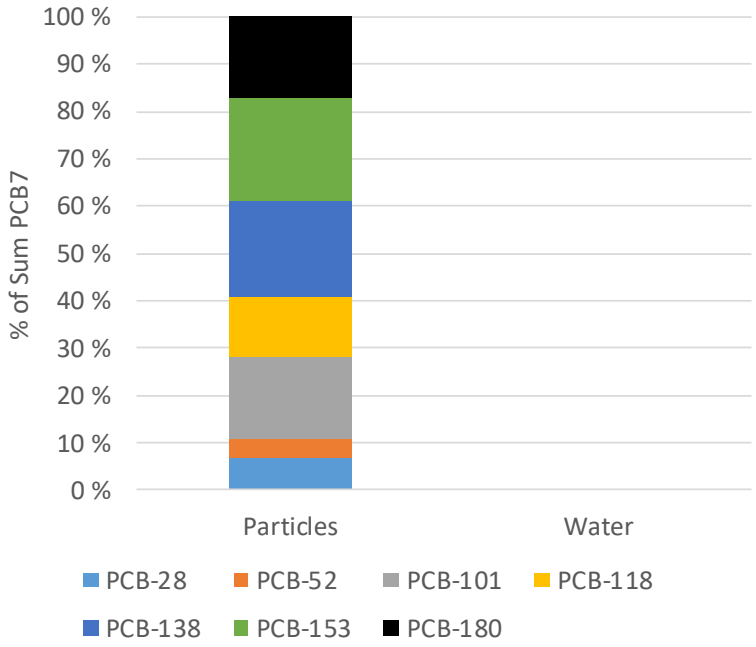
3.2.6 Storm water

The results of the chemical analysis of storm water can be found in the electronic Appendix. Dechlorane plus was found in concentrations of several ng/L, however only in the particulate fraction (Figure 25). PCB-concentrations were highest also in the particulate fraction. PCBs were not detected in the dissolved fraction (Figure 26). Given the hydrophobic nature of PCBs, they have a high affinity for the particulate phase and are usually associated with particles. BDE-concentrations were also higher in the particulate fraction, than in the dissolved fraction, as they were not detected in the dissolved fraction (Figure 27). BDE-209 constituted the highest percentage as in 2016 (Figure 27; Ruus et al. 2017; The Norwegian Environment Agency M-812). Interestingly, DBDPE was higher than BDE-209 both in the dissolved and in the particulate fraction (see electronic appendix). Furthermore, DBDPE was higher in the particulate fraction, than in the dissolved fraction.



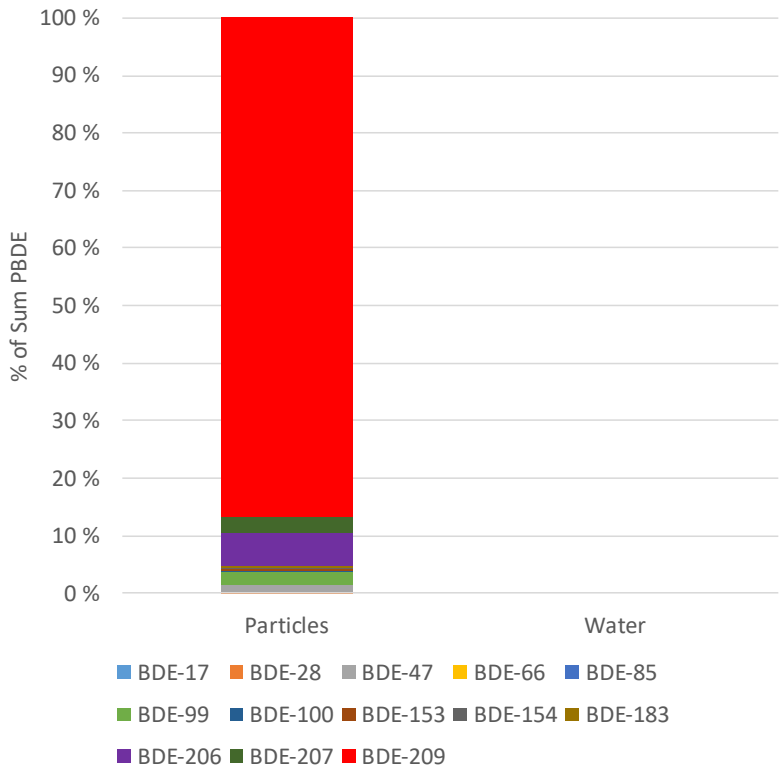
	Particles	Water
Dechlorane plus syn	4.377	n.d.
Dechlorane plus anti	11.917	n.d.

Figure 25. Relative contribution (%) of dechlorane plus syn and anti isomers to the sum of dechlorane plus in the particulate and dissolved fraction of storm water (mean of 2 samples.). Concentrations (ng/L; mean) are given in the associated table. Components that were not detected in any replicate samples of a fraction (particles or water) are noted n.d. Dechlorane plus syn and anti were the only dechlorane compounds detected in storm water.



	Particles	Water
PCB-28	1.06	n.d.
PCB-52	1.41	n.d.
PCB-101	3.04	n.d.
PCB-118	1.89	n.d.
PCB-138	3.42	n.d.
PCB-153	4.11	n.d.
PCB-180	3.89	n.d.

Figure 26. Relative contribution (%) of PCB-congeners to the sum of PCB7 in the particulate and dissolved fraction of storm water (mean of 2 samples. Non-detected components were assigned values of zero). Concentrations (ng/L; mean; non-detected components were assigned a value of zero) are given in the associated table. Components that were not detected in any replicate samples of a fraction (particles or water) are noted n.d.



	Particles	Water
BDE-17	0.038	n.d.
BDE-28	0.052	n.d.
BDE-47	2.885	n.d.
BDE-66	0.110	n.d.
BDE-85	0.146	n.d.
BDE-99	3.770	n.d.
BDE-100	0.800	n.d.
BDE-153	0.435	n.d.
BDE-154	0.304	n.d.
BDE-183	0.680	n.d.
BDE-206	3.534	n.d.
BDE-207	1.753	n.d.
BDE-209	90.105	n.d.

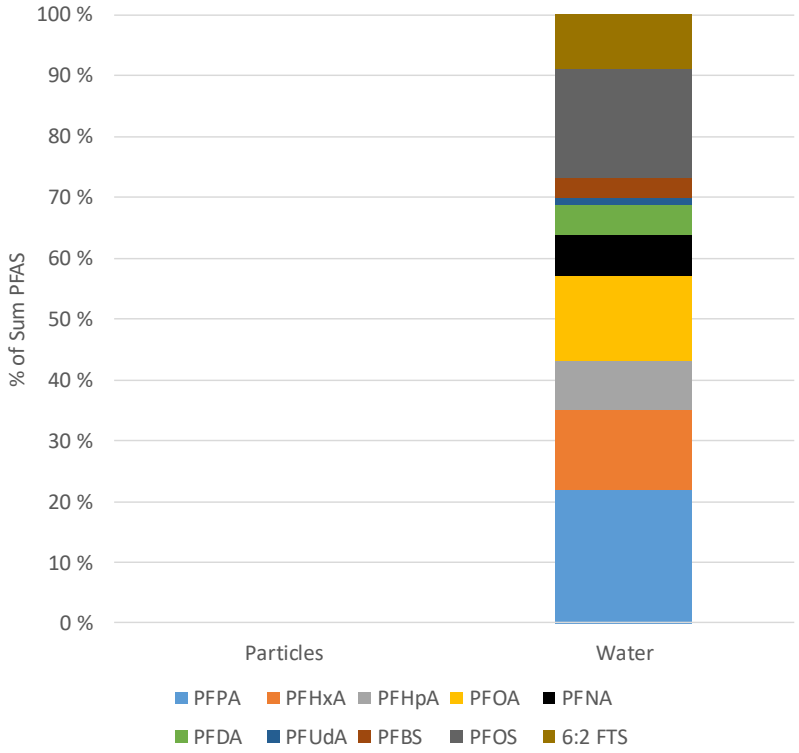
Figure 27. Relative contribution (%) of BDE-congeners to the sum of (detected) PBDEs in the particulate and dissolved fraction of storm water (mean of 2 samples. Non-detected components were assigned values of zero). Concentrations (ng/L; mean; non-detected components were assigned a value of zero) of detected components are given in the associated table. Components that were not detected in any replicate samples of a fraction (particles or water) are noted n.d.

PFAS compounds were only detected in the dissolved fraction of storm water (Figure 28). Nevertheless, inputs of several of the target compounds to the fjord via storm water are thus found. PFPA, PFOS and PFHxA displayed the highest concentrations (Figure 28). In 2016, PFPA and PFHxA also showed the highest concentration in the dissolved fraction of storm water (Ruus et al. 2017; The Norwegian Environment Agency M-812).

For several compounds, environmental quality standards for water are given through Norwegian law (The Water Regulation/"Vannforskriften"), according to the requirements of the Water Framework Directive. Furthermore, quality standards are given for even more compounds (The Norwegian Environment Agency M-608). For the target compounds of this study of which quality standards exist, the water concentrations (dissolved fraction) and quality standards are compared in Table 15 (quality standards for coastal water used, to elucidate the potential of surface water as source of contaminants to parts of the fjord).

Concentrations of bisphenol A, MCCPs, copper, zinc and PFOS exceeded the quality standards, reflecting runoff from the surrounding (urban) area. Copper, zinc and PFOS also exceeded the quality standards for sediment from station Cm21 (see chapter 3.2.1). It should be mentioned that for copper and zinc, the concentrations in the dissolved fraction of storm water did not only exceed the Annual Average (AA-)EQS, but also the Maximum Allowable Concentration (MAC-)EQS. Furthermore, for several compounds, the concentrations were higher in the particulate phase that in the dissolved fraction (see Appendix).

According to Skarbøvik et al. (2017; The Norwegian Environment Agency M-862), River Alna also brought some contaminants to the fjord: 8-10 g/yr PCB7, 3-3.6 g/yr ΣPBDE (excl. BDE-28), 0.21 kg/yr SCCPs, 0.26 kg/yr MCCPs, 61 g/yr bisphenol A, 0.16-0.24 g/yr TBBPA and 118 g/yr PFOS in 2016. Furthermore, the annual mean concentration of Cu, Zn, As, Cr, Cd, Pb, Ni and Hg in the river water was 2.95 µg/L, 9.99 µg/L, 0.37 µg/L, 0.41 µg/L, 0.03 µg/L, 0.46 µg/L, 0.90 µg/L and 1.33 µg/L respectively. As such, there are several pathways of these contaminants to the Inner Oslofjord.



	Particles	Water
PFPA	n.d.	9.08
PFHxA	n.d.	5.44
PFHpA	n.d.	3.35
PFOA	n.d.	5.91
PFNA	n.d.	2.75
PFDA	n.d.	2.05
PFUdA	n.d.	0.50
PFBS	n.d.	1.41
PFOS	n.d.	7.38
6:2 FTS	n.d.	3.69

Figure 28. Relative contribution (%) of PFAS compounds to the sum of (detected) PFASs in the particulate and dissolved fraction of storm water (mean of 2 samples. Non-detected components were assigned values of zero). Concentrations (ng/L; mean; non-detected components were assigned a value of zero) of detected components are given in the associated table. Components that were not detected in any replicate samples of a fraction (particles or water) are noted n.d.

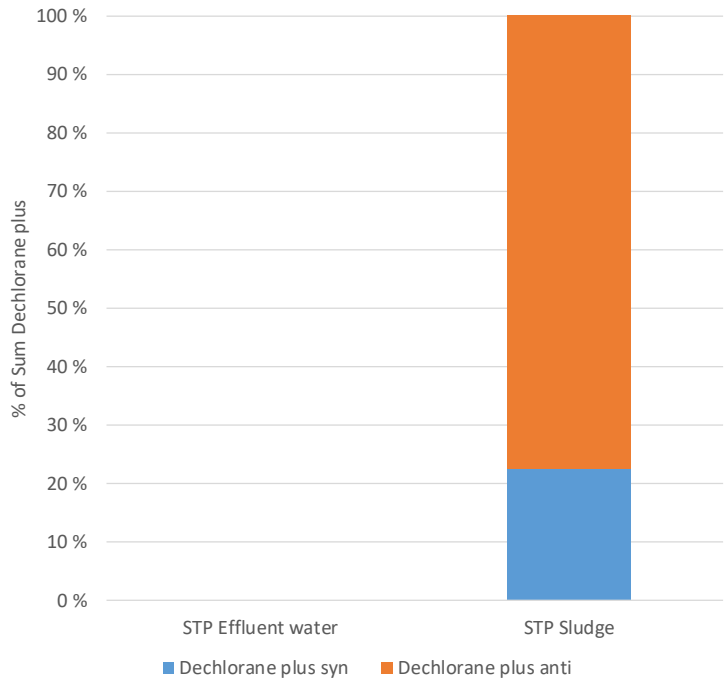
Table 15.
Concentrations of contaminants (µg/L) in storm water (dissolved fraction) and STP effluent water of which Norwegian quality standards (from the Norwegian Environment Agency; M-608) exist in coastal water. Red numbers indicate excess of the quality standard.

River basin specific compounds	AA-EQS (µg/L)	Storm water conc. (dissolved; µg/L)	Effluent water (STP) conc. (µg/L),
Bisphenol A	0.15	1.047	<0.11
Decamethylcyclopentasiloxane (D5)	0.17	n.a.	0.68
Medium chained chloroparafins (MCCPs)	0.05	0.0685	0.08
Copper (Cu)	2.6	9.0	n.a.
PCB7	0.0000024	<0.0016****	n.a.
PFOA	9.1	0.006	0.006
Zinc (Zn)	3.38	50.4	n.a.
TBBPA	0.254	<0.025	<0.015
Arsenic (As)	0.6	0.4	n.a.
Chromium (Cr)	3.4	1.0	n.a.
EU priority substances			
Cadmium (Cd)	0.2	0.1	n.a.
Lead (Pb)	1.3	0.4	n.a.
Nickel (Ni)	8.6	1.5	n.a.
Mercury (Hg)	0.07 ***	<0.002	n.a.
Brominated diphenyl ethers *	0.014 ***	<0.0013	<0.0013
Hexachlorobenzene	0.05 ***	<0.00067	n.a.
C10-13 chloroalkanes **	0.4	0.06	0.07
Pentachlorobenzene	0.0007	<0.00038	n.a.
Nonylphenol (4-)	0.3	<0.025	<0.015
Oktylphenol (4-*tert*-)	0.01	<1.6 ***	<0.95 ***
PFOS	0.00013	0.0074	0.0020

* Sum of BDE-28, -47, -99, -100, -153 and -154.
** Short chained chloroparaffins (SCCPs)
*** No AA-EQS for these substances, thus this is the MAC-EQS (M-608)
**** Too high limit of detection to evaluate

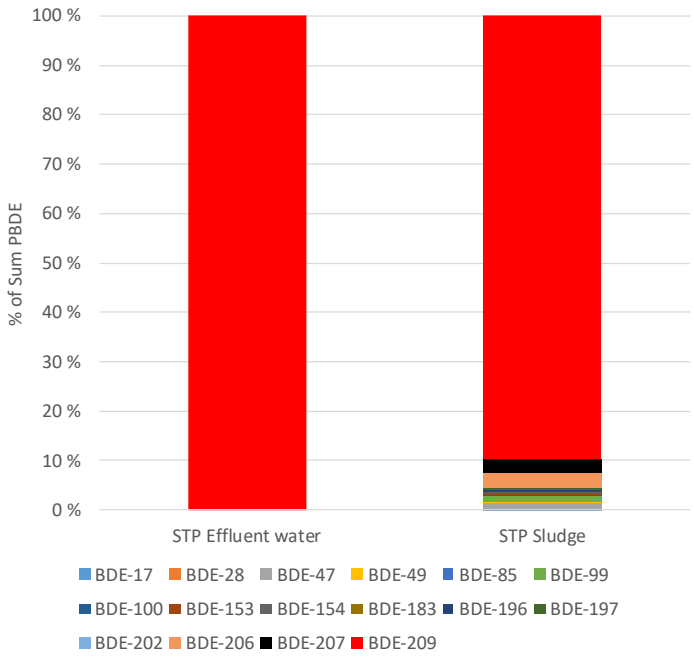
3.2.7 Sewage treatment plant (STP)

The results of the chemical analyses of effluent water and sludge from Bekkelaget STP can be found in the electronic Appendix. Dechlorane plus was found in the sludge (mean concentration, sum of *syn* and *anti* isomers, n=2, 9.5 ng/g dry wt.; Figure 29). Of the PBDEs, only BDE-209 was detected in the effluent water from Bekkelaget sewage treatment plant (Figure 30). This congener also showed, by far, the highest concentration in the sludge (Figure 30). Given the hydrophobic nature of these compounds, they have a high affinity for the particulate phase. Finding BDE-209 in the highest concentrations in sludge corresponds with other recent findings (Aigars et al. 2017) and with the historic market demand for deca-BDE mixtures (McGrath et al. 2017). As the main component of these mixtures, BDE-209 has been the most prevalent congener in a large majority of soil samples (McGrath et al. 2017). Another conspicuous result of the sludge analysis was that the alternative/"new" brominated flame retardants TBPH (BEH/TBP) and DBDPE were found in high concentrations (mean concentration, n=2, 140.5 and 113.6 ng/g dry wt., respectively; Figure 31).



