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Contaminants in coastal waters of Norway 2014

Miljøgifter i norske kystområder 2014

Foreword

This report presents the investigations of contaminants in coastal waters of Norway 2014 which also represents the Norwegian contribution to Coordinated Environmental Monitoring Programme (CEMP, a part of and referred to in earlier reports as the Joint Assessment and Monitoring Programme JAMP). CEMP is administered by the Oslo and Paris Commissions (OSPAR) in their effort to assess and remedy anthropogenic impact on the marine environment of the North East Atlantic. The current focus of the Norwegian contribution is on the levels, trends and effects of hazardous substances. The results from Norway and other OSPAR countries provide a basis for a paramount evaluation of the state of the marine environment. OSPAR receives guidance from the International Council for the Exploration of the Sea (ICES).

The 2014 investigations were carried out by the Norwegian Institute for Water Research (NIVA) by contract from the Norwegian Environment Agency (*Miljødirektoratet* where the former Climate and Pollution Agency is now a part of). The project leader at the Norwegian Environment Agency is Bård Nordbø and the project manager at NIVA is Norman W. Green

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Abstract

This programme examines the levels, trends and effects of contaminants in biota along the coast of Norway. The 2014-investigation included analyses of 136 different contaminants or biological effect parameters in five types of samples (blue mussel, dog whelk, common periwinkle, cod and passive samplers). The contaminants include metals (Hg, Cd, Pb, Cu, Zn, Ag, As, Ni, Cr and Co), organochlorines (e.g. PCBs, DDT), PAHs, polybrominated diphenyl ethers (PBDEs), perfluorinated alkylated substances (PFAS) as well as contaminants that have recently received more attention such as hexabromcyclododecane (HBCDs), chlorinated paraffins (SCCP, MCCP), phosphorus flame retardants (PFRs),bisphenol A (BPA), tetrabrombisphenol A (TBBPA), alkyphenols, phthalates, triclosan, Diuron and Irgarol. Biological effects parameters included VDSI, OH-pyrene metabolites, ALA-D and EROD. In the report, thirty representative substances or parameters were chosen for analyses of 759 time series (last 10 years). Of these there were statistically significant trends in 104 cases: 86 were downwards and 18 upwards. The dominance of downward trends indicated that contamination is decreasing for the measured substances. The downwards trends for TBT-concentrations and effect parameter (VDSI) confirmed that the legislation banning the use of TBT has been effective. Of the same 759 cases, 403 could be classified by the environmental classification system used by the Norwegian Environment Agency, 374 were classified as insignificantly polluted, 26 as moderately polluted, two as markedly polluted and one as extremely polluted. Some cases warrant special concern, such as upward trend for mercury in cod fillet and high concentrations of several organic pollutants in cod liver from the Inner Oslofjord. Very high concentrations of DDE in mussels from the Sørfjord were related to earlier use of DDT as pesticide in orchards along the fjord. The relation of fish length on mercury concentration and affect of different sizes of pooled samples were examined. Alternatives to using cod liver as a target tissue was discussed.

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2. Biologiske effekter	2. Biological effects
3. Marin	3. Marine
4. Norge	4. Norway

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English summary

This programme examines the levels, trends and effects of contaminants along the coast of Norway, including some new contaminants that have recently received more attention. As such, the programme provides a basis for assessing the state of the environment for the coastal waters with respect to contaminants. Most trends were downwards.

The Inner Oslofjord seems all together to be an area where contaminants tend to appear in high concentrations and hence warrant special concern. For example, the investigation found an upward trend for mercury (Hg) in cod fillet and high concentrations of lead (Pb), polychlorinated biphenyls (PCB), polybrominated diphenyl ethers (PBDEs), perfluorinated alkylated substances (PFAS) and alpha-hexabromocyclododecane (α–HBCD) in cod liver.

Monitoring contaminants and associated parameters along the Norwegian coast contributes to OSPAR's Coordinated Environmental Monitoring Programme (CEMP). The 2014-investigation monitored blue mussel (32 stations), dog whelk (9 stations), common periwinkle (1 station), cod (14 stations) and seawater using passive sampling (3 stations) along the coast of Norway from the Oslofjord and Hvaler region in the southeast to the Varangerfjord in the northeast. The stations are located both in areas with known or presumed point sources of contaminants, in areas of diffuse load of contamination like city harbour areas, and in more remote areas exposed to presumed low pollution. The programme for 2014 and some supplementary analyses of 2013-samples, included analyses of metals (Hg, cadmium (Cd), Pb, copper (Cu), zinc (Zn), silver (Ag), arsenic (As), nickel (Ni), chromium (Cr) and cobalt (Co)), PCBs, pesticides (DDE), polycyclic aromatic hydrocarbons (PAHs), PBDEs, PFAS, hexabromcyclododecanes (HBCD), short and medium chained chlorinated paraffins (SCCP and MCCP), organophosphorus flame retardants (PFRs), bisphenol A (BPA), tetrabrombisphenol A (TBBPA), alkylphenols, phthalates as well as biological effects parameters. Analyses of triclosan, Diuron and Irgarol were included in this programme for the first time.

The results from 2014 (exclusive passive sampling) supplied data for a total of 2105 data sets (contaminant-station-species) on 136 different contaminants. Thirty representative contaminants and biological effect parameters were chosen for presentation in this report. This selection had 759 time series of which there were statistically significant time (2005-2014) related trends in 104 cases: 86 were downwards and 18 upwards. The downward trends were primarily associated with concentrations of metals and to a lesser degree tributyltin (TBT) and effect of TBT (VDSI - *vas deferens sequence index*). The dominance of downward trends indicated that contamination was decreasing. The upward trends were mainly associated with metals, primarily mercury.

Of the 403 cases that could be classified by the system of the Norwegian Environment Agency, 374 were classified as insignificantly polluted (Class I), 26 as moderately polluted (Class II), 2 as markedly polluted (Class III), none as severely polluted (Class IV) and 1 as extremely polluted (Class V). Even though most concentrations observed can be considered moderately polluted or better, the cases that were worse represent an environmental challenge and cannot be disregarded. For example the extremely polluted blue mussel in the Sørfjord for DDE.

Passive samplers were deployed at three sites and included investigations of alkylphenols, HBCD and PBDEs. The results were mostly below limits of detection (particularly for the Hvaler and Ålesund sites). Only BDE47, α-HBCD, para-t-octylphenol and para-t-nonylphenol could be measured in waters of Ålesund. Concentrations appear in line with data from the previous MILKYS reports.

Concentrations of contaminants in fish

Cod fillet from the Inner Oslofjord was moderately polluted by mercury, and a significant upward trend was found for the period 1984-2014 using the OSPAR method which targets specific length-groups. The method is robust but a more rigorous analytical method indicated that results on contaminant levels in cod may have been biased by increased fish length possibly due to, inter alia poor recruitment of cod in recent years and a need for larger fish to suit the analytical demands on fish liver. Using a more rigorous trend analysis which took into account fish size, no significant trend was detected for the entire period. Upward trends were also found in cod from Farsund and Bømlo using the OSPAR method.

Cod liver from the Inner Oslofjord was markedly polluted with PCBs. Contamination of cod was otherwise generally low (insignificantly or moderately polluted). The high concentrations of PCBs observed in cod liver in the Inner Oslofjord are probably related to urban activities in the past in combination with little water exchange with the outer fjord.

PBDEs have been investigated in cod liver for several fjords since 2005. In 2014, the concentration of sum PBDEs was highest in the Inner Oslofjord and was lowest in Lofoten. BDE47 was the dominant congener in all samples. As for PCB, the high concentrations of PBDEs are probably related to urban activities and water exchange conditions.

PFAS has been investigated in cod liver for several fjords since 2005. PFOS, an abundant PFAS, was highest in cod from the Inner Oslofjord and lowest in the Inner Trondheimsfjord. PFOSA, also an abundant PFAS, was highest in the Inner Oslofjord and lowest in Tromsø harbour. The differences between the stations cannot be fully explained, but it appears likely that as for PCBs and PBDEs a combination of urban sources and restricted water exchange provide the highest concentrations in the Inner Oslofjord.

Concentrations of contaminants in blue mussel

Blue mussel from one station in the Sørfjord was extremely polluted with DDE. Mussels from one station in the Hardangerfjord were markedly polluted with the same contaminant. Contamination of this substance is related to earlier use of DDT as pesticide in orchards along the fjords (ca.1945-1970).

Blue mussel from Odderøy in the Kristiansandsfjord was markedly polluted with hexachlorobenzene (HCB). Concentrations of PBDEs, α-HBCD and SCCP in mussels were highest at Nordnes in Bergen harbour area.

New contaminants

Of the hexabromcyclododecanes, α–HBCD was the most abundant diastereomer. Cod liver from the Inner Oslofjord had the highest median concentration of HBCD. The high concentrations of HBCD are probably related to urban activities, as well as a reduced water exchange with the outer fjord.

Of the chlorinated paraffins, concentrations of short chain chlorinated paraffins (SCCP) were significantly higher in blue mussel from the Bergen harbour compared to the other stations. Medium chlorinated paraffins (MCCP) in blue mussel was highest in the Grenlandsfjord area (Croftholmen) whereas MCCP in cod liver was highest in the Inner Trondheimsfjord and Bømlo. Mussels filter surface waters, whereas cod are generally exposed to deeper water masses, hence concentrations in these two organisms are not readily comparable. The specific sources of the SCCP and MCCP are unknown, but could be the result of industrial activity in these fairly enclosed areas. Further investigations are warranted.

Most concentrations of organophosphorus flame retardants (PFRs) were below the detection limits in blue mussel and cod, but no conclusions could be drawn regarding the differences among the stations.

Bisphenol A, TBBPA, alkylphenol, triclosan, Diuron and Irgarol were generally not detected in blue mussel or cod, and no conclusion can be drawn regarding possible differences between stations. There is an indication that of the four alkylphenols, 4-tert-nonylphenol and 4-n-octylphenol, were the most dominant compounds in blue mussel and cod liver.

Biological effects

The ICES/OSPARs assessment criterion[1] (background assessment criteria, BAC) for OH-pyrene in cod bile was exceeded at all four stations (Inner Oslofjord, Farsund area, Inner Sørfjord and Bømlo-Sotra area) in 2014 and indicates that the fish have been exposed to PAH. The median concentration of OH-pyrene metabolites in bile from cod in the Inner Oslofjord was about 30 % lower than in 2012 and 10 % lower than in 2013, but still above the ICES/OSPARs BAC.

The ALA-D activity in the Inner Oslofjord and the Inner Sørfjord in 2014 showed lower activity than at Bømlo. Reduced activities of ALA-D reflect higher exposure to lead.

The median concentration of CYP1A protein levels and EROD activity in the Inner Oslofjord was about half of the level observed in 2013, and resembled that observed in 2012. The concentration was still below the ICES/OSPARs BAC. Concentrations over BAC would indicate possible impact by planar PCBs, PCNs, PAHs or dioxins.

The effects from TBT on dog whelk were relatively low (VDSI<0.448) at all eight stations. There were significant downward trends for all stations, except for Brashavn where no significant trend could be seen and previous VDSI-levels were low. The results indicate that the legislation banning the use of TBT has been effective.

Stable isotopes

The stabile isotope $\delta^{15}N$ is analysed as a measure of trophic position. Results showed very similar isotopic signatures in 2012, 2013 and 2014, suggesting a persistent spatial trend more than a temporal trend. The $\delta^{15}N$ data in cod is assessed in relation to concentrations of selected contaminants. Generally, as fish grow through its lifetime, they feed on larger prey organisms, thus a small increase in trophic level is likely to occur. At specific stations, concentrations of mercury and PCB-153 (contaminants with well-known biomagnifying properties) increased with higher $\delta^{15}N$, i.e. higher concentrations in individuals with slightly higher trophic position.

Pooled samples

In the attempt to obtain sufficient cod liver material for analyses, samples are often pooled and the impact of the number of fish in pooled samples on the time trend analyses was examined. Pooled samples of equal sizes, i.e. with the same number of individuals, provides a better statistical basis for detecting trends than pools of unequal sizes.

Dealing with small cod liver samples

The sometimes difficult task of obtaining sufficient cod liver material for the various analyses, was addressed by looking at some alternative sampling strategies. Moderate extention of catch time or small scale local extention of catch area (e.g. less than 10 km) to gather more fish of adequate

[1] Assessment criteria have specifically been compiled for the assessment of CEMP monitoring data on hazardous substances. They do not represent target values or legal standards.

size could be one approach. The results showed that this could also enhance the means to detect trends in cod. Though on a larger scale of extension (e.g. hundreds of kilometers) there can be local influences on particular stations, and if these are not accounted for more regional assessments of trends can be misleading.

Cod fillet and blood could potentially be used as an alternative to liver for monitoring PCBs (fillet), chlorinated paraffins (fillet) and PFAS (blood). The use of other fish species and passive sampling or lowering the limit of quantification (LOQ) was also discussed.

Sammendrag

Denne undersøkelsen omhandler nivåer, trender og effekter av miljøgifter langs norskekysten. I tillegg til en mer langsiktig overvåking er det også gjort analyser av enkelte nyere miljøgifter som har fått større oppmerksomhet de senere årene. Undersøkelsen gir grunnlag for vurdering av miljøstatus for miljøgifter langs kysten. Resultatene viser at det hovedsakelig var nedadgående trender for forekomst av de undersøkte miljøgiftene.

Indre Oslofjord er et område med forhøyede miljøgiftkonsentrasjoner som gir grunnlag for bekymring og behov for nærmere undersøkelser. For eksempel observeres oppadgående trend for kvikksølv (Hg) i torskefilet og høye konsentrasjoner av bly (Pb) polyklorerte bifenyler (PCB), polybromerte difenyletere (PBDE), perfluorerte alkylstoffer (PFAS) og alfa-heksabromsyklododekan (α–HBCD) i torskelever.

Undersøkelsen inngår som en del av OSPARs koordinerte miljøovervåkingsprogram Coordinated Environmental Monitoring Programme *(CEMP). I 2014 omfattet overvåkingen miljøgifter i blåskjell (32 stasjoner), purpursnegl (9 stasjoner), strandsnegl (én stasjon), torsk (14 stasjoner) og sjøvann (bruk av passive prøvetakere på tre stasjoner) langs norskekysten fra Oslofjord-Hvaler området i sørøst til Varangerfjorden i nordøst. Stasjonene er plassert både i områder med kjente eller antatt kjente punktkilder for tilførsler av miljøgifter, i områder med diffus tilførsel av miljøgifter slik som byens havneområder, og i fjerntliggende områder med antatt lav eksponering for miljøgifter. Undersøkelsen omfatter overvåking av metaller (Hg, kadmium (Cd), Pb, kobber (Cu), sink (Zn), sølv (Ag), arsen (As), nikkel (Ni), krom (Cr) og kobolt (Co)), tributyltinn, PCBer, pestisider (DDE), PBDEer, PFAS, heksabromsyklododekan (HBCD), korte- og mellomkjedete klorparafiner (SCCP og MCCP), fosfororganiske flammehemmere (PFRer), bisfenol A (BPA), tetrabrombisfenol A (TBBPA), alkyfenoler, ftalater samt biologiske effekt parametre. For første gang er det inkludert analyser av triklosan, Diruon og Irgarol i overvåkingen.*

2014-resultatene (eksklusive passive prøvetakere) omfatter totalt 2105 datasett (miljøgifter-stasjoner-arter) for 136 forskjellige miljøgifter. Et utvalg på 30 representative miljøgifter og biologiske parametere presenteres i denne rapporten. Dette utvalget består av 759 tidsserier hvorav 104 viste statistisk signifikante trender for perioden 2005 til 2014: 86 var nedadgående og 18 var oppadgående. De nedadgående trendene omfattet primært metaller og i noe mindre grad også tributyltinn (TBT) og effekt av TBT (VDSI – sædlederindeks). Dominansen av nedadgående trender indikerer avtagende nivåer av miljøgifter. De oppadgående trendene var i hovedsak metaller, primært kvikksølv.

Av de 403 tidsseriene som kunne klassifiseres i henhold til Miljødirektoratets klassifiseringssystem, var 374 klassifisert som ubetydelig-lite forurenset (klasse I), 26 som moderat forurenset (klasse II), to som markert forurenset (klasse III), ingen som sterkt forurenset (klasse IV) og én som meget sterkt forurenset (klasse V). Selv om det fleste observerte nivåene kan betraktes som moderat forurenset eller bedre, så var det noen observasjoner som viste en sterkere grad av forurensing og som dermed utgjør en miljøutfordring som en ikke kan se bort ifra. Et eksempel på dette er blåskjell i Sørfjorden som var meget sterkt forurenset av DDE.