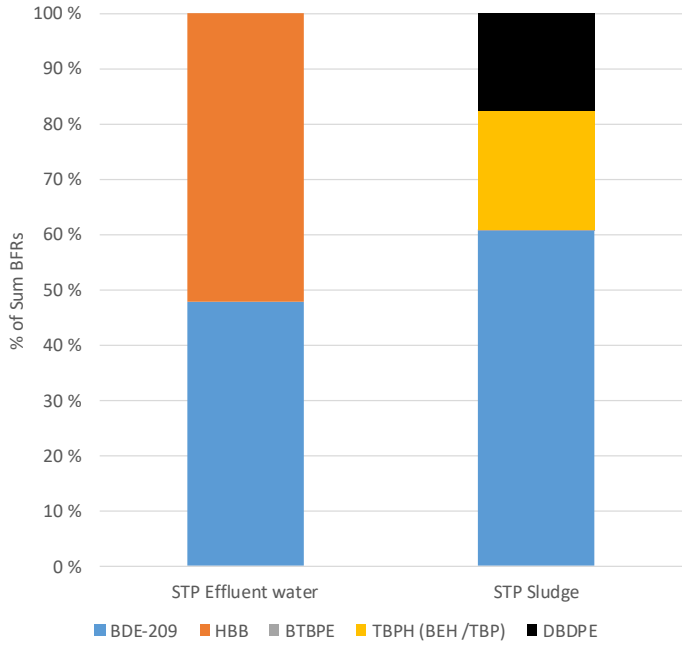
	STP Effluent water (ng/L)	STP Sludge (ng/g)
Dechlorane plus syn	n.d.	2.140
Dechlorane plus anti	n.d.	7.404

Figure 29. Relative contribution (%) of dechlorane plus syn and anti isomers to the sum of dechlorane plus in effluent water and sludge from a sewage treatment plant in the Inner Oslofjord (mean of 2 samples). Concentrations (ng/L or ng/g; mean) are given in the associated table. Components that were not detected in any replicate samples of a fraction (effluent water or sludge) are noted n.d. Dechlorane plus syn and anti were the only dechlorane compounds detected in STP water or sludge.



	Effluent water (ng/L)	Sludge (ng/g)
BDE-17	n.d.	0.157
BDE-28	n.d.	0.136
BDE-47	n.d.	5.615
BDE-49	n.d.	1.220
BDE-85	n.d.	0.108
BDE-99	n.d.	5.080
BDE-100	n.d.	1.210
BDE-153	n.d.	0.711
BDE-154	n.d.	0.433
BDE-183	n.d.	0.536
BDE-196	n.d.	2.040
BDE-197	n.d.	0.841
BDE-202	n.d.	0.796
BDE-206	n.d.	14.0
BDE-207	n.d.	13.0
BDE-209	0.550	453.0

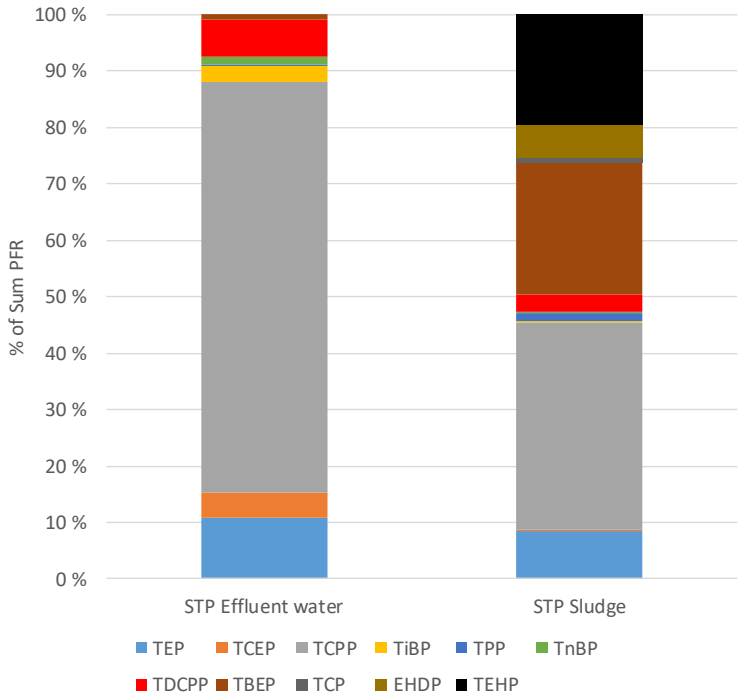
Figure 30. Relative contribution (%) of BDE-congeners to the sum of (detected) PBDEs in effluent water and sludge from a sewage treatment plant in the Inner Oslofjord (mean of 2 samples. Non-detected components were assigned values of zero). Concentrations (ng/L or ng/g; mean; non-detected components were assigned a value of zero) of detected components are given in the associated table. Components that were not detected in any replicate samples of a fraction (effluent water or sludge) are noted n.d.



	Effluent water (ng/L)	Sludge (ng/g)
BDE-209	0.550	453.0
HBB	0.043	0.130
BTBPE	n.d.	0.908
TBPH (BEH/TBP)	n.d.	140.5
DBDPE	n.d.	113.6

Figure 31. Relative contribution (%) of Brominated flame retardants (BFRs) to the sum of (detected) BFRs in effluent water and sludge from a sewage treatment plant in the Inner Oslofjord (mean of 2 samples. Non-detected components were assigned values of zero). Concentrations (ng/L or ng/g; mean; non-detected components were assigned a value of zero) of detected components are given in the associated table. Components that were not detected in any replicate samples of a fraction (effluent water or sludge) are noted n.d. PBDEs are represented by BDE-209, the congener displaying the highest concentrations (see Figure 30).

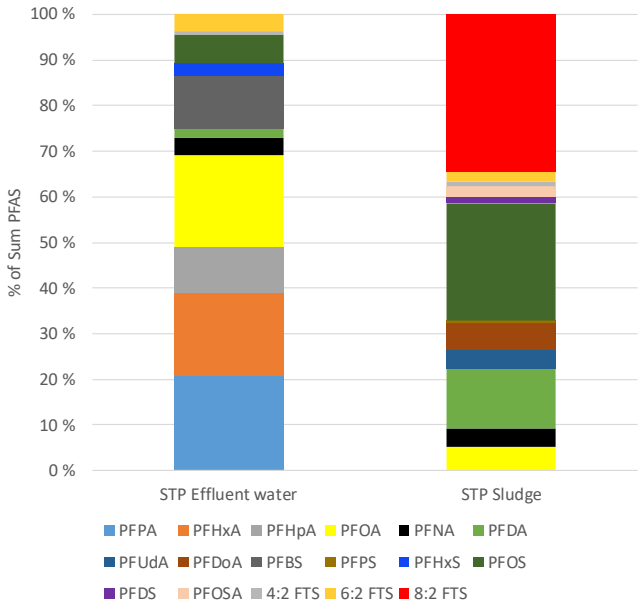
PFR compounds were present in both effluent water and sludge from Bekkelaget sewage treatment plant (Figure 32). TCPP was found in the highest concentration in both fractions (Figure 32). TBEP was found in the second highest concentration in the sludge (Figure 32). This corresponds with findings in storm water in the Urban fjord programme in 2016 (Ruus et al. 2017; The Norwegian Environment Agency M-812), when TCPP and TBEP were found in the highest concentrations (in both the dissolved and the particulate fraction).



	Effluent water (ng/L)	Sludge (ng/g)
TEP	298.90	282.21
TCEP	123.59	4.00
TCPP	1997.08	1226.95
TiBP	82.91	5.56
TPP	0.96	46.28
TnBP	44.21	5.57
TDCPP	181.08	107.88
TBEP	20.89	768.27
TCP	n.d.	38.24
EHDP	n.d.	188.24
TEHP	n.d.	649.46

Figure 32. Relative contribution (%) of PFR compounds to the sum of (detected) PFRs in effluent water and sludge from a sewage treatment plant in the Inner Oslofjord (Non-detected components were assigned values of zero). Concentrations (ng/L or ng/g) of detected components are given in the associated table. Components that were not detected in a fraction (effluent water or sludge) are noted n.d.

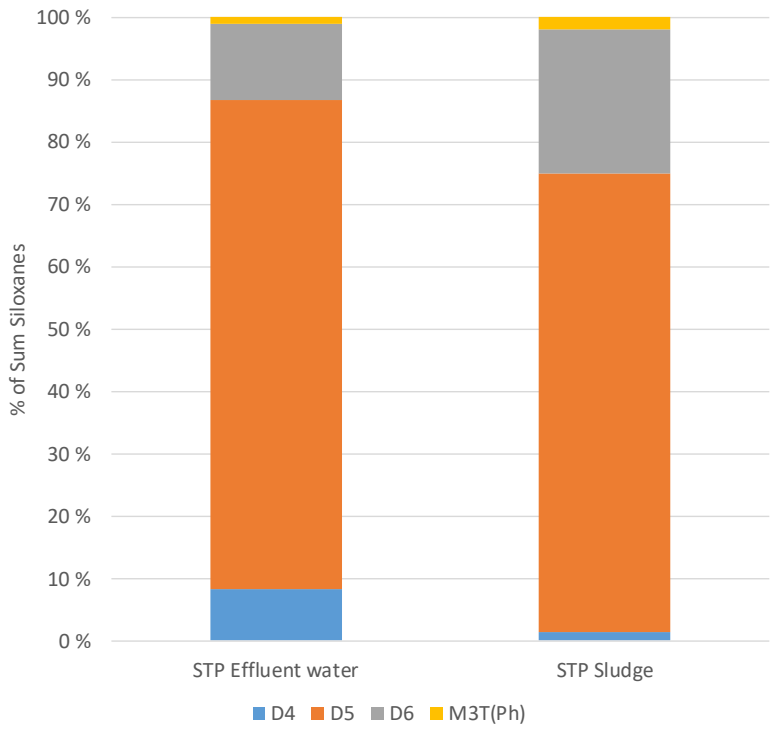
A number of PFAS compounds were detected in both effluent water and sludge from Bekkelaget sewage treatment plant (Figure 33).



	Effluent water (ng/L)	Sludge (ng/g)
PFPA	6.55	n.d.
PFHxA	5.79	n.d.
PFHpA	3.12	n.d.
PFOA	6.43	1.05
PFNA	1.18	0.79
PFDA	0.58	2.68
PFUdA	n.d.	0.88
PFDoA	n.d.	1.18
PFBS	3.69	n.d.
PFPS	n.d.	0.10
PFHxS	0.85	n.d.
PFOS	1.96	5.26
PFDS	n.d.	0.32
PFOSA	n.d.	0.44
4:2 FTS	0.27	0.17
6:2 FTS	1.04	0.50
8:2 FTS	n.d.	7.00

Figure 33. Relative contribution (%) of PFAS compounds to the sum of (detected) PFASs in effluent water and sludge from a sewage treatment plant in the Inner Oslofjord (mean of 2 samples. Non-detected components were assigned values of zero). Concentrations (ng/L or ng/g; mean; non-detected components were assigned a value of zero) of detected components are given in the associated table. Components that were not detected in any replicate samples of a fraction (effluent water or sludge) are noted n.d.

Siloxanes were detected in both effluent water and sludge from Bekkelaget sewage treatment plant (Figure 34). As in the other matrices analysed in this programme, D5 was present in the highest concentrations in both effluent water and sludge (Figure 34). The concentrations of D5 in effluent water from Bekkelaget STP were higher than previously observed in effluent water from HIAS STP (Ottestad, on Lake Mjøsa; mean 99 ng/L) and Rambekk STP (Gjøvik, on lake Mjøsa; mean 31 ng/L; van Bavel et al. 2016; The Norwegian Environment Agency M-596). Concentrations in sludge, on the other hand were lower than in sludge from HIAS STP (mean 7900 ng/g) and Rambekk STP (mean 6059 ng/g; van Bavel et al. 2016; The Norwegian Environment Agency M-596). M3T(Ph) was not detected in effluent water from HIAS and Rambekk STPs, while concentrations in sludge (mean 93 and 62 ng/g, respectively; van Bavel et al. 2016; The Norwegian Environment Agency M-596) appeared higher than in the present study.



	Effluent water (ng/L)	Sludge (ng/g)
D4	17.25	17.15
D5	677.31	960.66
D6	197.43	303.35
M3T(Ph)	33.31	24.85

Figure 34. Relative contribution (%) of siloxanes to the sum of siloxanes in effluent water and sludge from a sewage treatment plant in the Inner Oslofjord (mean of 2 samples for effluent water). Concentrations (ng/L or ng/g) of components are given in the associated table.

UV-chemicals (benzophenone, ethylhexylmethoxycinnamate and especially octocrylene) were detected in notable concentrations in samles from Bekkelaget sewage treatment plant, and especially sludge (see electronic appendix), reflecting their use in sunscreens and other cosmetics.

The antioxidant MB1 was not detected in neither STP effluent water (<15 ng/L), nor sludge (<5 ng/g). Previously concentrations of 25 to ~130 ng/L were observed in effluent water from HIAS STP (Ottestad, on Lake Mjøsa) and Rambekk STP (Gjøvik, on lake Mjøsa; van Bavel et al. 2016; The Norwegian Environment Agency M-596).

The last annual report from VEAS sewage treatment plant (STP) is from 2016 and they reported a discharge of 45 kg As, 49 kg Pb, 5.0 kg Cd, 552 kg Cu, 52 kg Cr, 0.33 kg Hg, 236 kg Ni and 1933 kg Zn that year (more than 90% of the measurements were below the limit of detection for Cd, Cr and Hg, and half of the LoD was reported for these; VEAS 2017).