Passive prøvetakere ble utplassert tre steder (Hvaler, Oslofjorden og Ålesund havn) og inkluderte undersøkelser av alkylfenoler, HBCD og PBDEer. Resultatene var stort sett under deteksjonsgrensen (særlig for prøver fra Hvaler og Ålesund). Bare i vann fra Ålesund havn og for BDE47, α-HBCD, para-t-octylfenol og para-t-nonylfenol ble det observert konsentrasjoner over deteksjonsgrensen. De påviste konsentrasjonene samsvarer med tidligere rapporterte data.

Konsentrasjoner av miljøgifter i fisk

Torskefilet fra indre Oslofjord var moderat forurenset av kvikksølv og det var en signifikant oppadgående trend for perioden 1984-2014. Trendberegningene er gjort etter en OSPAR metode basert på spesifikke lengde-grupper av fisk. Denne metoden anses som robust. En mer omfattende analytisk metode der en i større grad tar hensyn til lengden til enkeltfisk viste imidlertid ingen signifikant trend for hele perioden. Bakgrunnen for at de to analysene gir noe forskjellig resultat er at de i ulik grad tar hensyn til fiskens lengde og at en de senere årene har måttet benytte større individer av torsk for å sikre tilstrekkelig materiale til analysene. En oppadgående trend ble registrert i torsk fra Farsund og Bømlo ved bruk av OSPAR-metoden.

Torskelever fra indre Oslofjord var markert forurenset av PCBer. Torsk var ellers generelt lite forurenset (ubetydelig eller moderat forurenset) av denne gruppe forbindelser. De høye konsentrasjonene av PCBer som ble observert i torskelever fra indre Oslofjord har trolig sammenheng med urbane aktiviteter i kombinasjon med lav vannutskifting med ytre fjord.

PBDEer) er undersøkt i torskelever fra flere fjorder siden 2005. I 2014 var konsentrasjonen av sum PBDEer høyest i torsk fra indre Oslofjord og lavest i Lofoten. BDE47 var den dominerende av PBDEene i alle prøvene. Som for PCBer, er urban aktivitet og vannutskiftingsforhold trolig årsaker til de høye nivåene.

PFAS har blitt undersøkt i torskelever fra flere fjorder siden 2005. PFOS, en PFAS-forbindelse, var høyest i torskelever fra indre Oslofjord og lavest i indre Trondheimsfjord. PFOSA, også en PFAS-forbindelse, var høyest i indre Oslofjord og lavest i Tromsø havn. Nivåforskjellene mellom de ulike områdene kan foreløpig ikke forklares fullt ut, men det er sannsynlig at en kombinasjon av urbane kilder og begrenset vannutskifting gir de høyeste konsentrasjonene i indre Oslofjord, slik som resultatet var for PCBer og PBDEer.

Konsentrasjoner av miljøgifter i blåskjell

Blåskjell fra én stasjon i Sørfjorden var meget sterkt forurenset av DDE. I Hardangerfjorden var blåskjell fra én stasjon markert forurenset av den samme miljøgiften. Forurensning av denne miljøgiften skyldes tidligere bruk av DDT som sprøytemiddel i frukthager langs fjordene (ca. 1945-1970).

Blåskjell fra Odderøy i Kristiansandsfjorden var markert forurenset med heksaklorbenzen (HCB). Konsentrasjoner av PBDEer, α-HBCD og SCCP var høyest i blåskjell fra Nordnes i Bergen havneområde.

Nye miljøgifter

Av heksabromsyklododekaner var α-HBCD den mest dominerende diastereomeren. Torskelever fra indre Oslofjord hadde den høyeste median-konsentrasjonen av HBCD. De høye HBCD-konsentrasjonene er sannsynligvis relatert til urbane aktiviteter, samt lav vannutskifting med ytre fjord.

Det var signifikant høyere nivå av kortkjedete klorerte parafiner (SCCP) i blåskjell fra Bergen havn sammenlignet med de andre stasjonene. Mellomkjedete klorerte parafiner (MCCP) i blåskjell var høyest i Grenlandsfjorden (Croftholmen) mens MCCP i torskelever var høyest i indre Trondheimsfjord og Bømlo. Blåskjell filtrerer overflatevann, mens torsk generelt er eksponert for dypere vannmasser, derfor vil eksponeringen kunne være forskjellig og en kan ikke vente at konsentrasjonene i disse to organismene gir samme relative bilde av forurensningsnivå. De

spesifikke kildene til SCCP og MCCP er ukjent, men kan være et resultat av industriell aktivitet i disse relativt lukkede områdene. Dette bør undersøkes nærmere.

De aller fleste konsentrasjonene av fosfororganiske flammehemmere (PFRer) var under deteksjonsgrensene i blåskjell og torsk. Nivåene anses derfor som generelt lave, men ingen konklusjoner kan trekkes når det gjelder forskjeller mellom stasjonene.

Bisfenol A, TBBPA, alkylfenol, triclosan, Diruon og Irgarol ble i hovedsak ikke påvist i blåskjell eller torsk. Nivåene anses derfor som generelt lave men ingen konklusjon kan trekkes vedrørende mulige forskjeller mellom stasjonene. Resultatene tyder på at de fire alkyfenolene, 4-tert-nonylfenol og 4-n-oktylfenol var de mest dominerende.

Biologiske effekter
ICES/OSPARs vurderingskriterium for bakgrunnsnivå[1] («background assessment criteria»*, BAC) for OH-pyren i torskegalle ble overskredet på alle de fire stasjonene (indre Oslofjord, Farsund området, indre Sørfjord og Bømlo-Sotra området) i 2014, og dette viser at fisken har vært eksponert for PAH. Median-konsentrasjonen av OH-pyren metabolitter i torskegalle fra indre Oslofjord var ca. 30 % lavere enn i 2012 og 10 % lavere enn i 2013, men var fortsatt over ICES/OSPARs BAC.*

ALA-D aktivitet i indre Oslofjord og indre Sørfjorden i 2014 var lavere enn på Bømlo. Redusert aktivitet av ALA-D tyder på høyere eksponering for bly.

Nivåene av CYP1A protein og EROD-aktivitet i indre Oslofjord var omtrent halvparten av nivået i 2013, og mer lik nivået i 2012. Konsentrasjonen var fortsatt under ICES/OSPARs BAC. Konsentrasjoner over BAC indikerer mulig effekt av plane PCBer, PCNer, PAHer eller dioksiner.

Effektene av TBT på purpursnegl var relativt lave (VDSI <0.448) på alle de åtte stasjonene. Det var signifikant nedadgående trender på alle stasjonene bortsett fra ved Brashavn der ingen signifikant trend kunne ses og tidligere VDSI-nivåer har vært lave. Resultatene indikerer at forbudet mot bruk av TBT har vært effektivt.

Stabile isotoper
Den stabil isotopen $\delta^{15}N$ er analysert for å tolke en organismes posisjon i næringskjeden. Resultatene viste veldig like isotop-signaturer i 2012, 2013 og 2014. Data for stabile isotoper ($\delta^{15}N$) i torsk er vurdert i sammenheng med konsentrasjoner av utvalgte miljøgifter. I hovedsak spiser fisk større byttedyr etterhvert som de vokser, og dette medfører ofte overgang til høyere trofisk nivå. Det ble funnet økende konsentrasjon av kvikksølv og PCB-153 (miljøgifter med kjente biomagnifiserende egenskaper) med økende nivå av $\delta^{15}N$, dvs. høyere konsentrasjoner i individer på noe høyere trofisk nivå.

Bland prøver
For å få tilstrekkelig torskelever materiale til analyse lages det ofte blandprøver med materiale fra flere fisk. Hvilke effekt dette har på tidstrend analyser ble undersøkt. Resultatene tyder på at blandprøver av lik størrelse, m.a.o. med materiale fra samme antall individer, gir en bedre statistisk basis for å detektere trender enn blandprøver av ulike størrelser.

[1] Vurderingskriteriene er spesielt utarbeidet for vurdering av CEMP-overvåkingsdata for farlige forbindelser. De representerer ikke målverdier eller juridiske standarder.

Håndtering av små torskelever prøver

Ettersom det kan være vanskelig å få tilstrekkelig med vev (særlig torskelever) til alle analysene som ønskes, ble alternative strategier vurdert. En moderat forlengelse av fangstperioden eller en små skala utvidelse av fangstområde (f.eks. mindre enn 10 km) vil kunne være en måte å fange flere fisk av ønsket størrelse. Resultatene viste at dette også kunne øke muligheten for å detektere trender. Med større utvidelser (f.eks. flere hundre kilometer) vil lokale påvirkninger og eventuelle forskjeller i fiskens adferd kunne forstyrre bildet og gi misvisende trendanalyser.

Torskefilet og torskeblod kan potensielt brukes som alternativt til torskelever for overvåking av PCBer (filet), klorinerte parafiner (filet) og PFAS (blod). Bruk av andre fiskearter og passive prøvetakere eller senkning av deteksjonsgrense (LOQ) ble også vurdert.

Contents

1. Introduction

1.1 Background

The programme "Contaminants in coastal waters of Norway" (*Miljøgifter i norske kystområder* - MILKYS) is administered by the Norwegian Environment Agency (*Miljødirektoratet*). The programme focuses on the levels, trends and effects of hazardous substances in fjords and coastal waters, which also represents the Norwegian contribution to the Coordinated Environmental Monitoring Programme (CEMP). CEMP is a common European monitoring programme under the auspices of Oslo and Paris Commissions (OSPAR). The Norwegian contribution to CEMP addresses several aspects of OSPAR's assessment of hazardous substances. All the results in this report are considered part of the Norwegian contribution to the CEMP programme.

The objective for the performed monitoring is to obtain updated information on levels and trends of selected hazardous substances known or suspected to have a potential for causing detrimental biological effects.

Concentrations of hazardous substances in sediment, pore water, mussels and fish constitute time-integrating indicators for the quality of coastal water. Many of these substances have a tendency to accumulate in tissues (bioaccumulation) in the organisms, and show higher concentrations relative to their surroundings (water and in some cases also sediment). Hence, it follows that substances may be detected, which would otherwise be difficult to detect when analysing water or sediment only. Using concentrations in biota as indicators, as opposed to using water or sediment, are of direct ecological importance as well as being important for human health considerations and quality assurance related to commercial interests involved in harvesting marine resources.

MILKYS applies the OSPAR CEMP methods. These OSPAR methods suggest *inter alia* monitoring of blue mussel, snails and Atlantic cod on an annual basis.

An overview of MILKYS stations in Norway is shown in maps in Appendix D. The program has included monitoring in sediment (cf. Green *et al.* 2010a) and to a larger degree biota, the main emphasis being:

- Oslofjord-area, including the Hvaler area, Singlefjord and Grenlandsfjord area, since 1981.
- Sørfjord/Hardangerfjord since 1987.
- Orkdalsfjord area and other areas in outer Trondheimsfjord, 1984-1996 and 2004-2005.
- Arendal and Lista areas since 1990.
- Lofoten area since 1992.
- Coastal areas of Norway's northern most counties Troms and Finnmark since 1994.

The previous investigations have shown that the Inner Oslofjord area has elevated levels of polychlorinated biphenyls (PCBs) in cod liver, mercury, lead and zinc in sediments and moderately elevated concentrations of mercury in cod fillet. Investigations of the Sørfjord/Hardangerfjord have shown elevated levels of PCBs, dichlorodiphenyltrichloroethane (DDT, using dichlorodiphenyldichloroethylene (DDE) - principle metabolite of DDT as an indicator), cadmium, mercury and lead. Investigations in Orkdalsfjord focused on three blue mussel stations. The results from these investigations have been reported earlier (Green *et al.* 2007, Green & Ruus 2008). It can be noted that environmental status is classified according to environmental quality criteria based on the classification system of the Norwegian Environment Agency (Molvær *et al.* 1997), or

presumed background levels (Appendix C) and must not be confused with limit values for human consumption and associated advice issued by the Norwegian Food Safety Authorities. Furthermore, EU Water Framework Directive (WFD) of 2000 (2000/60/EC) entered into Norwegian law in 2005 to which the MILKYS programme has had to take into account.

In addition to the monitoring of Oslofjord area and Sørfjord/Hardangerfjord, MILKYS also includes the annual monitoring of contaminants at selected stations in Lista and Bømlo areas on the south and west coast of Norway, respectively. During the periods 1993-1996 and 2006-2007, MILKYS also included sampling of blue mussel from reference areas along the coast from Lofoten to the Russian border. The sampling also includes fish from four key areas north of Lofoten in the Finnsnes-Skjervøy area, Hammerfest-Honningsvåg area, and Varanger Peninsula area. Fish from the Lofoten and Varanger Peninsula areas are sampled annually. The intention is to assess the level of contaminants in reference areas, areas that are considered to be little affected by contaminants, and to assess possible temporal trends.

Biological effects methods, BEM or biomarkers were introduced in the Norwegian MILKYS in 1997. The purpose of these markers is, by investigations on molecular/cell/individual level, to give warning signals if biota is affected by toxic compounds and to assist in establishing an understanding of the specific mechanisms involved. The reason to use biological effects methods within monitoring programmes is to evaluate whether marine organisms are affected by contaminant inputs. Such knowledge cannot be derived from tissue levels of contaminants only. One reason is the vast number of chemicals (known and unknown) that are not analysed. Another reason is the possibility of combined effects (“cocktail effects”) of multiple chemical exposures. In addition to enabling conclusions on the health of marine organisms, some biomarkers assist in the interpretation of contaminant bioaccumulation. The biological effects component of MILKYS includes imposex in snails as well as biomarkers in fish. The methods were selected for specificity as to which contaminants impact the parameter and robustness.

The state of contamination is divided into three issues of concern: levels, trends and effects. Different monitoring strategies are used, in particular with regard to the selection of indicator media (blue mussel, snail, cod liver etc.) and selection of chemical analyses. Sample frequency is annual for biota). The programme underwent an extensive revision in 2012, both in regard to stations and chemical analyses. Monitoring of flatfish was discontinued but three more cod-stations were added bringing the total to 15. The blue mussel stations were reduced from 38 to 26. Choice of chemical analyses for each station has changed considerably after 2011 (Appendix E). Pesticide and dioxin analyses were discontinued except for DDTs at some stations in the Sørfjord/Hardangerfjord. However, many new analyses were added, including analyses of: short- and medium chain chlorinated paraffins (SCCP and MCCP), phenols (e.g. bisphenol A, tetrabrombisphenol A), organophosphorus flame retardants and stabile isotopes. The Norwegian Pollution and Reference Indices (cf. Green *et al*. 2011b, 2012a) are not included in the revised programme but passive sampling of contaminants in water has been added.

The change in the programme has meant that many time series were discontinued since 2012. However independent funding from the Norwegian Ministry of Climate and Environment ensured that some of these time series have been maintained after 2012. This involved extra analyses (mostly pesticides) of MILKYS-samples, and collection and analyses of some blue mussel and flatfish stations that otherwise would have been discontinued. This additional funding also ensured that investigation of biological effect in cod from the Inner Sørfjord and from Bømlo on the West Coast could be continued. The results for blue mussel and cod from these investigations are included in this report.

Where possible, MILKYS is integrated with other national monitoring programmes to achieve a better practical and scientific approach for assessing the levels, trends and effects of micropollutants. In particular, this concerns sampling for the Norwegian sample bank, a programme funded by the Norwegian Ministry of Climate and Environment to sustain time trend monitoring and local (county) investigations. There is also coordination with Comprehensive Study on Riverine Inputs and Direct Discharges (RID) and The Norwegian Costal Monitoring Programme (*Kystovervåkingsprogrammet*, KYO). Both programmes are operated by NIVA on behalf of Norwegian Environment Agency.

1.2 Purpose

An aim of the Norwegian Environment Agency is to obtain an overview of the status and trends of the environment as well as to assess the importance of various sources of pollution. The Norwegian Environment Agency seeks to develop a knowledge-base for the public and for the management of the environment.

The programme Contaminants in Coastal Waters of Norway (MILKYS) is used as a tool to promote cessation of discharges, emissions and losses of hazardous substances by the year 2020. This will be accomplished through:

1. Monitoring the levels of a selection of hazardous substances in biota and water;
2. Evaluating the bioaccumulation of priority hazardous substances in biota of coastal waters;
3. Assessing the effectiveness of previous remedial action;
4. Considering the need for additional remedial action;
5. Assessing the risk to biota in coastal waters;
6. Fulfilling obligations to regional sea convention (OSPAR).

MILKYS is part of the Norwegian contribution to CEMP and is designed to address issues relevant to OSPAR (OSPAR 2014) including OSPAR priority substances (OSPAR 2007). The programme will also contribute to the demands on Norway by the EU Water Framework Directive (WFD) (2000/60/EC) and its daughter directive the Environmental Quality Standards Directive (EQSD – 2013/39/EU) to achieve good chemical and ecological status. The results from MILKYS can also be useful in addressing aspects of the EU's Marine Strategy Framework Directive (MSFD) (2008/56/EC). One of the goals of WFD and MSFD is to achieve concentrations of hazardous substances in the marine environment near background values for naturally occurring substances and close to zero for manmade synthetic substances. OSPAR has also adopted this goal (OSPAR 1998).