As such, effluent water from the sewer of the population in the urban environment of Oslo is also a pathway of several compounds to the Inner Oslofjord marine environment.

As mentioned, for several compounds, environmental quality standards (EQS) for water are given through Norwegian law (The Water Regulation/"Vannforskriften"), according to the requirements of the Water Framework Directive. Furthermore, quality standards are given for even more compounds (The Norwegian Environment Agency M-608). For the target compounds of this study of which quality standards exist, the concentrations in effluent water from Bekkelaget STP and the quality standards are also compared in Table 15 (quality standards for coastal water used, to elucidate the potential of surface water as source of contaminants to parts of the fjord). D5, MCCPs and PFOS exceeded AA-EQS.

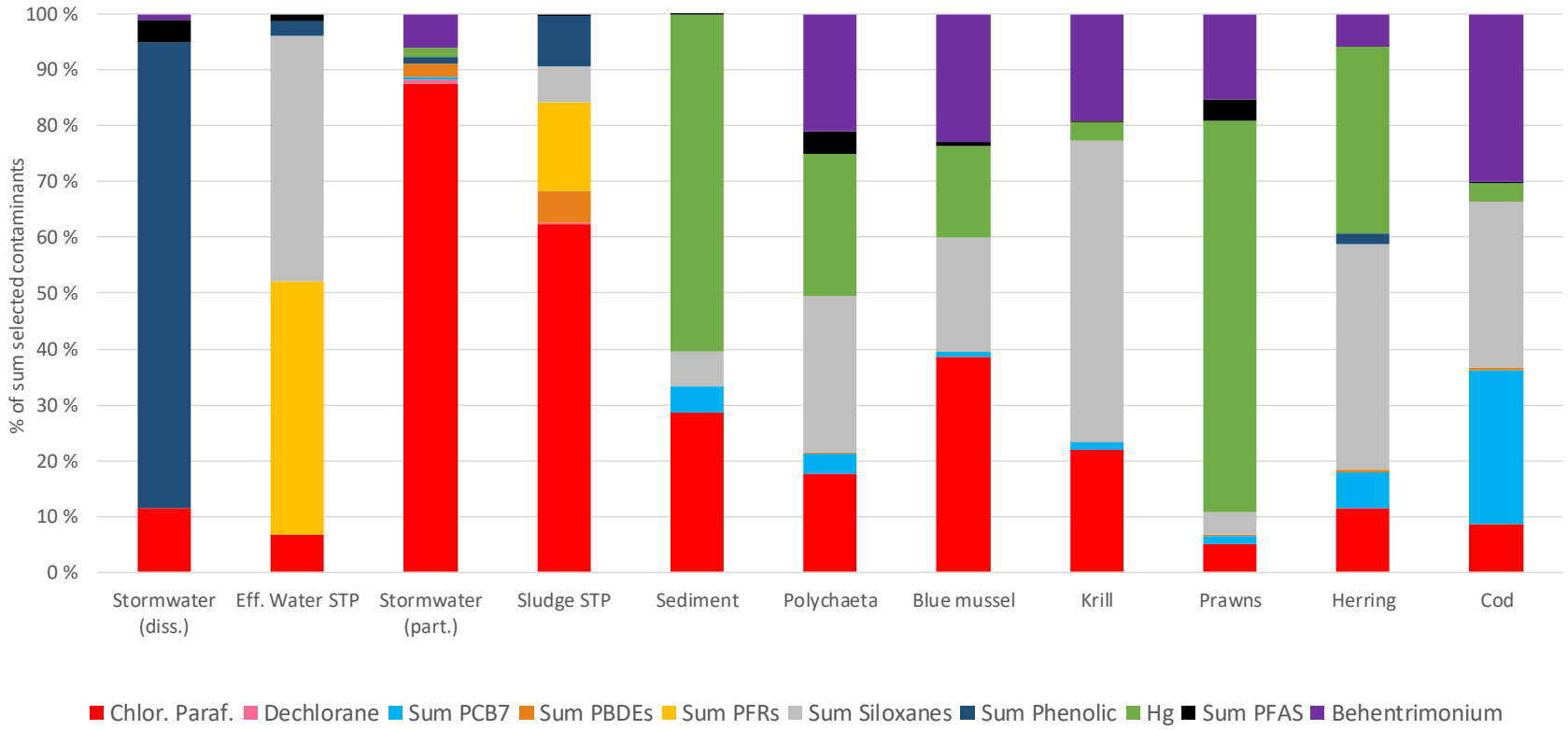
3.3 Interspecies and matrix comparisons

In terms of sources and sinks of contaminants in the marine ecosystem of the Inner Oslofjord, it is of interest to give general impression of the dominating contaminants/groups of contaminants in the different species and matrices analysed. Figure 35 shows relative contribution of selected contaminants/groups of contaminants to the sum of these contaminants/groups of contaminants in storm water (dissolved and particulate fractions) entering the Oslofjord, sediments of the Inner Oslofjord, and polychaetes, blue mussel, krill, prawns, herring and cod (liver) from the Inner Oslofjord, as well as in effluent water (entering the Oslofjord) and sludge from Bekkelaget STP. The selected contaminants were chlorinated paraffins (sum of SCCPs and MCCPs), dechlorane compounds, sum PCB7, sum PBDEs, sum PFRs, sum siloxanes, sum phenolic compounds, Hg, sum PFAS compounds and behentrimonium (sum of ATAC-C20 and -C22; See Table 5 for specifics regarding the constituents of the sums of contaminant groups).

Chlorinated paraffins apparently constitute major proportions in all species/matrices, especially in the particulate fraction of stormwater and sludge from the sewage treatment plant, as well as in mussels (Figure 35). PCBs and PBDEs do not constitute very high (<5 %) proportions of the sum of contaminants, except for PCBs in the lipid rich tissues herring muscle and cod liver (PCBs were not analysed in samples from the STP; Figure 35). PBDEs

constituted ~6% of the sum of the selected contaminants in sludge from the sewage treatment plant (Figure 35). PFRs were only analysed in samples from the STP where they apparently constituted a major proportion, especially in the effluent water (Figure 35). Siloxanes (not analysed in storm water) constituted major proportions of the sum of contaminants in effluent water from the STP, as well as in organisms in the Inner Oslofjord marine food web. As in 2016 (Ruus et al. 2017; The Norwegian Environment Agency M-812), siloxanes were the major constituents of the sum of contaminants in krill (Figure 35). Phenolic compounds constituted major proportions of the sum of contaminants in storm water (the dissolved fraction), and to some degree in sludge from the STP (Figure 35). Hg (not analysed in samples from the STP) constituted major proportions of the sum of contaminants in sediments and organisms from the Inner Oslofjord, especially in prawns (Figure 35). PFAS compounds were only notable constituents of the sum of contaminants in the dissolved phase of storm water, as well as in polychaetes and prawns (Figure 35). Behentrimonium (not analysed in sediment or samples from the STP) apparently constitute major proportions in organisms of the Inner Oslofjord marine food web (Figure 35).

A.


100 %
90 %
80 %
70 %
60 %
50 %
40 %
30 %
20 %
10 %
0 %
% of sum selected contaminants
Stormwater (diss.)
Eff. Water STP
Stormwater (part.)
Sludge STP
Sediment
Polychaeta
Blue mussel
Krill
Prawns
Herring
Cod
Chlor. Paraf.
Dechlorane
Sum PCB7
Sum PBDEs
Sum PFRs
Sum Siloxanes
Sum Phenolic
Hg
Sum PFAS
Behentrimonium

B.

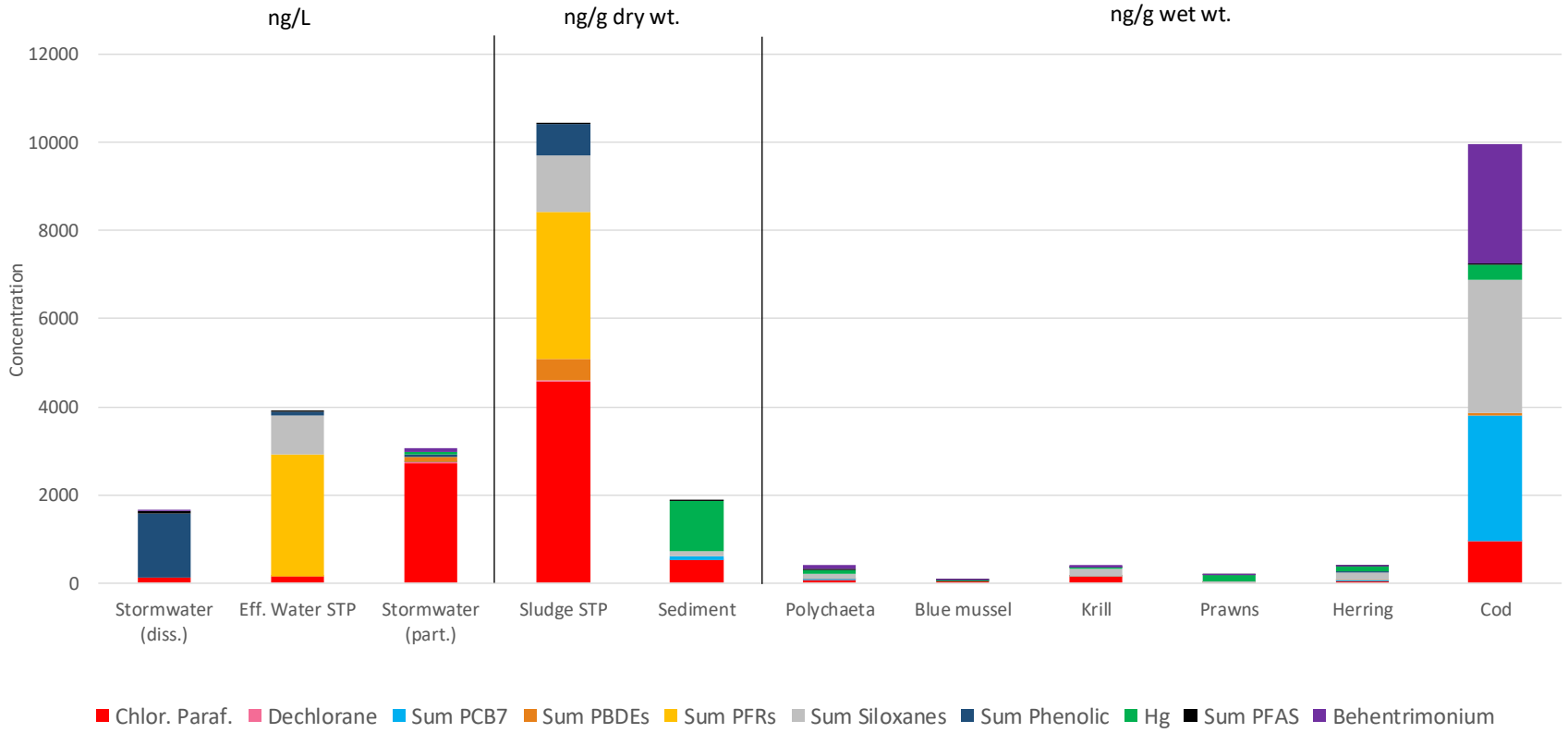


Figure 35. Relative contribution of selected contaminants/groups of contaminants to the sum of these contaminants/groups of contaminants (A.), as well as concentrations (B.), in storm-water (dissolved and particulate fractions) entering the Oslofjord, sediments of the Inner Oslofjord, and polychaetes, blue mussel, krill, prawns, herring and cod from the Inner Oslofjord, as well as in effluent water (entering the Oslofjord) and sludge from Bekkelaget STP. Note that Behentrimonium was not analysed in sediment or samples form the STP, PFRs were only analysed in samples from the STP, siloxanes were not analysed in storm water, and PCBs and Hg were not analysed in samples from the STP. Note: Dechlorane is dechlorane plus (syn and anti isomers), except in cod where dechlorane 602 and 603 also were detected).

3.4 Support parameters

Miscellaneous support parameters were measured for the different matrices/samples/organisms: Particle fraction <63 µm (% dry wt.) and TOC (µg/mg dry wt.) in sediment,DOC (mg C/L) and suspended solids (mg/L) in effluent water from Bekkelaget STP, TOC (µg/mg dry wt) in sludge from Bekkelaget STP, $\delta^{13}C$, $\delta^{15}N$, C:N (W%), trophic position (deduced from $\delta^{15}N$,) weight of egg (g) and eggshell thickness (mm) for herring gull eggs from the Inner and Outer Oslofjord and in eider duck eggs from the Inner Oslofjord (not eggshell thickness), $\delta^{13}C$, $\delta^{15}N$, C:N (W%), trophic position (deduced from $\delta^{15}N$), wing length (mm), head length (mm) and body mass (g) for herring gull (blood) from the Inner and Outer Oslofjord and in eider duck (blood) from the Inner Oslofjord, $\delta^{13}C$, $\delta^{15}N$, C:N (W%), trophic position (deduced from $\delta^{15}N$), age (yr), body length (cm), body mass (g), liver weight (g), gonad weight (g) and sex of cod from the Inner Oslofjord, and $\delta^{13}C$, $\delta^{15}N$, C:N (W%) and trophic position (deduced from $\delta^{15}N$) of the organisms of the Inner Oslofjord food web. The measurements of these support parameters are presented in Tables A1-A10 in the Appendix. The lipid content of all biological samples is given in the electronic Appendix.

3.5 Biological effect parameters

Acetylcholinesterase (AChE) activity in muscle (microsomal fraction) was measured in cod, as well as the physiological parameters liversomatic index (LSI) and gonadosomatic index (GSI). These parameters are presented in Table 16.

In vertebrates acetylcholine (ACh) acts as an excitatory transmitter in the somatic nervous system. ACh also serves as both a pre ganglionic and a post ganglionic transmitter in the parasympathetic nervous system. Cholinesterase enzymes (ChE) are responsible for the removal of ACh from the synaptic cleft by hydroxylation. AChE may be inhibited by various substances/contaminants in the aquatic environment, such as organophosphates (Burgeot et al., 2012; Assis et al. 2010; Di Tuoro et al., 2011).

Table 16.
Biological effect parameters measured for cod from the Inner Oslofjord.

Sample no.	Sex	AChE *	GSI	LSI
1	M	11.12	0.83	2.31
16	F	16.41	0.96	3.96
3	F	10.76	1.12	8.85
4	F	9.82	4.12	3.31
5	M	8.69	0.85	5.26
6	F	11.81	6.28	2.43
7	F	7.78	0.86	2.46
8	F	13.39	0.46	3.00
9	M	11.00	0.27	8.28
10	M	10.93	0.45	5.71
11	M	11.81	0.31	3.07
12	F	15.67	2.69	4.07
13	M	10.79	0.38	6.54
14	F	13.02	3.80	2.54
15	M	12.75	0.10	3.79

***Acetylcholinesterase activity (nmol ATC/min/mg protein)**

In the 2015 "Urban fjord" programme, a statistically significant negative relationship (log-log) was observed between the concentration of Hg (analysed in muscle) and AChE in cod (Ruus et al. 2016; The Norwegian Environment Agency M-601). This finding was interesting, since inhibition of AChE is a known marker of exposure to organophosphate pesticides, but the role of Hg as an anticholinesterase agent is not as well established. Shaw and Panigrahi (1990) did however show a significant negative correlation between brain residual Hg levels and AChE activity in fish. They suggested that Hg might be exerting its influence by combining with the SH-group of the enzyme leading to conformational changes and thus inactivation. Vieira et al. (2009) also found that Hg inhibited AChE activity in the head of the common goby (*Pomatoschistus microps*), also leading to decreased swimming performance. However, in 2015, AChE activity in the muscle of cod also showed statistically significant negative relationships with length, weight and age of cod (Ruus et al. 2016; The Norwegian Environment Agency M-601), and since Hg was shown to correlate with length and weight of cod, the results were inconclusive regarding likely causality (Ruus et al. 2016; The Norwegian Environment Agency M-601). In 2016, AChE activity did not show a statistically significant negative relationship with the length of cod, or between AChE activity and Hg liver concentrations (Ruus et al. 2017; The Norwegian Environment Agency M-812).