2. Material and methods

2.1 Sampling

2.1.1 Stations

Samples for the investigation of contaminants were collected along the Norwegian coast, from the Swedish border in the south to the Russian border in the north (***Figure 1***, ***Figure 2***, ***Figure 3***, Appendix D). The sampling involved blue mussel at 32 stations (34 were planned) and these include eight funded directly by the Ministry of Climate and Environment (see Chapter 1.1), dog whelk at eight stations (nine were planned), periwinkle at one station and cod at 14 stations where 15 stations were planned. In addition, contaminants in seawater were investigated using passive sampling at three stations.

Samples were collected annually and analysed according to OSPAR guidelines (OSPAR 2003b, 2012)[1]. The data was screened and submitted to ICES by agreed procedures (ICES 1996). Blue mussel, snails (dog whelk and periwinkle) and Atlantic cod are the target species selected for MILKYS to indicate the degree of contamination in the sea. Blue mussel is attached to shallow-water surfaces, thus reflecting exposure at a fixed point (local pollution). Mussels and snails are abundant, robust and widely monitored in a comparable way. The species are, however, restricted to the shallow waters of the shore line. Cod is a widely distributed and commercially important fish species. It is a predator and, as such, will reflect contamination levels in their prey.

As mentioned above (see Chapter 1.1) the results from some supplementary monitoring to maintain long-term trends are included in this report. These concern some contaminants in blue mussel and cod (cf. ***Table 2***).

Some details on methods applied in previous years of monitoring are provided in Green *et al*. 2014.

[1] See also http://www.ospar.org/work-areas/hasec

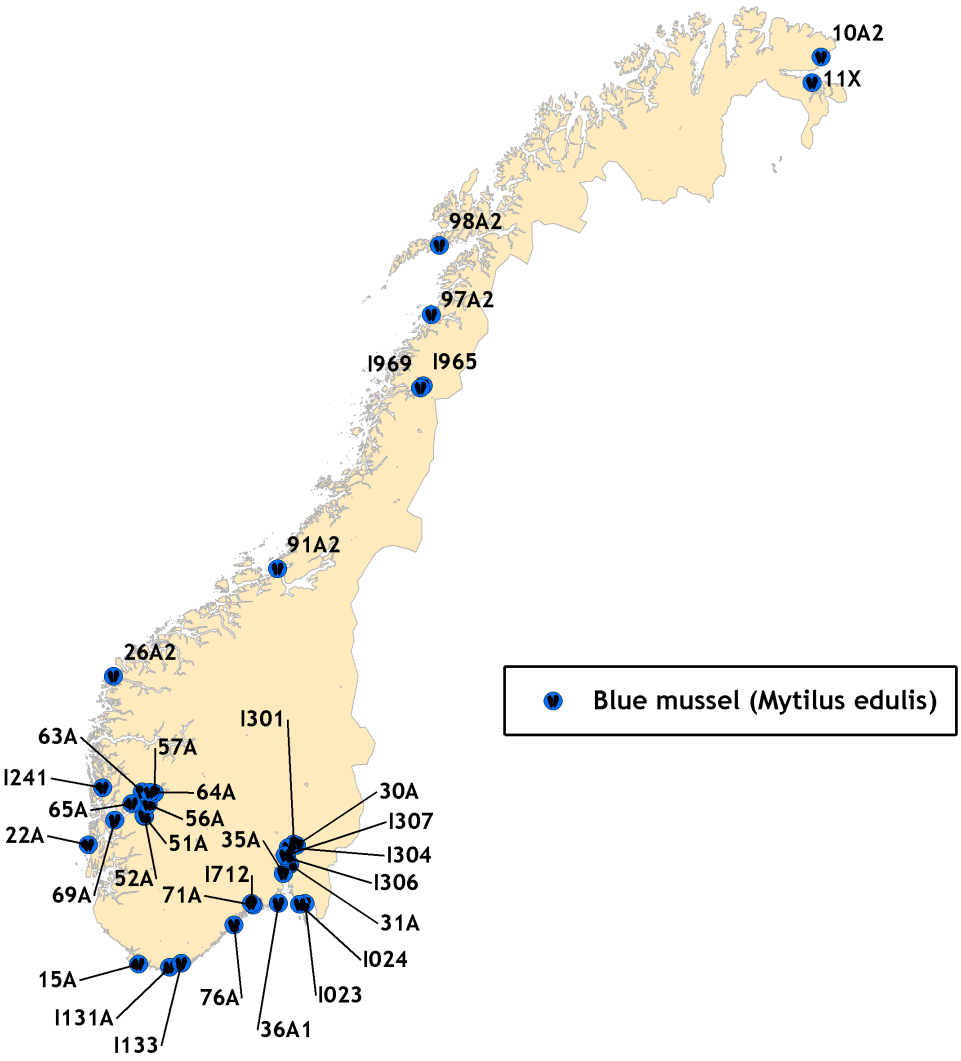

***Figure 1**. Stations where blue mussel were sampled in 2014. See also station information in detailed maps in Appendix D.*

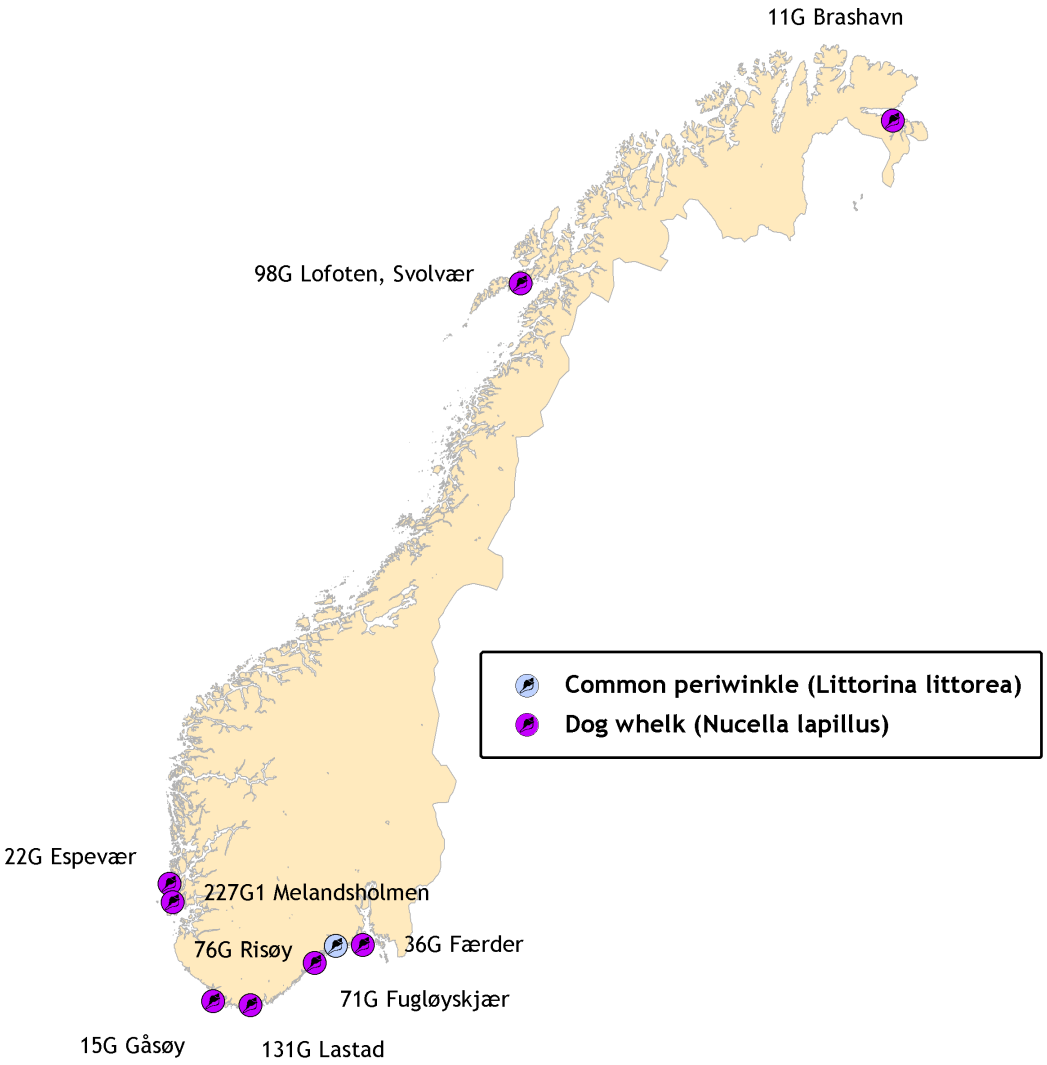

***Figure 2**. Stations where dog whelk and periwinkle were sampled in 2014. See also station information in detailed maps in Appendix D.*