In 2017, there was a significant negative relationship between AChE-activity and the length of cod (Figure 36). There was, however, no significant relationship between AChE-activity and muscle Hg-concentration (Figure 37). Note also that there was no statistically significant relationship (p=0.056) between mercury in cod and the length of cod (Figure 20). As such, it is possible that the negative relationship between AChE-activity and the length of cod may be a result of lower AChE:muscle protein-ratio in larger cod, while the possible inhibition of AChE by Hg is still inconclusive.

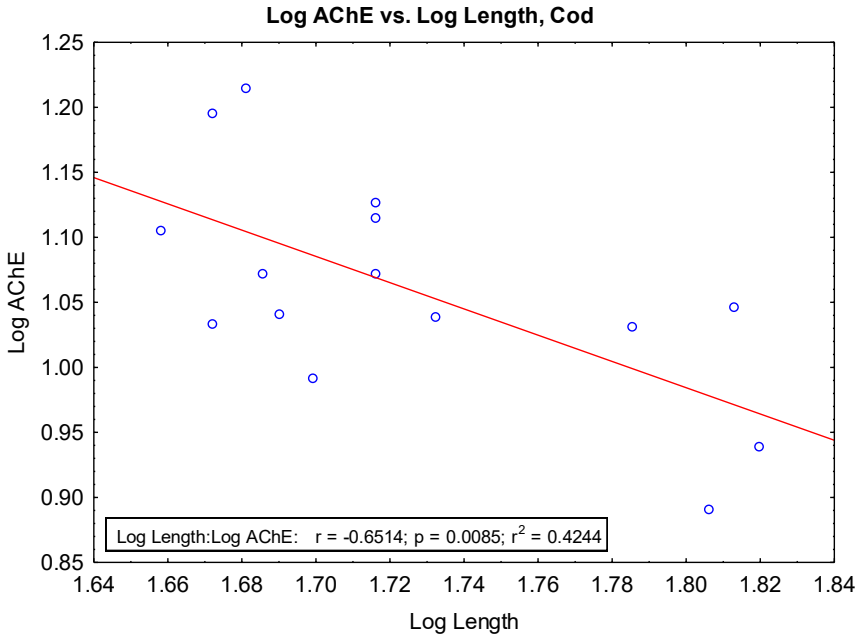


Figure 36. Acetylcholinesterase (AChE) activity (nmol ATC/min/mg protein; log-transformed) in muscle of cod from the Inner Oslofjord against length (cm: log-transformed) of cod.

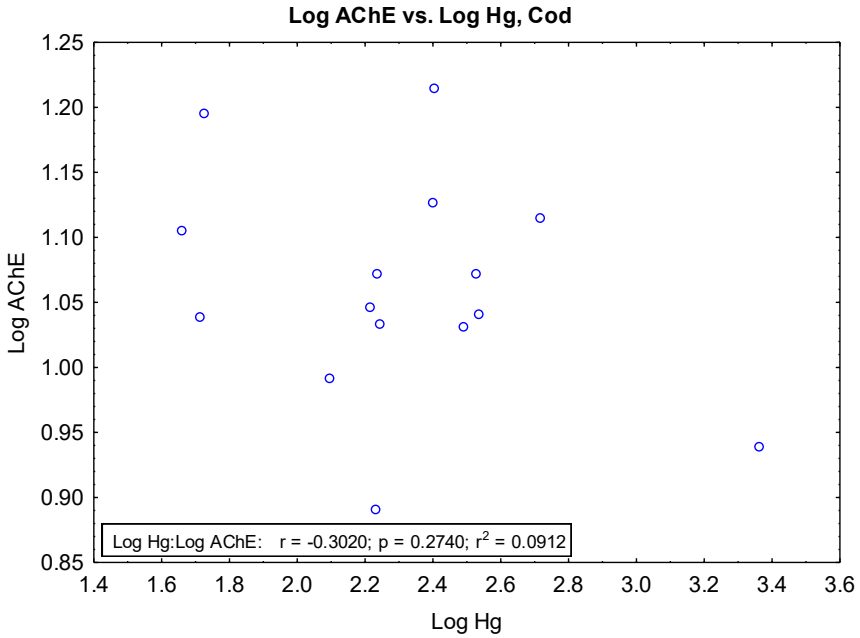


Figure 37. Acetylcholinesterase (AchE) activity (nmol ATC/min/mg protein; log-transformed) in muscle of cod from the Inner Oslofjord against Hg-concentration (ng/g wet wt.; log-transformed in muscle of cod).

3.6 Mixture toxicity / cumulative risk

Of the measured contaminants, $PNEC_{pred,}$ $PNEC_{oral}$ and/or EQS_{biota} values were only found for 27 compounds or compound groups (Table 17). All values ($PNEC_{pred}$, $PNEC_{oral}$ and EQS_{biota}) are hereby referred to as $PNEC_{pred}$ and refer to secondary poisoning of terrestrial organisms from eating contaminated prey. The risk of secondary poisoning of seabirds feeding on blue mussels, polychaetes or herring was calculated by summing up the $MEC/PNEC_{pred}$ values as described earlier and is presented in the following subchapters.

Table 17.
Available PNEC values for the analysed contaminants (µg/kg).

Compound	$PNEC_{pred}$[a]	$PNEC_{oral}$	QS_{biota}[b]	Value used
Bisphenol A	2670			2670
Cadmium (Cd)		160[c]		160
Decamethylcyclopentasiloxane (D5)	13000		15217	13000
Hexachlorobenzene (HCB)			10	10
Dodecamethylcyclohexasiloxane (D6)		667000[g]		667000
Sum DDT (50-29-3, 789-02-6, 72-55-9, 72-54-8)			609	609
Lead (Pb)	3600			3600
Medium chained chloroparafins (MCCP)	10000		170	10000
Mercury (Hg)	400		20	400
Nickel (Ni)	8500[d]			8500
Nonylphenol (4-)	10000		3000	10000
OctaBDE (BDE183, 184, 191, 196, 197, 202, 206, 207)	6700			6700
Octamethylcyclotetrasiloxane (D4)		1700[f]		1700
Octylphenols (octylphenol and 4-tert-octylphenol)	10000		0.004 (4t only)	10000
PeCB			50	50
PCBs (sum 7 PCBs)			1	1
PentaBDE (BDE-99 + BDE-100)	1000			1000
DecaBDE (BDE-209)	833000			833000
Sum PBDE (BDE-28, -47, -99, -100, -153, -154)			0.0085	0.0085
PFOA			91.3	91.3
PFOS	13		9.1	13
Short chained chloroparafins (SCCP)	5500		6000	5500
TCEP			7304	7304
TCP	1700			1700
TCPP	11600			11600
Tetrabromobisphenol A	667000			667000
[a]Obtained from Andersen et al. (2012) [b] M-608 and EQS directive 2013/39/EU [c]EU RAR Cd 2007 [d]EU RAR Ni 2008 [e]ECHA 2015, [f]Brooke et al., 2009b. [g]Brooke et al., 2009a				

3.6.1 Risk of secondary poisoning for predators of blue mussels

The sum of MEC/$PNEC_{pred}$ values based on measured concentrations in blue mussels was 6.33 (Table 18) which is indicative of a risk to predators of these organisms. The main risk drivers for secondary poisoning of seabirds feeding on blue mussels are the sum of PBDEs (MEC/$PNEC_{pred}$ = 3.77), Cd (MEC/$PNEC_{pred}$ = 1.33), and sum of 7 PCBs (MEC/$PNEC_{pred}$ = 0.74), constituting 92% of the total sum of MEC/$PNEC_{pred}$ (Figure 38). Sum PBDEs and Cd had a MEC/PNEC ratio above 1 indicating that they constitute a risk by themselves. Ten of the detected compounds (PFBS, PFOSA, TBA, Cr, Fe, Cu, Zn, As, Ag, and Sb) were not included in the calculations due to a lack of $PNEC_{pred}$ values potentially leading to an underestimation of the risk. On the other hand, the risk contribution of the main risk drivers (sum PBDE and Sum PCB7) are calculated by the use of $QS_{biota,hh}$ values which are more conservative than $PNEC_{pred}$, $PNEC_{oral}$ and $QS_{biota,\ secpois}$ values, potentially leading to an overestimation of the risk.

Table 18.
Calculation of MEC/PNECpred ratios for blue mussels.

Compound	$MEC_{average}$ (µg/kg)	MEC/PNEC
Sum PBDE (BDE-28, -47, -99, -100, -153, -154) [a]	0.0320	3.77
Cd	210	1.33
Sum 7 PCB [a]	0.74	0.74
Ni	2750	0.32
Pb	460	0.13
Hg	13	0.03
SCCP	21	$3.8E^{-3}$
D5	17	$1.3E^{-3}$
MCCP	10	$1.0E^{-3}$
Sum MEC/PNEC		6.33

[a] MEC/PNEC values calculated based on $QS_{biota,hh}$ values

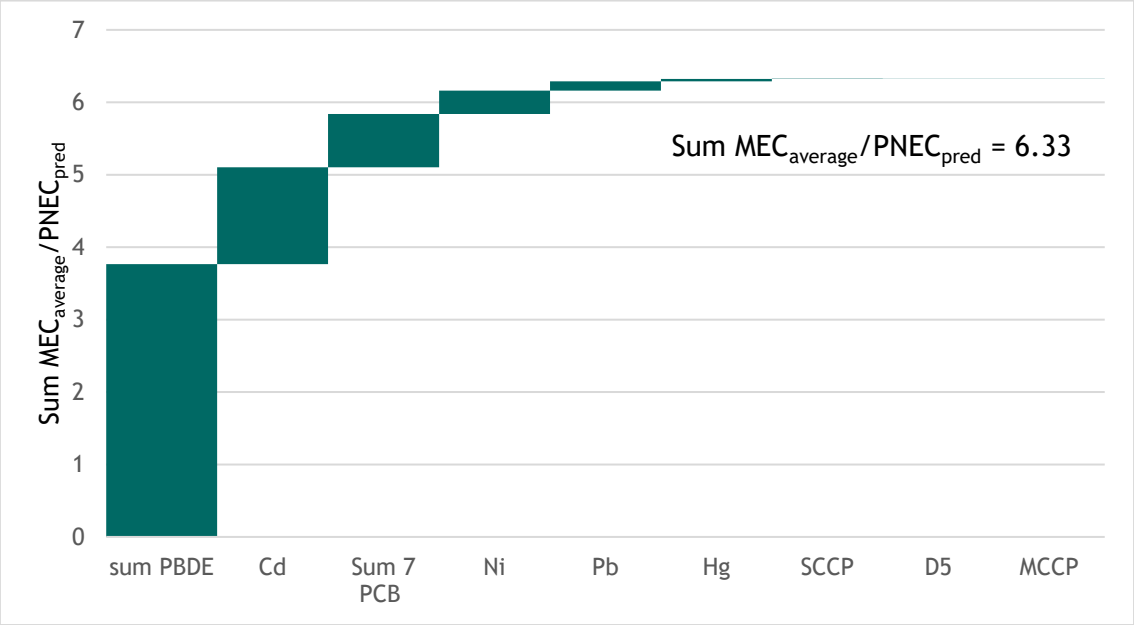


Figure 38. Contribution plot of MEC/PNEC$_{pred}$ summation for values measured in blue mussels. Values for sum PBDE (BDE-28, -47, -99, -100, -153, -154), and sum 7 PCB were calculated based on QS$_{biota,hh}$, whereas all other values were calculated based on PNEC$_{pred}$ values.

3.6.2 Risk of secondary poisoning for predators of polychaetes

The sum of MEC/PNEC$_{pred}$ values based on measured concentrations in polychaetes was 28.79 (Table 19) which is indicative of a risk to predators of these organisms. The individual MEC/PNEC$_{pred}$ ratios are presented in Table 19. The main risk drivers for secondary poisoning of seabirds feeding on polychaetes are the sum of PBDEs (MEC/PNEC$_{pred}$ = 13.5), and sum of 7 PCBs (MEC/PNEC$_{pred}$ = 13.2), constituting 93% of the total sum of MEC/PNEC$_{pred}$ (Figure 39). Both risk drivers had a MEC/PNEC ratio above 1 indicating that they constitute a risk by themselves. Of the detected compounds in polychaetes, 24 were excluded from the cumulative risk prediction due to lack of PNEC$_{pred}$ values.

Table 19.
Calculation of MEC/PNECpred ratios for polychaetes

Compound	$MEC_{average}$ (µg/kg)	MEC/PNEC
Sum PBDE (BDE-28, -47, -99, -100, -153, -154) [a]	0.11	13.47
Sum 7 PCB [a]	13	13.22
Pb	2247	0.62
Cd	95	0.60
Ni	3216	0.38
Hg	101	0.25
PFOS	2.5	0.20
HCB [a]	0.20	0.02
SCCP	61	0.01
D5	108	$8.3E^{-3}$
PFOA	0.71	$7.7E^{-3}$
MCCP	12	$1.2E^{-3}$
PeCB [a]	0.028	$5.7E^{-4}$
D6	7.0	$1.1E^{-5}$
PBDE-209	0.11	$1.3E^{-7}$
Sum MEC/PNEC		28.79

[a] MEC/PNEC values calculated based on $QS_{biota,hh}$ values

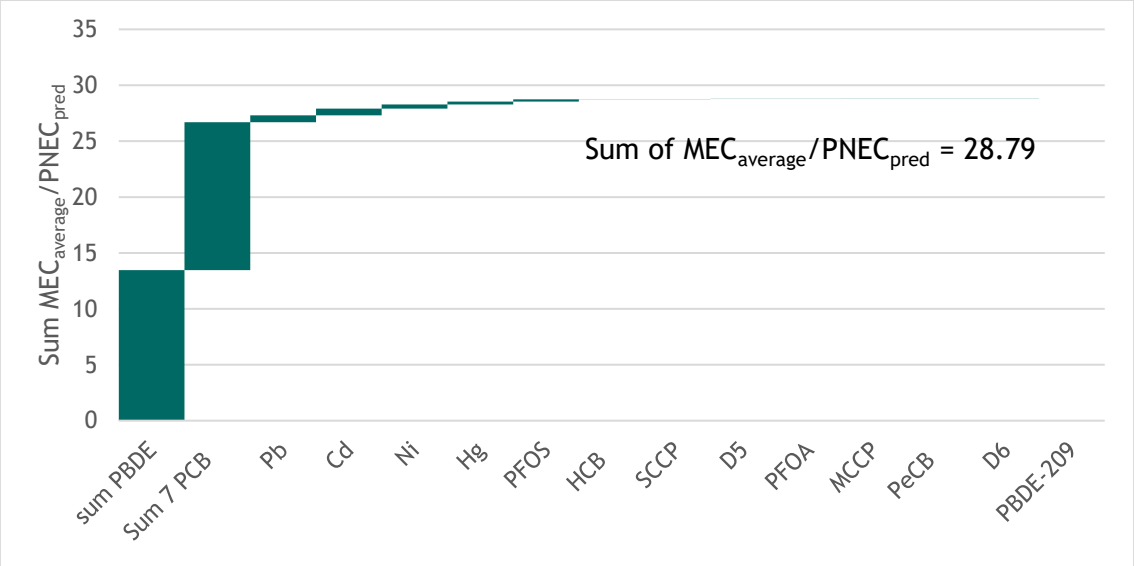


Figure 39. Contribution plot of $MEC/PNEC_{pred}$ summation for values measured in polychaetes. Values for sum PBDE (BDE-28, -47, -99, -100, -153, -154), sum PCB7, HCB and PeCB were calculated based on $QS_{biota,hh}$, whereas all other values were calculated based on $PNEC_{pred}$ values.

3.6.3 Risk of secondary poisoning for predators of herring

The sum of MEC/$PNEC_{pred}$ values based on measured concentrations in herring was 168 (Table 20) which is indicative of a risk to predators of these organisms. The individual MEC/$PNEC_{pred}$ ratios are presented in Table 20. The main risk drivers for secondary poisoning of seabirds feeding on herring are sum PBDE (MEC/$PNEC_{pred}$ = 141) and sum of 7 PCBs (MEC/$PNEC_{pred}$ =26.4), constituting 99.7% of the total sum of MEC/$PNEC_{pred}$ (Figure 40). These main risk drivers were the only compounds(group) that had a MEC/PNEC ratio above 1, indicating that they constitute a risk by themselves. Of the detected compounds in herring, 16 were excluded from the cumulative risk prediction due to lack of $PNEC_{pred}$ values.

Table 20.
Calculation of MEC/PNECpred ratios for herring

Compound	$MEC_{average}$ (µg/kg)	MEC/PNEC
Sum PBDE (BDE-28, -47, -99, -100, -153, -154) [a]	1.2	141.04
Sum PCB7 [a]	26	26.38
Hg	137	0.34
Ni	941	0.11
HCB [a]	0.64	0.06[a]
PFOS	0.17	0.01
D5	162	0.01
Cd	1.7	0.01
SCCP	30	$5.5E^{-3}$
Pb	8.00	$2.2E^{-3}$
MCCP	17	$1.7E^{-3}$
PeCB [a]	0.071	$1.4E^{-3}$
D4	2.4	$1.4E^{-33}$
D6	5.2	$7.8E^{-6}$
Sum MEC/PNEC		167.98

[a] MEC/PNEC values calculated based on $QS_{biota,hh}$ values

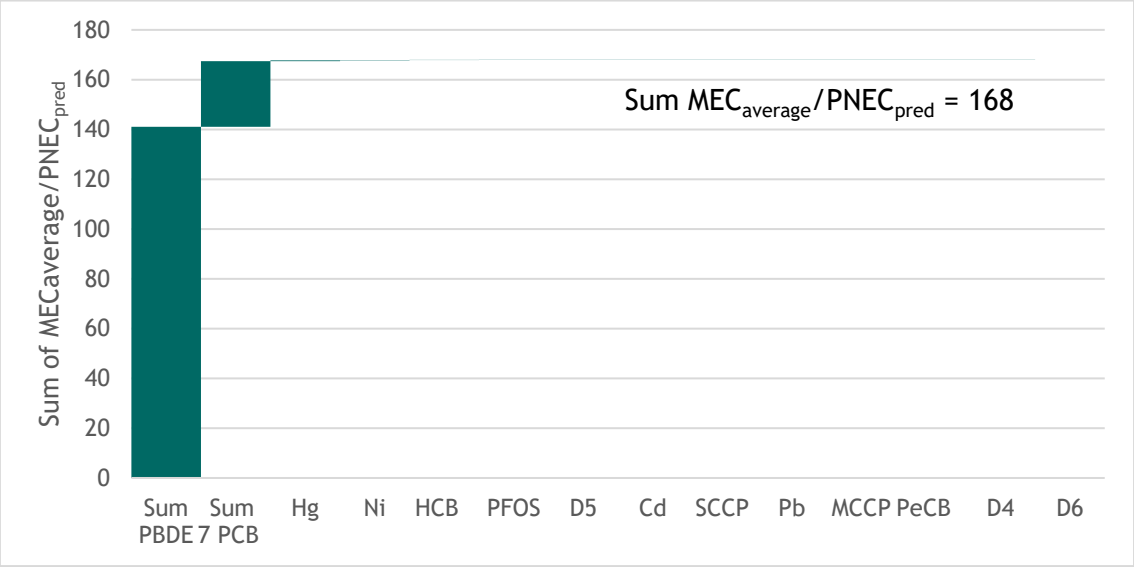


Figure 40. Contribution plot of MEC/PNEC$_{pred}$ summation for values measured in herring. Values for sum PBDE (BDE-28, -47, -99, -100, -153, -153, -154), sum PCB7, HCB, and PeCB were calculated based on QS$_{biota,hh}$, whereas all other values were calculated based on PNEC$_{pred}$ values.

For all food sources, Sum PBDE and Sum PCB were among the main risk drivers. The limit values used for these compound groups are the QS$_{biota,secpois,hh}$. As explained previously, this value has a different protection goal than PNEC$_{pred}$ values and could lead to a more conservative risk estimate for these compound groups, potentially overestimating the risk. The results should therefore be interpreted with caution. The sum of MEC$_{average}$/PNEC$_{pred}$ for all food sources are similar to last year where Sum PBDE and Sum PCB were also observed to be the main risk drivers.

For the 2016 and 2017 data, QS$_{biota}$ values were compiled alongside PNEC$_{pred}$ and PNEC$_{oral}$ values to extend the list of compounds that could be included in the cumulative risk assessment. The choice of values were made more consistent by prioritising PNEC$_{pred}$ and PNEC$_{oral}$ values over QS$_{biota\ sec\ pois}$ and QS$_{biota,hh}$ values, hence the PNEC$_{pred}$ value for Hg was chosen over the QS$_{biota}$ value used for the 2015 data. The PBDE congeners covered by PNEC$_{pred}$ for penta PBDEs overlap with the QS$_{biota}$ value for sum PBDE. As the QS$_{biota}$ for sum PBDEs covers more congeners (PBDE-28, -47, -99, -100, -153, -154) than the PNEC$_{pred}$ value for penta-PBDEs, the QS$_{biota}$ value was used when assessing the 2016 data to cover as many compounds as possible. In addition, QS$_{biota}$ for sum PCB7 which was not used for the 2015 data was included for assessing the 2016 and 2017 data.

For predators of blue mussels, Cd was the second largest contributor to the total sum of MEC/PNEC$_{pred}$ in 2017 with a value above 1, indicating that this compound also poses a risk to predators of blue mussles by itself. The risk contribution from Cd has increased slightly from 1.01 in blue mussels sampled in 2015 to 1.08 in blue mussels sampled in 2016 to 1.33 in blue mussles sampled in 2017. A different picture was observed in polychaetes where the risk contribution from Cd increased from 1.15 to 1.38 from 2015-2016, but then decreased to 0.60 in 2017.

The combination of $PNEC_{pred}$, $PNEC_{oral}$ and $QS_{biota,secpois,hh}$ limit values was performed in order to include as many compounds as possible in these assessments. The large contribution of sum PBDEs and sum PCB7 indicate that the data source from which the $PNEC_{pred}$ is based, is of importance, and the combination of $PNEC_{pred}$ and $QS_{biota,secpois,hh}$ add some uncertainty to the estimates. In addition, no grouping of chemicals based on their mode of action or adverse effects were performed, potentially contributing to an overestimation of the risk. Another aspect adding uncertainty to the performed assessment is that $PNEC_{pred}$ values were only found for a limited number of compounds and compound groups (27), leading to exclusion of several detected compounds from the risk estimation, potentially contributing to an underestimation of the risk. As several aspects in the performed cumulative risk assessment can potentially lead to an over- or under-estimation of the risk, the results should be interpreted with caution and considered as a first tier screening for potential cumulative risk.

3.6.4 Risk for effects on herring gull and eider duck from exposure in eggs

The approach of summing up $MEC/PNEC_{pred}$ values is considered a conservative first-tier approach in order to filter out scenarios with low environmental risk. The calculated sum of $MEC_{average}/PNEC_{pred}$ based on blue mussels, polychaetes, or herring as food source all indicated a risk of secondary poisoning, mainly by the risk drivers sum PBDEs, sumPCB7 and Cd. In order to evaluate the risk for birds based on the measured concentrations, relevant toxicity data for the same species group with the same exposure concentration denomination (e.g. ng/g egg) as the measured concentrations is required.

In a recent study from the Norwegian Environment Agency (Andersen et al. 2014), the combined risk of effects in sea bird eggs were calculated by comparing MEC in eggs with effect data from exposure in eggs compiled from literature. These effect data were adopted in this study in order to evaluate the combined risk for effects on herring gull eggs from the Inner and Outer Oslofjord, and on eider duck eggs from the Inner Oslofjord. As the effect data does not separate between type of effect (e.g. mortality, reduced number of eggs) or effect level (e.g. LOEC, EC(D)10, EC(D)50), and assessment factors are not used in this study, the applied approach is considered as an approximation to the environmental risk assessment of chemical mixtures, tier-two. The results should therefore be interpreted with caution. The risk of combined effects of the compounds was calculated based on average (MEC_a) and median (MEC_m) values of the measured egg concentrations in 15 eggs. As seen from Tables 21, 22 and 23, using average measured concentrations led to a higher sum of MEC/Effect ratios than when using median measured concentrations. In both cases (average and median values) the sum of MEC/effect was higher than 1 in herring gull eggs, indicating a risk for effect on the eggs of the mixture of contaminants. Only the sum of MEC_a/effect was higher than 1 for eider duck eggs.

The sum MECa/effect was slightly higher in gull eggs from the Inner Oslofjord (3.07) than from the Outer Oslofjord (2.89). None of the assessed compounds had MEC/effect ratios above 1 (using average or median concentration). The common main risk drivers at both locations appear to be Sum PCBs, Cu, As (Figure 41, Figure 42). In addition, PBDE 99 is a main risk driver in eggs from the inner Oslofjord (MECa/effect = 0.68), but contribute less to the risk in eggs from the outer Oslofjord (MECa/effect = 0.16). Interestingly, PFOS had a higher MEC_a/effect value in the outer Oslofjord (0.38) than in the inner Oslofjord (0.26).

Table 21.
Calculation of MEC/effect ratios for herring gull eggs from the Inner Oslofjord

Compound	MEC_a (ng/g egg)	MEC_m (ng/g egg)	Effect value (ng/g egg)*	MEC_a/effect	MEC_m/effect
Sum PCB	300	260	400	0.751	0.651
Cu	816	814	1160	0.703	0.701
PBDE 99	6.75	1.02	10	0.675	0.102
As	54.1	35.0	180	0.300	0.194
PFOS	25.6	10.6	100	0.256	0.106
PBDE 100	1.71	0.963	10	0.171	0.0963
Hg	62.7	61.2	400	0.157	0.153
Ni	30.1	22.8	1000	0.0309	0.0228
PBDE 85	0.191	0.0107	10	0.0191	0.00107
PBDE 119	0.0394	0.0229	10	0.00394	0.00229
Cd	0.14	0.12	100	0.00139	0.00122
PBDE 126	0.00586	0.00339	10	0.00059	0.00034
Sum				3.07	2.03

*Effect values were obtained from Andersen et al. (2014)

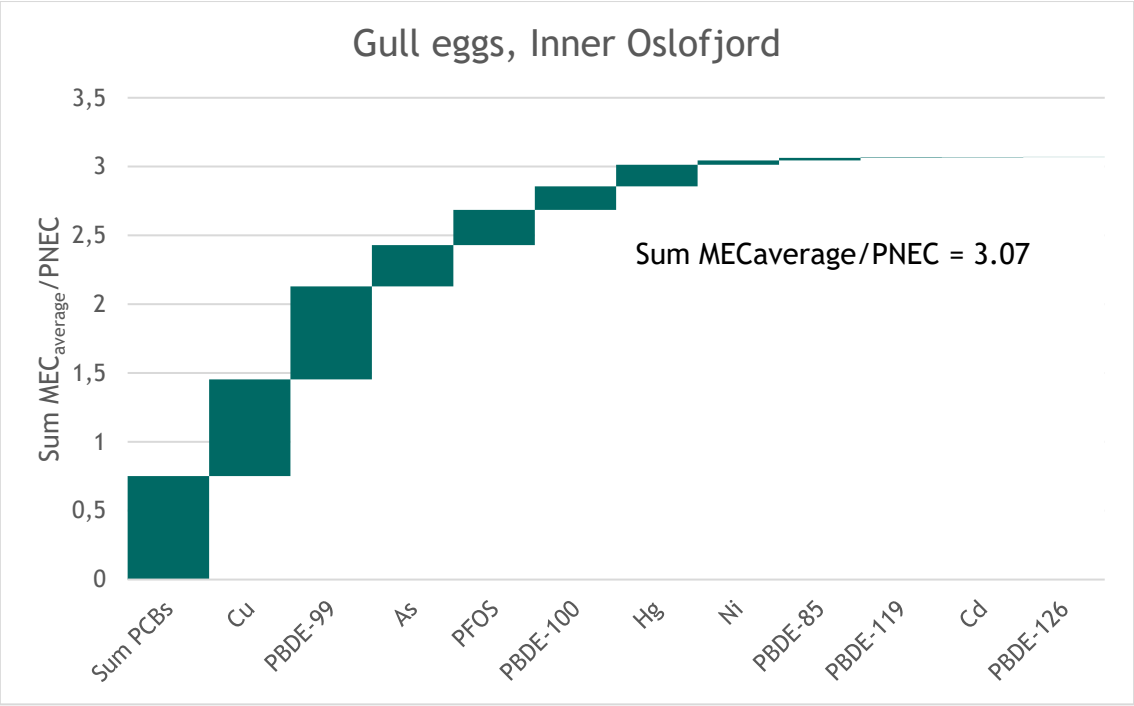


Figure 41. Contribution plot of MEC/$PNEC_{pred}$ summation for values measured in herring gull eggs from the Inner Oslofjord.