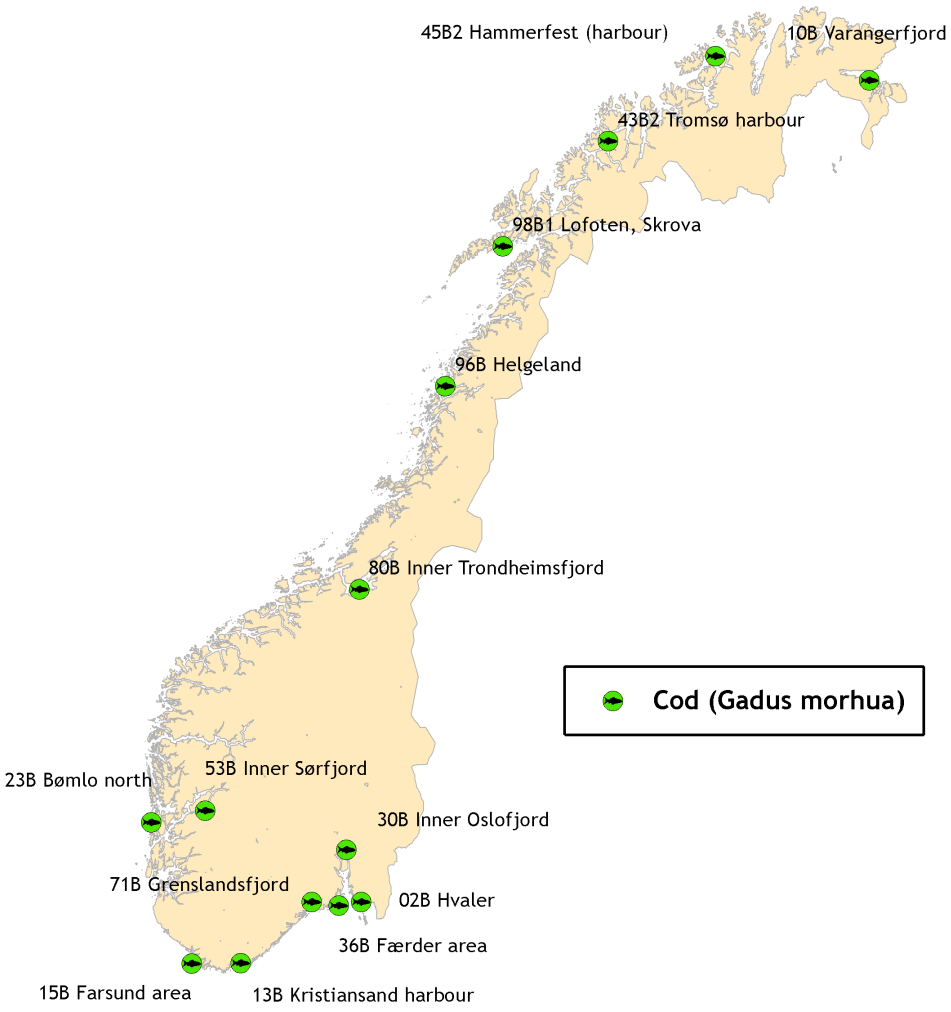

***Figure 3**. Stations where cod were sampled in 2014. Note that biological effects methods were applied to cod samples from the Inner Oslofjord. See also station information in detailed maps in Appendix D.*

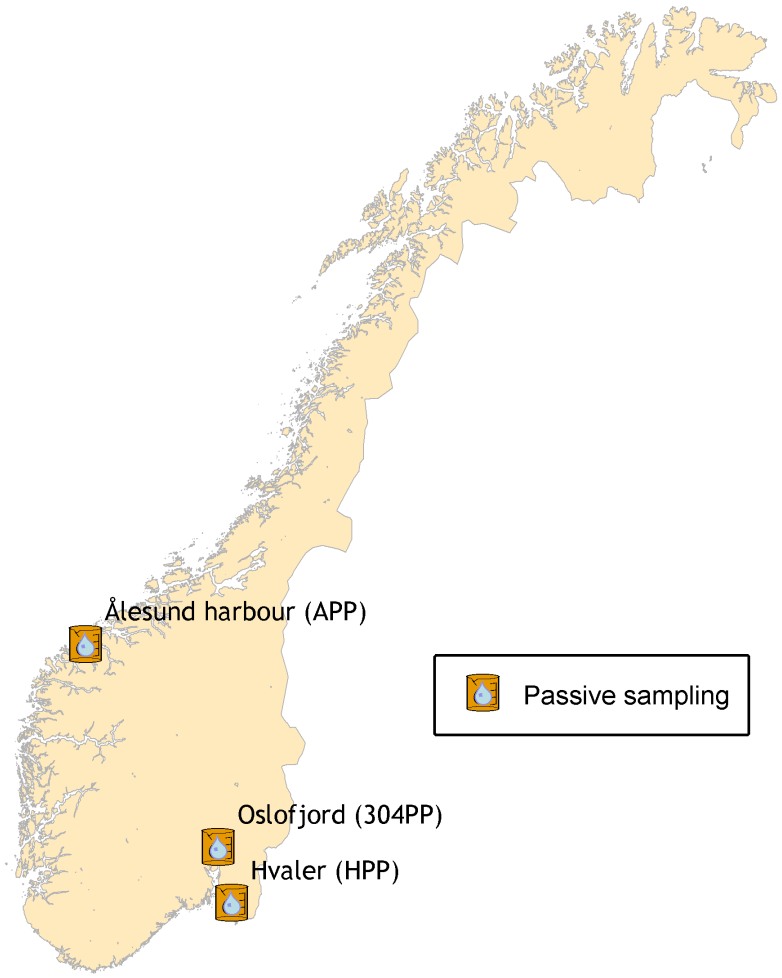

***Figure 4**. Stations where passive sampling was employed in 2013-2015.*

2.1.2 Blue mussel

Sufficient sample of blue mussel (*Mytilus edulis*), both with respect to count and mass, were found at 32 of the 34 stations planned (including eight funded directly by the Ministry of Climate and Environment). The stations are located as shown in ***Figure 1*** (see also maps in Appendix D). The stations were chosen to represent highly polluted or reference locations distributed along the Norwegian coast. It has been shown that the collected species are not all *Mytilus edulis* (Brooks & Farmen 2013) but possible differences in contaminant uptake were assumed to be negligible and not taken into account for this investigation.

The blue mussel samples were collected from 3 September to 12 November 2014.

Generally, blue mussel was not abundant on the exposed coastline from Lista (southern Norway) to the north of Norway. A number of samples were collected from dock areas, buoys or anchor lines. All blue mussels were collected by NIVA except for the blue mussels collected in the Ranfjord, Lofoten and Varangerfjord, which were collected by local contacts.

Three pooled samples of 20 individuals each were collected in the size range of 3-5 cm. Shell length was measured by slide callipers. The blue mussel was scraped clean on the outside by using knives or scalpels before taking out the tissue for the analysis. Mussels were shucked and frozen (-20°C).

For certain stations and prior to the 2012-investigations the intestinal canal was emptied (depuration) in mussels following OSPAR guidelines (OSPAR 2012, cf. Green *et al.* 2012a). There is some evidence that for a specific population/place the depuration has no significant influence on the body burden of the contaminants measured (cf. Green 1989; Green *et al.* 1996, Green *et al.* 2001). This practice was discontinued in 2012.

2.1.3 Dog whelk and periwinkle

Concentrations and effects of organotin were investigated at eight stations for dog whelk (*Nucella lapillus*) and one station for periwinkle (*Littorina littorea*) (***Figure 2***, see also maps in Appendix D). TBT-induced development of male sex-characters in female dog whelks, known as imposex, was quantified by the *Vas Deferens Sequence Index* (VDSI) analysed according to OSPAR-CEMP guidelines. The VDSI ranges from zero (no effect) to six (maximum effect) (Gibbs *et al.* 1987). Detailed information about the chemical analyses of the animals is given in Følsvik *et al.* (1999).