Table 22.
Calculation of MECaverage/effect ratios for herring gull eggs from the Outer Oslofjord

Compound	MEC_a (ng/g egg)	MEC_m (ng/g egg)	Effect value (ng/g egg)*	MEC_a/effect	MEC_m/effect
As	124	130	180	0.690	0.724
Cu	786	785	1160	0.678	0.677
Sum PCB	252	169	400	0.631	0.423
PFOS	38.5	32.1	100	0.385	0.321
Hg	84.5	50.2	400	0.211	0.125
PBDE 99	1.56	1.23	10	0.156	0.123
PBDE 100	0.906	0.87	10	0.0906	0.087
Ni	40.3	38.1	1000	0.0403	0.0381
PBDE 119	0.0650	0.0167	10	0.0065	0.00167
PBDE 85	0.0273	0.0107	10	0.00273	0.00107
Cd	0.17	0.16	100	0.00173	0.00161
PBDE 126	0.00195	0	10	0.0002	0
Sum				2.89	2.52

*Effect values were obtained from Andersen et al. (2014)

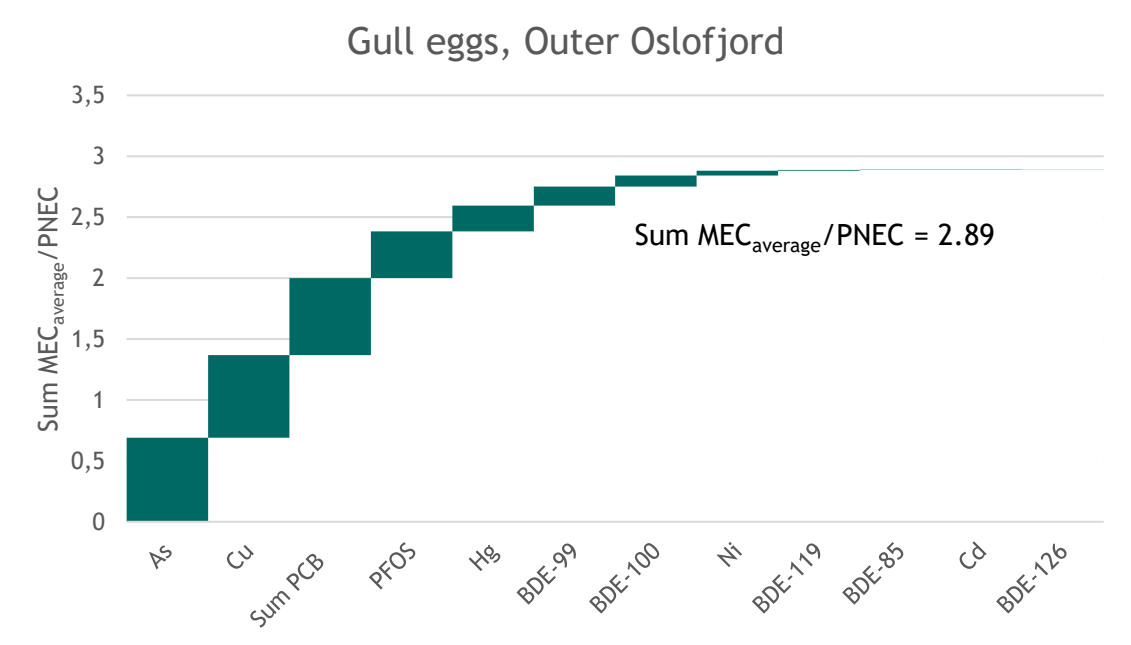


Figure 42. Contribution plot of MEC/$PNEC_{pred}$ summation for values measured in herring gull eggs from the Outer Oslofjord.

Eggs from eider ducks were also analysed for selected contaminants. None of the assessed compounds had MEC/effect ratios above 1 (using average or median concentration). The main contributors to the sum of MEC_a/effect was sum PCBs, Hg, and PBDE-99 (Figure 43), which is similar to the results from the gull eggs. The lower sum $MEC_{average}$/PNEC for eider duck compared to the gulls can be due to the lower number of compounds and elements analysed for in eider ducks. Especially as Cu, As, Cd, Ni and PFOS contributed to the total sum in gull eggs but were not analysed for in eider ducks.

Table 23.
Calculation of MEC/effect ratios for eider duck eggs

Compound	MEC_a (ng/g egg)	MEC_m (ng/g egg)	Effect value (ng/g egg)*	MEC_a/effect	MEC_m/effect
Sum PCB	255	199	400	0.636	0.497
Hg	154	138	400	0.384	0.346
BDE-99	0.245	0.224	10	0.0245	0.0224
BDE-100	0.227	0.168	10	0.0227	0.0168
BDE-119	0.0103	0.00966	10	0.00103	0.000966
BDE-85	0.00505	0.00515	10	0.000505	0.000515
BDE-126	0.00115	0	10	0.000115	0
Sum				1.07	0.884

*Effect values were obtained from Andersen et al. (2014)

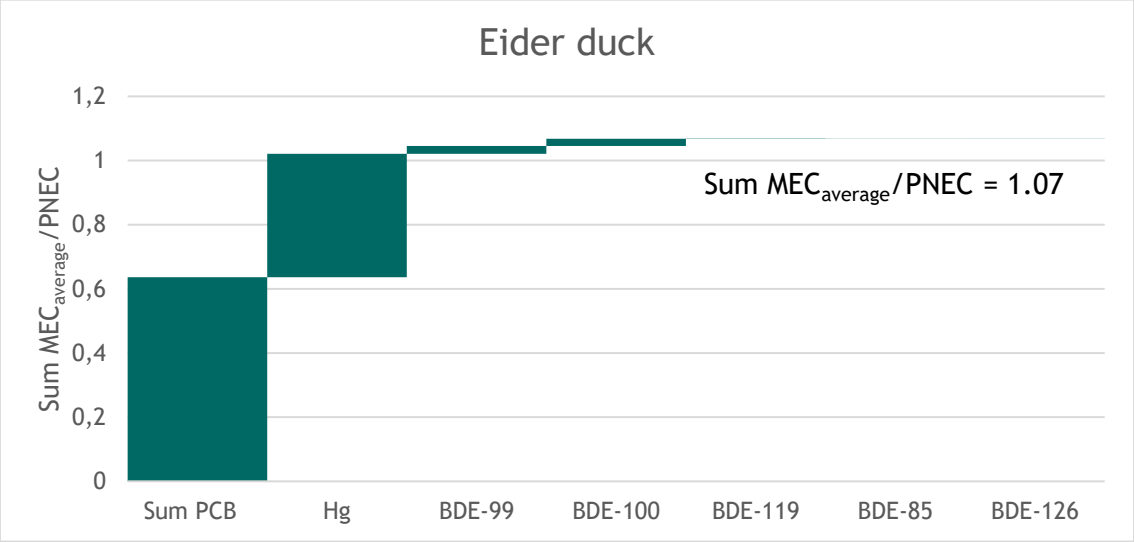


Figure 43. Contribution plot of MEC/PNEC$_{pred}$ summation for values measured in eider duck eggs from the Inner Oslofjord.

Based on the measured concentrations of pollutants in herring gull and eider duck eggs, and effect data compiled by Andersen et al. (2014), there is a risk for combined effects, mainly

driven by the sum of PCBs, PBDE-99, and the metals Cu, As, and Hg depending on bird population. These findings are similar to that observed by Herzke et al. (2015; The Norwegian Environment Agency M-354) where a sum MEC/effect for compounds measured in sparrow hawk eggs were higher than 1. As many as 76 detected compounds (herring gull eggs, inner Oslofjord) were excluded from the assessment due to lack of effect data, adding some uncertainty to the estimation and a potential underestimation of the risk. The results should be interpreted with caution due to the nature of the effect data. The effect data do not correspond to the same endpoint, the same species or the same effect level, adding additional uncertainty to the performed assessment.

3.7 Concluding remarks

In this programme, a large number of chemical parameters have been quantified, in addition to a few biological effect parameters and support parameters. Concentrations of different chemicals in different compartments of the Inner Oslofjord marine ecosystem are documented.

The sediments of the inner Oslofjord is a potential source of environmental contaminants to sediment dwelling organisms and the contaminants may thus enter the food chain. Several of the target compounds of this study were detected in the sediment, such as PCBs, PBDEs and other brominated flameretardants, S/MCCPs, siloxanes, phenolic compounds, metals, PFAS compounds, UV chemicals and dechlorane. Inputs to the fjord via storm water and STP effluent water for several of the compounds is also shown, including also phenolic compounds, PFRs (only STP effluent) and behentrimonium (only storm water). Some compounds exceeded environmental quality standards. These were in sediments: D5, PCB7, Cu, Zn, As, Ni, Pb, Hg and PFOS, in storm water: Bisphenol A, MCCPs, Cu, Zn and PFOS, and in STP effluent water: D5, MCCPs and PFOS.

Some changes were made in the programme from 2016 to 2017, and in 2017 the programme included additional sampling of herring gull (eggs and blood) also in the Outer Oslofjord, as well as sampling of eider duck (eggs and blood) in the Inner Oslofjord. The results of the stable isotope analysis suggest that the marine species (fish and invertebrates) represent members of the marine food web of the Inner Oslofjord. The differences in $\delta^{15}N$ seem to reflect expected trophic relationships; blue mussel (filters particulate organic matter from the water) < zooplankton (herbivore) = polychaetes (different modes of living, largely detritivorous) < herring (pelagic fish feeding on zooplankton) = prawns (some scavenging behaviour) < cod (mesopelagic fish, predator on fish and benthic organisms). The food web spans over 2 to 3 (~2.3) trophic levels with blue mussel defined at trophic level 2. Furthermore, the isotopic signatures of the eider duck correspond much better with a member of the Inner Oslofjord Marine food web, compared to herring gull, because of their marine diet.

The biomagnification potential of contaminants were evaluated by calculation of Trophic Magnification Factors (TMFs) and several contaminants, and especially legacy contaminants with well-known biomagnifying properties, displayed a positive significant relationship between (log_{10}-)concentrations and trophic position (deduced from the $\delta^{15}N$ isotopic ratio) in the studied Inner Oslofjord marine food web. This suggests that the selected food web is suitable for studying biomagnification in the Oslo fjord. For several compounds, this was the

case also when eider duck was included in the food web. PFOSA, As and Ag were also compounds that displayed a significant TMF>1.

Behentrimonium (ATAC-C20 and ATAC-C22) are quaternary ammonium compounds (QACs). QACs are widely used as ingredients in industrial applications and household products, and were included in the Urban fjord programme in 2017. Interestingly, behentrimonium showed significant biomagnification (significant TMF<1) on a wet weight basis, with high concentrations in cod liver, but not on a lipid weight basis.

Dechlorane plus is used as a flame retardant in plastics and polymers, such as nylon, polyurethane, polypropylene, neoprene and silicone rubber. Dechlorane plus was also included in the Urban fjord programme in 2017 and was detected in particulate phases (particulate fraction in storm water, sewage sludge and sediment). Furthermore, it was found in polychaetes, cod and bird eggs (herring gulls from the Inner and Outer Oslofjord, as well as eider duck from the Inner Oslofjord).

In addition to cyclic siloxanes (D4, D5 and D6), M3T(Ph) was analysed in the Urban fjord programme in 2017. It was detected in several matrices, however in modest concentrations compared to the cyclic siloxanes and especially D5.

4,4'-methylenebis[2,6-bis(1,1-dimethylethyl)-phenol (MB1) is used as an industrial anti-oxidant and additive to plastics. It was analysed in the Urban fjord programme in 2017. It was, however, not detected in any samples.

UV-chemicals (octocrylene, benzophenone and ethylhexylmethoxycinnamate) were, as previously, detected in very few samples in 2017. They were however found in notable concentrations in samples from Bekkelaget STP, and especially in sludge. Furthermore, phenolic compounds were detected in few samples in 2017, however, the limit of detection was high for some of the compounds, due to blank issues (high concentrations in blank samples).

The concentrations of PBDEs (e.g. BDE-47 and -209) and D5 in herring gull eggs from the present study (Inner Oslofjord) displayed concentrations that were higher than those previously observed in herring gull eggs sampled from remote colonies in Norway, indicating urban influence. On the other hand, concentrations of “legacy” contaminants, such as PCB-153 and sumPCB7 appeared lower in the eggs from Oslofjorden. There were also some differences in concentrations of contaminants between Herring gulls of the Inner and Outer Oslofjord, although many appeared similar. For instance, several of the PFAS compounds (e.g. PFOS) was found in higher concentrations in the gulls of the Outer Oslofjord (both blood and eggs), possibly related to local contamination in the area because of an earlier airport in proximity of the colony. Higher $\delta^{15}N$ ratios in the Outer Oslofjord gulls, than in the Inner Oslofjord gulls could suggest that the Outer Oslofjord gulls include more diet items of marine origin than the inner Oslofjord gulls.

Interestingly, in blood of gulls, concentrations of DBDPE were higher than concentratioins of any PBDE congeners, as also observed in 2016. DBDPE is a substitute for BDE-209 and future monitoring will indicate potential temporal trends. DBDPE was also higher than any PBDE congener both in the dissolved and in the particulate fraction of stormwater. Concentrations

of DBDPE was higher in the particulate fraction, than in the dissolved fraction. Furthermore, DBDPE was found in sludge from Bekkelaget sewage treatment plant.

While the concentrations of PCBs in sparrow hawk eggs from the Oslo area appeared higher than in the herring gull eggs from the Inner Oslofjord area, BDE-209 and siloxanes appeared higher in the gull eggs than in the sparrow hawk eggs. This is possibly reflecting that while the sparrow hawk feeds mostly on birds, the herring gull might feed on human waste and leftovers.

The risk of secondary poisoning of seabirds feeding on blue mussels, polychaetes or herring was calculated by summing up the $MEC/PNEC_{pred}$ values. Available $PNEC_{pred}$ values ($PNEC_{pred}$ and $QS_{biota,secpois,hh}$ for compounds where no $PNEC_{pred}$ was available) were only found for 26 compounds or compound groups leading to exclusion of several detected compounds from the cumulative risk estimation. All three food sources were estimated to pose a risk for the predating seabirds, with sum PBDEs and sum PCBs being among the main risk drivers in all food sources and with the addition of Cd in especially blue mussels. As the values used for calculation of sum PBDE and sum PCB7 are the $QS_{biota,secpois,hh}$ it should be noted that these values are considered to be more conservative than $PNEC_{pred}$ values, leading to a potential overestimation of the risk and the results should be interpreted with caution. The combination of $PNEC_{pred}$ and $QS_{biota,secpois,hh}$ add uncertainty to the estimates as they are derived by different methods.

The combined risk of effects in herring gull (Inner and Outer Oslofjord) and eider duck (Inner Oslofjord) eggs were calculated by comparing average (MEC_a) and median (MEC_m) values of the measured egg concentrations in 15 eggs from each species/site with effect data from exposure in eggs. Using average measured concentrations led to a higher sum of MEC/Effect ratios than when using median measured concentrations. In both cases (average and median values) the sum of MEC/effect was higher than 1 in herring gull eggs, indicating a risk for effect on the eggs of the mixture of contaminants. Only the sum of MEC_a/effect was higher than 1 for eider duck eggs. None of the assessed compounds had MEC/effect ratios above 1 (using average or median concentration). The sum MECa/effect was slightly higher in gull eggs from the Inner Oslofjord (3.07) than from the Outer Oslofjord (2.89).

Overall, there is a risk for combined effects in birds, mainly driven by the sum of PCBs, PBDE-99, and the metals Cu, As, and Hg depending on bird population.

In summary, it is shown that sediments and organisms in the inner Oslofjord contain different contaminants in different concentrations, both legacy contaminants and contaminants of more emerging concern. Some pathways for these contaminants into the fjord are also shown. For instance, chlorinated paraffins apparently constitute major proportions in all species/matrices examined. PCBs constituted a large proportion of the sum of contaminants in the lipid rich cod livers. Furthermore, siloxanes were important constituents of the sum of contaminants in cod liver, as in other species of the marine food web, especially krill and herring. A combined risk assessment showed that apex predators, such as seabirds (herring gull), might be at risk to negative effects of contaminants. Legacy contaminants were still important risk drivers.

4. References

Aigars J, Suhareva N, Poikane R. 2017. Distribution of polybrominated diphenyl ethers in sewage sludge, sediments, and fish from Latvia. *Environments* 4, 12; doi:10.3390/environments4010012

Amlund H. 2005. The disposition of arsenobetaine in Atlantic salmon, *Salmo salar* L., and Atlantic cod, *Gadus morhua* L. Dissertation for the degree of *Doctor scientiarum*, University if Bergen, Bergen 2005.

Andersen S, Gudbrandsen M, Haugstad K, Hartnik T. 2012. Noen miljøskadelige stoffer i avløpsslam – forekomst og miljørisiko. Oslo, Norwegian Climate and Pollution Agency report TA-3005/2012, 37pp.

Andersen S, Guldbrandsen M, Farmen E. 2014. Kartlegging av miljøgifter i sjøfuglegg fra Røst og Sklinna – Beregning av fare for effekter. Oslo, Norway. Norwegian Environment Agency report M-177/2014, 12pp.

Assis CRD, Castro PF, Amaral IPG, Carvalho EVMM, Carvalho LB, Bezerra RS. 2010. Characterization of acetylcholinesterase from the brain of the amazonian tambaqui (*Colossoma macropomum*) and in vitro effect of organophosphorus and carbamate pesticides. Environmental Toxicology and Chemistry. 29:2243-2248.

Backhaus T, Faust M. 2012. Predictive environmental risk assessment of chemical mixtures: a conceptual framework. *Environmental Science & Technology* 46:2564–2573.

Backhaus T, Karlsson M. 2014. Screening level mixture risk assessment of pharmaceuticals in STP effluents. *Water Res*. 49:157–165.

Bocquené G, Galgani F. 1998. Biological effects of contaminants: Cholinesterase inhibition by organophosphate and carbamate compounds. International Council for the Exploration of the Sea: 1998.

Borgå K, Fjeld E, Kierkegaard A, McLachlan MS. 2012. Food Web Accumulation of Cyclic Siloxanes in Lake Mjosa, Norway. *Environmental Science & Technology* 46:6347-6354.

Bosch AC, O'Neill B, Sigge GO, Kerwath SE, Hoffman LC. 2016. Heavy metals in marine fish meat and consumer health: A review. *Journal of the Science of Food and Agriculture* 96:32–48.

Brooke DN, Crookes MJ, Gray D, Robertson S. 2009b. Environmental risk assessment report: Octamethylcyclotetrasiloxane. Environment Agency, ISBN: 978-1-84911-031-0

Brooke DN, Crookes MJ, Gray D, Robertson S. 2009a. Environmental risk assessment report: Dodecamethylcyclohexasiloxane. Environment Agency, ISBN: 978-1-84911-030-3.

Bundschuh M, Goedkoop W, Kreuger J. 2014. Evaluation of pesticide monitoring strategies in agricultural streams based on the toxic-unit concept--experiences from long-term measurements. *Science of the Total Environment* 484:84–91.

Burgeot T, Bocquene G, Forget-Leray J, Guilhermino L, Martinex-Gomez C, Lehtonen K. 2012. Background document: acetylcholine esterase assay as a method for assessing neurotoxic effects in aquatic organisms. P.49-53. In: Integrated marine environmental monitoring of chemicals and their effects. ICES Cooperative research report 315, Nov 2012, Davies IM, Vethaak D.

Companioni-Damas EY, Santos FJ, Galceran MT. 2012. Analysis of linear and cyclic methylsiloxanes in sewage sludges and urban soils by concurrent solvent recondensation-large volume injection-gas chromatography-mass spectrometry. *Journal of Chromatography A* 1268:150–156.

Di Tuoro D, Portaccio M, Lepore M, Arduini F, Moscone D, Bencivenga U, Mita DG. 2011. An acetylcholinesterase biosensor for determination of low concentrations of paraoxon and dichlorvos. New Biotechnology. 29:132-138.

ECHA. 2015. Opinion on the application for approval of the active substance: Triclosan. Biocidal Products Committee. ECHA/BPC/066/2015.

Eikenberry BCS, Riva-Murray K, Knightes CD, Journey CA, Chasar LC, Brigham ME, Bradley PM. 2015. Optimizing fish sampling for fish-mercury bioaccumulation factors. *Chemosphere* 135:467-473.

Finizio A, Villa S, Vighi M. 2005. Predicting pesticide mixtures load in surface waters from a given crop. *Agriculture, Ecosystems & Environment* 111:111–118.

Fisher NS, Wang W-X. 1998. Trophic transfer of silver to marine herbivores: a review of recent studies. *Environ Toxicol Chem* 17:562-571.

Fisk AT, Hobson KA, Norstrom RJ. 2001. Influence of chemical and biological factors on trophic transfer of persistent organic pollutants in the Northwater Polynya marine food web. *Environ Sci Technol* 35:732-738.

Fjeld E, Bæk K, Rundberget JT, Schlabach M, Warner NA. 2017. Environmental pollutants in large Norwegian lakes, 2016. Norwegian Environment Agency report M-807/2017, 88 pp.

Franklin J. 2015. How reliable are field-derived biomagnification factors and trophic magnification factors as indicators of bioaccumulation potential? Conclusions from a case study on per- and polyfluoralkyl substances. *Integrated Environmental Assessment and Management* 12:6-20.

Gentes ML, Mazerolle MJ, Giroux JF, Petenaude-Monette M, Verreault J. 2015. Tracking the sources of polybrominated diphenyl ethers in birds: Foraging in waste management facilities results in higher DecaBDE exposure in males. *Environ Res* 138:361-371.

Green NW, Knutzen J. 2003. Organohalogens and metals in marine fish and mussels and some relationships to biological variables at reference localities in Norway. *Marine Pollution Bulletin* 46:362-374.

Green NW, Schøyen M, Øxnevad S, Ruus A, Hjermann D, Severinsen G, Høgåsen T, Beylich B, Håvardstun J, Lund E, Tveiten L, Bæk K. 2017. Contaminants in coastal waters of Norway 2016. Norwegian Environment Agency report M-856/2017, 201 pp.

Green NW, Schøyen M, Hjermann D, Øxnevad S, Ruus A, Lusher A, Beylich B, Lund E, Tveiten L, Håvardstun J, Jenssen MTS, Ribeiro AL, Bæk K. 2018. Contaminants in coastal waters of Norway 2017. Norwegian Environment Agency report M-1120/2018.

Hallanger IG, Warner NA, Ruus A, Evenset A, Christensen G, Herzke D, Gabrielsen GW, Borgå K. 2011. Seasonality in contaminant accumulation in arctic marine pelagic food webs using trophic magnification factor as a measure of bioaccumulation. Environ Toxicol Chem 30:1026-1035.

Heimstad ES, Nygård T, Herzke D, Bohlin-Nizzetto P. 2017. Environmental pollutants in the terrestrial and urban environment 2017. Norwegian Environment Agency report M-1076/2018, 234 pp.

Helland A, Aberg G, Skei J. 2002. Source dependent behaviour of lead and organic matter in the Glomma estuary, SE Norway: evidence from isotope ratios. *Marine Chemistry* 78:149-169.

Herzke D, Nygård T, Heimstad ES, Uggerud H, Hanssen L, Götsch A. 2014. Environmental pollutants in the terrestrial and urban environment. Oslo, Norway. Norwegian Environment Agency report M-261/2014, 113 pp.

Herzke D, Nygård T, Heimstad ES, Uggerud HT. 2015. Environmental pollutants in the terrestrial and urban environment, part II. Oslo, Norway. Norwegian Environment Agency report M-354/2015, 103 pp.

Hobson KA, Welch HE. 1992. Determination of trophic relationships within a high Arctic marine food web using delta-C-13 and delta-N-15 analysis. *Marine Ecology Progress Series* 84:9-18.

Houde M, De Silva AO, Muir DCG, Letcher RJ. 2011. Monitoring of perfluorinated compounds in aquatic biota: An updated review. *Environmental Science & Technology* 45:7962-7973.

Huber S, Warner NA, Nygård T, Remberger M, Harju M, Uggerud HT, Kaj L, Hanssen L. 2015. A broad cocktail of environmental pollutants found in eggs of three seabird species from remote colonies in Norway. *Environmental Toxicology and Chemistry* 34:1296-1308.

Jaeger I, Hop H, Gabrielsen GW. 2009. Biomagnification of mercury in selected species from an Arctic marine food web in Svalbard. *Science of the Total Environment* 407:4744-4751.

Jones HJ, Swadling KM, Tracey SR, Macleod CK. 2013. Long term trends of Hg uptake in resident fish from a polluted estuary. *Marine Pollution Bulletin* 73:263-272.

Julshamn K, Duinker A, Nilsen BM, Nedreaas K, Maage A. 2013. A baseline study of metals in cod (Gadus morhua) from the North Sea and coastal Norwegian waters, with focus on mercury, arsenic, cadmium and lead. *Marine Pollution Bulletin* 72:264-273.

Kaj L, Wallberg P, Brorström-Lundén E. 2014. Quatenary ammonium compounds – Analyses in a Nordic cooperation on screening. TemaNord report 2014:556 from the Nordic Council of Ministers. 66 pp.

Kelly BC, Ikonomou MG, Blair JD, Surridge Bm Hoover D, Grace R, Gobas FAPC. 2009. Perfluoralkyl contaminants in an Arctic marine food web: Trophic magnification and wildlife exposure. *Environmental Science & Technology* 43:4037-4043.

Krogseth IS, Kierkegaard A, McLachlan MS, Breivik K, Hansen KM, Schlabach M. 2013. Occurrence and Seasonality of Cyclic Volatile Methyl Siloxanes in Arctic Air. *Environmental Science & Technology* 47:502-509.

Krumbein WC, Pettijohn FC.1938. Manual of sedimentary petrography. Appelton-Century-Crofts, New York.

Langford KH, Reid M, Thomas KV. 2011. Multi-residue screening of prioritised human pharmaceuticals, illicit drugs and bactericides in sediments and sludge. *Journal of Environmental Monitoring* 13:2284–2291.

Langford KH, Reid MJ, Fjeld E, Øxnevad S, Thomas KV. 2015. Environmental occurrence and risk of organic UV filters and stabilizers in multiple matrices in Norway. *Environment International* 80:1-7

Langford KH, Thomas KV. 2008. Inputs of chemicals from recreational activities into the Norwegian coastal zone. *Journal of Environmental Monitoring* 10:894.

Langford KH, Thomas KV. 2009. Determination of pharmaceuticals compounds in hospital effluents and their contribution to wastewater treatment works. *Environment International*. 35:766-770.

Layman CA, Araujo MS, Boucek R, Hammerschlag-Peyer CM, Harrison E, Jud ZR, et al. 2012. Applying stable isotopes to examine food-web structure: an overview of analytical tools. *Biological Reviews* 87:545-562.

Maneekarn Y, Chaichana R, Satapanajaru T. 2014. Toxicity, bioaccumulation and biomagnification of silver nanoparticles in green algae (*Chlorella sp.*), water flea (*Moina macroscopa*), blood worm (*Chironomus spp.*) and silver barb (*Barbonymus gonionotus*). *Chemical Speciation & Bioavailability* 26:257-265

McGoldrick DJ, Durham J, Leknes H, Kierkegaard A, Gerhards R, Powell DE, et al. 2011. Assessing inter-laboratory comparability and limits of determination for the analysis of cyclic volatile methyl siloxanes in whole Rainbow Trout (*Oncorhynchus mykiss*). *Chemosphere* 85:1241-1247.

McGrath TJ, Ball AS, Clarke BO. 2017. Critical review of soil contamination by polybrominated diphenyl ethers (PBDEs) and novel brominated flame retardants (NBFRs); concentrations, sources and congener profiles. *Environmental Pollution* 230:741-757.

Mizutani H, Kabaya Y, Wada E. 1991. Nitrogen and carbon isotope compositions relate linearly in cormorant tissues and its diet. *Isotopenpraxis* 27:166-168.

Moschet C, Wittmer I, Simovic J, Junghans M, Piazzoli A, Singer H, Stamm C, Leu C, Hollender J. 2014. How a Complete Pesticide Screening Changes the Assessment of Surface Water Quality. *Environmental Science & Technology* 48:5423–5432.

Petersen K, Stenrød M, Tollefsen KE. 2013. Initial environmental risk assessment of combined effects of plant protection products in six different areas in Norway. Oslo, Norway: NIVA report 6588-2013, 35 pp.

Post DM. 2002. Using stable isotopes to estimate trophic position: Models, methods, and assumptions. *Ecology* 83:703-718.

Powell DE, Schøyen M, Øxnevad S, Gerhards R, Böhmer T, Koerner M, Durham J, Huff DW. 2018. Bioaccumulation and trophic transfer of cyclic volatile methylsiloxanes (cVMS) in the aquatic marine food webs of the Oslofjord, Norway. *Science of the Total Environment* 622-623:127-139; http://www.sciencedirect.com/science/article/pii/S0048969717332941

Ruus A, Allan I, Beylich B, Bæk K, Schlabach M, Helberg M. 2014. Environmental Contaminants in an Urban Fjord. Report M205-2014 from the Norwegian Environment Agency. 120 pp.

Ruus A, Allan I, Beylich B, Bæk K, Schlabach M, Helberg M. 2015. Environmental Contaminants in an Urban Fjord, 2014. Report M375-2015 from the Norwegian Environment Agency. 110 pp + Appendix.

Ruus A, Bæk K, Karina Petersen, Allan I, Beylich B, Schlabach M, Warner N, Helberg M. 2016. Environmental Contaminants in an Urban Fjord, 2015. Report M601-2016 from the Norwegian Environment Agency. 84 pp + Appendix.

Ruus A, Bæk K, Karina Petersen, Allan I, Beylich B, Schlabach M, Warner N, Borgå K, Helberg M. 2017. Environmental Contaminants in an Urban Fjord, 2016. Report M812-2017 from the Norwegian Environment Agency. 114 pp + Appendix.

Ruus A, Øverjordet IB, Braaten HFV, Evenset A, Christensen G, Heimstad ES, Gabrielsen GW, Borgå K. 2015. Methylmercury biomagnification in an Arctic pelagic food web. *Environmental Toxicology and Chemistry* 34:2636-2643.

Sackett DK, Cope WG, Rice JA, Aday DD. 2013. The influence of fish length on tissue mercury dynamics: Implications for natural resource management and human health risk. *International Journal of Environmental Research and Public Health* 10:638-659.

Safe SH. 1994. Polychlorinated biphenyls (PCBs): environmental impact, biochemical and toxic responses, and implications for risk assessment. *Crit Rev Toxicol* 24:87-149.

Schlabach M, Strand Andersen M, Green N, Schøyen M, Kaj L. 2007. Siloxanes in the Environment of the inner Oslofjord, NILU report, 986/2007 (TA-2269/2007); https://www.nilu.no/Default.aspx?tabid=62&ctl=PublicationDetails&mid=764&publicationid=18508

Shaw BP, Panigrahi AK. 1990. Brain AChE activity studies in some fish species collected from a mercury contaminated estuary. *Water, Air, and Soil Pollution* 53:327-334.

Skarbøvik E, Allan I, Sample JE, Greipsland I, Selvik JR, Schanke LB, Beldring S, Stålnacke P, Kaste Ø. 2017. Riverine inputs and direct discharges to Norwegian coastal waters - 2016. Report M862-2017 from the Norwegian Environment Agency. 85 pp + Appendix.

Sparham C, Van Egmond R, O'Connor S, Hastie C, Whelan M, Kanda R, Franklin O. 2008. Determination of decamethylcyclopentasiloxane in river water and final effluent by headspace gas chromatography/mass spectrometry. *Journal of Chromatography A* 1212:124–129.

Sweeting CJ, Polunin NVC, Jennings S. 2006. Effects of chemical lipid extraction and arithmetic lipid correction on stable isotope ratios of fish tissues. *Rapid Commun Mass Spectrom* 2006; 595-601.

The Norwegian Environment Agency. 2016. Quality Standards for water, sediment and biota. Report M-608. 24 pp.