Effects (imposex, ICES1999) and concentrations of organotin in dog whelk were investigated using 50 individuals from each station. Individuals were kept alive in a refrigerator (at +4°C) until possible effects (imposex) were quantified. All snails were sampled by NIVA except for the dog whelk collected in Lofoten and in the Varangerfjord. The snail samples were collected from 5 August to 5 November 2014.

2.1.4 Atlantic cod

Fifteen individuals of Atlantic cod (*Gadus morhua)* were to be sampled for each station. This was accomplished at 14 stations, except at Hvaler (st. 02B) (***Figure 3***) where only 8 individuals were caught.

The cod were sampled from 25 August to 13 December 2014. All the cod were sampled by local fishermen except for the cod in the Inner Oslofjord (st. 30B) that was collected by NIVA by trawling from the research vessel *F/F Trygve Braarud* owned and operated by the University of Oslo. Instructions were given to the fisherman to catch coastal cod. Coastal cod is more attached to one place than open ocean cod which migrate considerably farther than coastal cod. Some spot checks were taken using otoliths which confirmed, at least for these samples, that only coastal cod were caught. The otoliths are stored for further verification if necessary. If possible cod were sampled in five length classes (***Table 1***), three individuals in each class. Tissue samples from each fish were prepared in the field and stored frozen (-20°C) until analysis or the fish was frozen directly and later prepared at NIVA.

Table 1*. Target length groups for sampling of cod.*

Size-class	Cod (mm)
1	370-420
2	420-475
3	475-540
4	540-615
5	615-700

Livers were in general not large enough to accommodate all the analyses planned (see Appendix E). The Ullerø area, Hammerfest harbour, Inner Trondheimsfjord and Sandnessjøen area were the four stations where all 15 individuals had sufficient liver size to complete all of the intended analyses. The general lack of material was partially compensated for by making pooled samples of livers. These are noted in the tables below. The concerns using pooled samples or small sample size in cod are discussed in Chapter 3.8 and 3.9, respectively.

The age of the fish was determined by noting the number opaque and hyaline zones in otoliths.

2.2 Chemical analyses of biological samples

2.2.1 Choice of chemical analyses and target species/tissues

An overview of chemical analyses performed on 2014-samples as well as supplementary 2013-samples is shown in *Table 2*. Note that the table also includes an overview of some supplementary investigations funded by the Ministry of Climate and Environment that are relevant to this report.

***Table 2**. Analyses and target organisms of 2014 and supplementary analyses of 2013 samples. The value indicates the total number of stations investigated of which those funded by the Ministry of Climate and Environment as a supplement are indicated in parentheses*.*

Parameter	Blue mussel	Dog whelk	Common periwinkle	Cod fillet	Passive samplers	Cod liver
Metals	32 (8)					13
Cadmium (Cd), copper (Cu), lead (Pb), zinc (Zn), silver (Ag), arsenic (As), chrome (Cr), nickel (Ni), cobalt (Co) and tin (Sn)						
Mercury (Hg)	32 (8)			14		
Total-Hg						
Organotin	8(8)	8	1			
monobutyltin (MBT), dibutyltin (DBT), tributyltin (TBT), trifenyltin (TPT)						
PCB-7	29 (8)					13
PCB-28, 52, 101, 118, 138, 153, and 180						
HCB, OCS, 5CS	0 (15)					0 (7, 18)
∑DDT	19 (15)					7 (6)
p-p`-DDT, p-p`-DDE, p-p`-DDD						
PAH-16	11					
Polybrominated diphenyl ethers (PBDEs)	10				3	8
BDE47, 99, 100, 126, 153, 154, 183, 196 and 209						
Hexabromcyclododecane (HBCDs)	10				3	10
α, ß, γ-HBCD						
Perfluorinated alkylated substances (PFAS)						8
PFNA, PFOA, PFHpA, PFHxA, PFOS, PFBS, PFOSA						
Chlorinated paraffins	10					10
SCCP (C10-C13) and MCCP (C14-C17)						
Phosphorus flame retardants (PFRs)	10					10
tri-iso-butylphosphate (TIBP)						
tributylphosphate (TBP)						
tri(2-chlorethyl)phosphate (TCEP)						
tri(1-chlor-2-propyl)phosphate (TCPP)						
tri(1,3-dichlor-2-propyl)phosphate (TDCP)						
tri(2-butoxyethyl)phosphate (TBEP)						
triphenylphosphate (TPhP)						
2-ethylhexyl-di-phenylphosphate (EHDPP)						
tetrekis-(2-chloroethyl)dichlorisopentyldiphosphate (V6)						

Parameter	Blue mussel	Dog whelk	Common periwinkle	Cod fillet	Passive samplers	Cod liver
dibutylphenylphosphate (DBPhP)						
butyldiphenylphosphate (BdPhP)						
tris(2-ethylhexyl)phosphate (TEHP)						
tris-o-cresylphosphate (ToCrP)						
tricresylphosphate (TCrP)						
Alkylphenol	10				3	9
Octylphenol, nonylphenol						
Tetrabrombisphenol A (TBBPA)	10					9
Bisphenol A (BPA)	10					9
Supplementary analyses for 2013 (or earlier [2]) samples						
Phthalates *(44 samples)*	*4*					*4*
PBDEs [1] *(9 samples)*	*3*					
SCCP, MCCP *(124 samples)* [2]						*2*
Alkylphenols *(19 samples)*	*4*					*1*
Triclosan *(36 samples)*	*3*					*4*
Diuron, Irgarol *(40 samples)*	*6*					*4*

*) Supplementary investigations funded by the Ministry of Climate and Environment involved additional analyses on samples from blue mussel stations 30A, I301, I304, 31A, 36A1, 71A, I712, 51A, 56A, 65A, 22A, 10A2 and 11X; cod stations30B, 36B, 15B, 53B, 23B, 98B1 and 10B; as well as all analyses for blue mussel stations: 52A, 57A, 63A, 69A, I133, I306, I307.

[1]) Including: BDE28, -47, -99, -100, -153, -154, -183, -196, -202, -206, -207 and -209.

[2]) SCCP and MCCP: West coast station (st. 23B in 1994, 1997, 2005, 2013), Inner Sørfjord (st. 53B in 1990, 1994, 1997, 2000, 2005, 2009).

An overview of the applied analytic methods is presented in ***Table 3***. Chemical analyses were performed separately for each cod liver, if possible, otherwise a pooled sampled was taken (see «count» for the relevant tables, e.g. ***Table 11***). Mercury was analysed on a fillet sample from each cod. Furthermore, Biological Effects Methods (BEM) were performed on individual cod.

***Table 3**. Overview of method of analyses (see Appendix B for description of chemical codes). Limit of detection (LOD) or limit of quantification (LOQ1) is indicated. See 2.2.2 for description of the labs used for the different analysis.*