Thomas KV, et al. 2014. New bisphenols, organic peroxides, fluorinated siloxanes, organic UV filters and selected PBT substances. Miljødirektoratet M-176/2014

Van Bavel B, Thomas KV, Langford K, Reid M, Vogelsang C, Øxnevad S, Bæk K, Fjeld E, Brooks S, Schlabach M, Rostowski P, Warner N, Borgen A, Halse AK, Davanger K, Gundersen H. 2016. Screening programme 2015: Benzothiazoles, siloxanes, pigments & PBT compounds. Report M596-2016 from the Norwegian Environment Agency. 49 pp.

VEAS. 2017. Årsrapport for 2016 – En renere Oslofjord. 66 pp.

Vieira LR, Gravato C, Soares AMVM, Morgado F, Guilhermino L. 2009. Acute effects of copper and mercury on the estuarine fish Pomatoschistus microps: Linking biomarkers to behaviour. *Chemosphere* 76:1416-1427.

Wang H, Ho KT, Scheckel KG, Wu F, Cantwell MG, Katz DR, Horowitz DB, Boothman WS, Burgess RM. 2014. Toxicity, Bioaccumulation, and Biotransformation of Silver Nanoparticles in Marine Organisms. *Environmental Science & Technology* 48:13711–13717.

Warner NA, Evenset A, et al. 2010. Volatile Siloxanes in the European Arctic: Assessment of Sources and Spatial Distribution. Environmental Science & Technology 44(19):7705-7710.

Warner NA, Kozerski G, et al. 2012. Positive vs. false detection: A comparison of analytical methods and performance for analysis of cyclic volatile methylsiloxanes (cVMS) in environmental samples from remote regions. Chemosphere 93:749-756.

Warner NA, Kozerski G, Durham J, Koerner M, Gerhards R, Campbell R, et al. 2013. Positive vs. false detection: A comparison of analytical methods and performance for analysis of cyclic volatile methylsiloxanes (cVMS) in environmental samples from remote regions. *Chemosphere* 93:749-756.

Wolfe MF, Schwarzbach S, Sulaiman RA. 1998. Effects of mercury on wildlife: A comprehensive review. Environmental Toxicology and Chemistry 17:146-160.

Appendix

Concentrations in individual samples and composition of (calculated) pooled samples of cod are available as electronic appendix

Table A1.
Support parameters measured for sediment from the inner Oslofjord.

Area	<63 µm (% dry wt.)	TOC (µg/mg dry wt.)
Inner Oslofjord (station Cm21)	73	33.8

Table A2.
Support parameters measured for effluent water and sludge from Bekkelaget sewage treatment plant.

Sample	DOC (mg C/L)	TOC (µg/mg dry wt.)	Suspended solids (mg/L)
Effluent water (June)	6.9		15.6
Effluent water (August)	7.4		<1.6
Sludge (June)		267	
Sludge (August)		263	

Table A3.
Support parameters measured for herring gull eggs from the Inner Oslofjord area.

Sample no.	Specimen/ nest	$\delta^{34}S$	$\delta^{13}C$	$\delta^{15}N$	C:N (W%)	Trophic position	Weight, egg (g)	Eggshell thickness (mm)
1	J5549	n.a.	-25.15	10.79	8.35	3.23	88.2	0.41
2	JCL23	n.a.	-26.13	7.70	7.94	2.42	62.43	0.37
3	JCL59	n.a.	-26.94	6.92	9.10	2.21	79.97	0.36
4	JCL67	n.a.	-26.37	8.91	8.86	2.73	84.52	0.40
5	JCL68	n.a.	-26.11	9.73	9.00	2.95	86.66	0.39
6	JCL72	n.a.	-25.62	9.26	6.53	2.82	85	0.37
7	JCP52	n.a.	-26.03	7.80	7.94	2.44	68.39	0.38
8	JJP01	n.a.	-26.50	8.80	8.17	2.70	74.21	0.36
9	JJP03	n.a.	-25.39	8.97	5.93	2.75	80.75	0.37
10	JJP05	n.a.	-24.73	8.13	6.32	2.53	72.86	0.39
11	JJP06	n.a.	-26.23	9.81	6.88	2.97	91.48	0.37
12	JJP07	n.a.	-26.85	8.27	8.85	2.56	69.47	0.39
13	JJP18	n.a.	-24.67	10.82	6.57	3.23	74.52	0.37
14	JJP19	n.a.	-25.35	8.93	6.23	2.74	97.78	0.39
15	JJP21	n.a.	-24.81	8.94	4.85	2.74	61.8	0.37

Table A4.
Support parameters measured for herring gull eggs from the Outer Oslofjord area.

Sample no.	Specimen/ nest	$\delta^{34}S$	$\delta^{13}C$	$\delta^{15}N$	C:N (W%)	Trophic position	Weight, egg (g)	Eggshell thickness (mm)
1	J884A	n.a.	-24.02	11.83	6.60	3.50	87.44	0.38
2	JJP33	n.a.	-26.61	8.83	7.85	2.71	87.82	0.39
3	JJP24	n.a.	-25.17	9.96	5.67	3.01	80.83	0.36
4	JJP25	n.a.	-22.97	12.68	5.70	3.72	90.37	0.40
5	JJP27	n.a.	-25.96	10.54	6.86	3.16	86.94	0.40
6	JJP28	n.a.	-23.86	9.83	5.20	2.98	88.3	0.40
7	JJP32	n.a.	-25.71	11.06	7.77	3.30	81.36	0.40
8	JJP34	n.a.	-23.89	10.17	4.95	3.07	90.25	0.39
9	JJP35	n.a.	-24.41	10.80	4.81	3.23	75.81	0.36
10	JJP36	n.a.	-26.01	11.98	8.16	3.54	84.08	0.37
11	JJP39	n.a.	-25.53	10.24	6.63	3.08	67	0.37
12	JJP41	n.a.	-23.30	14.37	7.36	4.17	89.15	0.39
13	JJP42	n.a.	-25.80	9.63	7.02	2.92	80.28	0.40
14	JJP46	n.a.	-25.11	10.62	6.22	3.18	71.93	0.37
15	JJP47	n.a.	-26.32	10.79	8.33	3.23	73.54	0.37

Table A5.
Support parameters measured for herring gull blood from the Inner Oslofjord.

Sample no.	Specimen/ nest	$\delta^{34}S$	$\delta^{13}C$	$\delta^{15}N$	C:N (W%)	Trophic position	Wing (mm)	Head (mm)	Weight (g)
1	J5549	n.a.	-24.36	10.17	5.02	3.06	432	117.4	930
2	JCL23	n.a.	-24.71	7.74	3.41	2.43	418	120.5	870
3	JCL59	n.a.	-25.18	7.39	4.49	2.33	427	110.9	770
4	JCL67	n.a.	-24.05	8.41	3.49	2.60	427	115.7	870
5	JCL68	n.a.	-24.87	9.09	5.30	2.78	437	120.9	890
6	JCL72	n.a.	-24.27	7.73	3.58	2.42	430	117	990
7	JCP52	n.a.	-24.11	8.47	4.00	2.62	422	117.8	930
8	JJP01	n.a.	-24.16	9.06	3.66	2.77	426	118.8	885
9	JJP03	n.a.	-24.68	7.67	3.89	2.41	434	118.4	910
10	JJP05	n.a.	-24.24	8.08	3.93	2.51	436	120.8	965
11	JJP06	n.a.	-24.79	9.19	4.93	2.81	437	117.2	860
12	JJP07	n.a.	-24.38	8.48	4.43	2.62	438	120.4	950
13	JJP18	n.a.	-24.19	8.36	3.71	2.59	429	113.6	830
14	JJP19	n.a.	-24.02	8.70	3.67	2.68	429	117.4	900
15	JJP21	n.a.	-24.74	8.10	3.97	2.52	415	115.8	900

Table A6.
Support parameters measured for herring gull blood from the Outer Oslofjord.

Sample no.	Specimen/ nest	$\delta^{34}S$	$\delta^{13}C$	$\delta^{15}N$	C:N (W%)	Trophic position	Wing (mm)	Head (mm)	Weight (g)
1	J884A	n.a.	-21.85	12.27	3.49	3.62	430	119.7	1000
2	JJP33	n.a.	-24.58	8.81	3.38	2.71	440	119.7	1090
3	JJP24	n.a.	-23.32	9.63	3.25	2.92	440	116.5	935
4	JJP25	n.a.	-22.26	10.58	3.40	3.17	447	118.2	1050
5	JJP27	n.a.	-24.76	9.66	3.46	2.93	428	117.7	900
6	JJP28	n.a.	-23.72	9.63	3.45	2.92	429	119.2	830
7	JJP32	n.a.	-24.09	9.99	3.48	3.02	429	118.2	925
8	JJP34	n.a.	-23.77	10.45	3.43	3.14	439	119	935
9	JJP35	n.a.	-24.96	9.34	3.34	2.85	414	113.7	875
10	JJP36	n.a.	-24.21	9.18	3.37	2.80	440	117.8	960
11	JJP39	n.a.	-24.14	9.56	3.36	2.90	415	118.1	820
12	JJP41	n.a.	-22.08	12.69	3.26	3.73	443	115.8	955
13	JJP42	n.a.	-24.78	9.64	3.52	2.92	426	116.5	845
14	JJP46	n.a.	-23.58	10.20	3.41	3.07	438	120.7	930
15	JJP47	n.a.	-24.70	8.55	3.43	2.64	412	116.8	830

Table A7.
Support parameters measured for eider duck eggs from the Inner Oslofjord area.

Sample no.	Specimen/ nest	$\delta^{34}S$	$\delta^{13}C$	$\delta^{15}N$	C:N (W%)	Trophic position	Weight, egg (g)	Eggshell thickness (mm)
1	CA...21507	n.a.	-23.36	12.01	9.68	3.55	105.3	n.a.
2	CA...21510	n.a.	-21.31	14.33	10.38	4.16	91.2	n.a.
3	CA...21512	n.a.	-21.57	13.07	10.29	3.83	105.6	n.a.
4	CA...47440	n.a.	-22.63	12.36	10.13	3.64	112.4	n.a.
5	CA...47439	n.a.	-22.48	12.75	10.12	3.74	106.7	n.a.
6	CA...47438	n.a.	-22.41	11.81	9.59	3.49	113.2	n.a.
7	CA...47441	n.a.	-19.78	12.21	10.16	3.60	94	n.a.
8	CA...47442	n.a.	-22.04	12.48	9.62	3.67	91.8	n.a.
9	CA...47443	n.a.	-22.74	12.88	10.07	3.78	103.2	n.a.
10	CA...47445	n.a.	-22.16	12.14	11.58	3.58	111.1	n.a.
11	CA...47258	n.a.	-21.26	14.06	9.23	4.09	119.9	n.a.
12	CA...47259	n.a.	-23.34	11.80	9.21	3.49	105.2	n.a.
13	CA...47260	n.a.	-22.73	11.46	9.37	3.41	112.8	n.a.
14	CA...47261	n.a.	-22.70	12.87	9.78	3.78	107.7	n.a.
15	CA...47262	n.a.	-23.19	13.20	9.76	3.86	106.4	n.a.

Table A8.
Support parameters measured for eider duck blood from the Inner Oslofjord.

Sample no.	Specimen/ nest	$\delta^{34}S$	$\delta^{13}C$	$\delta^{15}N$	C:N (W%)	Trophic position	Wing (mm)	Head (mm)	Weight (g)
1	CA...21507	n.a.	-20.33	11.13	3.60	3.32	294	124.2	1695
2	CA...21510	n.a.	-18.11	13.67	3.57	3.99	308	127	1465
3	CA...21512	n.a.	-21.77	12.53	5.20	3.69	311	130	1660
4	CA...47440	n.a.	-21.65	12.61	4.63	3.71	300	NA	1850
5	CA...47439	n.a.	-21.06	11.70	4.35	3.47	313	NA	2290
6	CA...47438	n.a.	-19.90	12.17	3.66	3.59	306	NA	2170
7	CA...47441	n.a.	-18.19	12.49	4.10	3.68	304	123.4	1770
8	CA...47442	n.a.	-19.36	12.61	3.66	3.71	305	124.6	1660
9	CA...47443	n.a.	-19.84	12.66	3.55	3.72	300	124.5	1720
10	CA...47445	n.a.	-22.17	12.10	5.85	3.57	314	128.1	2080
11	CA...47258	n.a.	-20.80	13.42	4.62	3.92	306	128.1	1525
12	CA...47259	n.a.	-22.50	12.24	5.00	3.61	308	127.7	1875
13	CA...47260	n.a.	-21.92	12.07	4.92	3.56	315	125.5	1820
14	CA...47261	n.a.	-21.29	12.45	4.64	3.66	302	121.7	1870
15	CA...47262	n.a.	-22.25	12.51	4.89	3.68	302	126.9	1630

Table A9.
Support parameters measured for Cod from the Inner Oslofjord.

Sample no.	$\delta^{13}C$	$\delta^{15}N$	C:N (W%)	Trophic position	Age (yr)	Length (cm)	Weight (g)	Liver weight (g)	Gonad weight (g)	Sex
1	-20.07	16.91	3.90	4.47	3	65	2640	61	22	M
16	-17.59	16.64	3.42	4.40	6	48	1060	42	10.2	F
3	-18.88	19.11	3.40	5.05	6	61	2600	230	29	F
4	-19.53	15.06	3.40	3.98	2	50	1360	45	56	F
5	-20.85	15.37	3.29	4.06	4	66	3060	161	26	M
6	-20.34	17.98	3.39	4.75	7	52	1480	36	93	F
7	-19.42	16.90	3.49	4.47	5	64	2600	64	22.3	F
8	-19.52	18.42	3.48	4.87	3	52	1400	42	6.5	F
9	-19.73	17.31	3.65	4.57	6	49	1280	106	3.5	M
10	-25.00	9.75	3.27	2.59	6	54	1540	88	7	M
11	-18.42	16.96	3.60	4.48	3	48.5	1140	35	3.5	M
12	-18.59	16.56	3.50	4.38	3	47	1080	44	29	F
13	-17.43	15.23	3.14	4.03	3	47	1040	68	4	M
14	-18.31	18.55	3.48	4.90	3	52	1420	36	54	F
15	-17.68	13.02	3.14	3.44	2	45.5	1040	39.4	1	M

Table A10.
Support parameters measured for compartments of the Inner Oslofjord marine food web; polychaetes, blue mussel, krill, prawns, herring, cod (mathematically derived pooled samples).

Species	Sample sub no.	$\delta^{13}C$	$\delta^{15}N$	C:N (W%)	Trophic position
Polychaeta	1	-18.90	12.24	3.77	3.24
Polychaeta	2	-19.53	12.06	4.84	3.19
Polychaeta	3	-18.13	13.79	3.92	3.65
Blue mussel	1	-19.38	7.48	4.99	1.99
Blue mussel	2	-18.63	7.31	5.34	1.94
Blue mussel	3	-18.98	7.78	5.00	2.07
Krill	1	-20.43	13.17	4.01	3.49
Krill	2	-20.42	12.71	4.04	3.37
Krill	3	-20.33	8.64	4.03	2.29
Prawns	1	-17.54	14.32	3.35	3.79
Prawns	2	-17.62	14.21	3.28	3.76
Prawns	3	-17.66	14.61	3.38	3.87
Herring	1	-20.52	13.38	4.20	3.54
Herring	2	-21.06	13.42	4.55	3.55
Herring	3	-19.73	12.86	3.55	3.40
Cod (pool 1)	1	-18.37	15.81	3.40	4.18
Cod (pool 2)	2	-20.54	15.95	3.41	4.22
Cod (pool 3)	3	-19.36	16.99	3.50	4.49

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