Name	[CAS-number]	Lab.	LOD	LOQ1	Est. uncertainty	Standard or internal method	Accreditation status
Metals							
cadmium (Cd)	7440-43-9	**NIVA/EFM**		0.001 mg/kg	20 %	Standard method NS EN ISO 17294-2	ISO 17025, accredited
copper (Cu)	7440-50-8	**NIVA/EFM**		0.03 mg/kg	20 %	Standard method NS EN ISO 17294-2	ISO 17025, accredited
lead (Pb)	7439-92-1	**NIVA/EFM**		0.03 mg/kg	20 %	Standard method NS EN ISO 17294-2	ISO 17025, accredited
zinc (Zn)	7440-66-6	**NIVA/EFM**		0.5 mg/kg	20 %	Standard method NS EN ISO 17294-2	ISO 17025, accredited
silver (Ag)	7440-22-4	**NIVA/EFM**		0.03 mg/kg	20 %	Standard method NS EN ISO 17294-2	ISO 17025, accredited
arsenic (As)	7440-38-2	**NIVA/EFM**		0.03 mg/kg	20 %	Standard method NS EN ISO 17294-2	ISO 17025, accredited
chrome (Cr).	7440-47-3	**NIVA/EFM**		0.02 mg/kg	20 %	Standard method NS EN ISO 17294-2	ISO 17025, accredited
nickel (Ni)	7440-02-0	**NIVA/EFM**		0.04 mg/kg	20 %	Standard method NS EN ISO 17294-2	ISO 17025, accredited
cobalt (Co)	7440-48-4	**NIVA/EFM**		0.005 mg/kg	20 %	Standard method NS EN ISO 17294-2	ISO 17025, accredited
tin (Sn)	7440-31-5	**NIVA/EFM**		0.1 mg/kg	20 %	Standard method NS EN ISO 17294-2	ISO 17025, accredited
Total-Hg	7439-9-76	**NIVA/EFM**		0.005 mg/kg	25 %	Standard method	ISO 17025, accredited
PCBs							
PCB-28	7012-37-5	**NIVA/EFM**		0.05 µg/kg low fat. 1 µg/kg high fat	40 %	Internal method	ISO 17025
PCB-52	35693-99-3	**NIVA/EFM**		0.05 µg/kg low fat. 1 µg/kg high fat	30 %	Internal method	ISO 17025
PCB-101	37680-73-2	**NIVA/EFM**		0.05 µg/kg low fat. 1 µg/kg high fat	40 %	Internal method	ISO 17025
PCB-118	31508-00-6	**NIVA/EFM**		0.05 µg/kg low fat. 1 µg/kg high fat	30 %	Internal method	ISO 17025
PCB-138	35065-28-2	**NIVA/EFM**		0.05 µg/kg low fat. 1 µg/kg high fat	30 %	Internal method	ISO 17025
PCB-153	35065-27-1	**NIVA/EFM**		0.05 µg/kg low fat. 1 µg/kg high fat	40 %	Internal method	ISO 17025
PCB-180	35065-29-3	**NIVA/EFM**		0.05 µg/kg low fat. 1 µg/kg high fat	40 %	Internal method	ISO 17025
p-p`-DDT	50-29-3	**NIVA/EFM**		0.2 µg/kg low fat. 4 µg/kg high fat	60 %	Internal method	ISO 17025
p-p`-DDE	82413-20-5	**NIVA/EFM**		0.05 µg/kg low fat. 1 µg/kg high fat	40 %	Internal method	ISO 17025
p-p`-DDD	72-54-8	**NIVA/EFM**		0.1 µg/kg low fat. 2 µg/kg high fat	50 %	Internal method	ISO 17025
PBDEs							
BDE47	5436-43-1	**NIVA/EFM**		0.005 µg/kg mussels. 0.1 µg/kg high fat	30 %	Internal method	ISO 17025
BDE99	60348-60-9	**NIVA/EFM**		0.01 µg/kg mussels. 0.1 µg/kg high fat	40 %	Internal method	ISO 17025
BDE100	189084-64- 8	**NIVA/EFM**		0.01 µg/kg mussels. 0.1 µg/kg high fat	40 %	Internal method	ISO 17025
BDE126*	366791-32-4	**NIVA/EFM**		0.01 µg/kg mussels	50 %	Internal method	ISO 17025
BDE153	68631-49-2	**NIVA/EFM**		0.02 µg/kg mussels. 0.1 µg/kg high fat	40 %	Internal method	ISO 17025
BDE154	207122-15-4	**NIVA/EFM**		0.02 µg/kg mussels. 0.1 µg/kg high fat	40 %	Internal method	ISO 17025
BDE183	207122-16-5	**NIVA/EFM**		0.03 µg/kg mussels. 0.3 µg/kg high fat	40 %	Internal method	ISO 17025
BDE196	32536-52-0	**NIVA/EFM**		0.05 µg/kg mussels. 0.3 µg/kg high fat	40 %	Internal method	ISO 17025
BDE209	1163-19-5	**NIVA/EFM**		0.5 µg/kg mussels. 0.5 µg/kg high fat	50 %	Internal method	ISO 17025
α, ß, γ-HBCD	134237-α (-50-6), ß (-51-7), γ (-52-8)	**EF-GFA**		0.006 ng/g	40 %	Internal method, validated	ISO 17025
Tetrabrombisphenol A (TBBPA)	79-94-7	**EF-GFA**		0.5 ng/g	40 %	Internal method, validated	ISO 17025
Bisphenol A (BPA)	80-05-7	**EF-GFA**		1-5 ng/g	40 %	Internal method, validated	ISO 17025
PFAS							
PFNA	375-95-1	**NIVA**	0.4 µg/kg		30 %	Internal method, validated	Not accredited but follows the routines and systems of ISO 17025
PFOA	335-67-1	**NIVA**	0.4 µg/kg		40 %	Internal method, validated	Not accredited but follows the routines and systems of ISO 17025

Name	[CAS-number]	Lab.	LOD	LOQ1	Est. uncertainty	Standard or internal method	Accreditation status
PFHpA	375-85-9	**NIVA**	0.4 µg/kg		30 %	Internal method, validated	Not accredited but follows the routines and systems of ISO 17025
PFHxA	307-24-4	**NIVA**	0.4 µg/kg		30 %	Internal method, validated	Not accredited but follows the routines and systems of ISO 17025
PFOS	1763-23-1	**NIVA**	0.1 µg/kg		25 %	Internal method, validated	Not accredited but follows the routines and systems of ISO 17025
PFBS	29420-49-3	**NIVA**	0.1 µg/kg		30 %	Internal method, validated	Not accredited but follows the routines and systems of ISO 17025
PFOSA	4151-50-2	**NIVA**	0.1 µg/kg		30 %	Internal method, validated	Not accredited but follows the routines and systems of ISO 17025
S/MCCP							
SCCP (C10-C-13)	85535-84-8	**EF-GFA**		0.6-3.5 ng/g	50 %	Internal method based on AIR OC 147, validated	ISO 17025
MCCP (C14-C17)	85535-85-9	**EF-GFA**		5-10 ng/g	50 %	Internal method based on AIR OC 147, validated	ISO 17025
Phenols							
Octylphenol	27193-28-8 (1806-26-4, 67632-66-0, 140-66-9,)	**EF-GFA**		10-50 ng/g	40 %	Internal method, validated	ISO 17025
4-nonylphenol	104-40-5 (25154-52-3, 84852-15-3)	**EF-GFA**		10-50 ng/g	40 %	Internal method, validated	ISO 17025
Tin compounds							
Monobutyltin (MBT)	2406-65-7 (78763-54-9)	**EF-GFA**		0.5 ng/g	40 %	Internal method, validated	ISO 17025
Dibutyltin (DBT)	1002-53-5	**EF-GFA**		0.5 ng/g	40 %	Internal method, validated	ISO 17025
Tributyltin (TBT)	688-73-3	**EF-GFA**		0.5 ng/g	30 %	Internal method, validated	ISO 17025
Trifenyltin (TPT)	668-34-8	**EF-GFA**		0.5 ng/g	40 %	Internal method, validated	ISO 17025
PFRs							
tri-iso-butylphosphate (TIBP)*	126-71-6	**EF-GFA**		20-200 ng/1 g fat	40 %	Internal method, under development	ISO 17025
tributylphosphate (TBP)	126-73-8	**EF-GFA**		20-200 ng/1 g fat	40 %	Internal method, under development	ISO 17025
tri(2-chlorethyl)phosphate (TCEP)	115-96-8	**EF-GFA**		20-200 ng/1 g fat	40 %	Internal method, under development	ISO 17025
tri(1-chlor-2-propyl) phosphate (TCPP)	13674-84-5	**EF-GFA**		20-200 ng/1 g fat	40 %	Internal method, under development	ISO 17025
tri(1,3-dichlor-2-propyl) phosphate (TDCP)	13674-87-8	**EF-GFA**		20-200 ng/1 g fat	40 %	Internal method, under development	ISO 17025
tri(2-butoxyethyl) phosphate (TBEP)	78-51-3	**EF-GFA**		20-200 ng/1 g fat	40 %	Internal method, under development	ISO 17025
triphenylphosphate (TPhP)	115-86-6	**EF-GFA**		20-200 ng/1 g fat	40 %	Internal method, under development	ISO 17025
2-ethylhexsyl-di-phenylphosphate (EHDPP)*	1241-94-7	**EF-GFA**		20-200 ng/1 g fat	40 %	Internal method, under development	ISO 17025
tetra is-(2-chloroethyl)dichlorisopentyldiphosph		**EF-GFA**		100-1000 ng/1 g fat	40 %	Internal method, under development	ISO 17025

Name	[CAS-number]	Lab.	LOD	LOQ1	Est. uncertainty	Standard or internal method	Accreditation status
ate (V6)							
dibutylfenylphosphate (DBPhP)**	2528-36-1	**EF-GFA**		100-1000 ng/1 g fat	40 %	Internal method, under development	ISO 17025
butyldifenylphosphate (BdPhP)**	2752-95-6	**EF-GFA**		100-1000 ng/1 g fat	40 %	Internal method, under development	ISO 17025
tris(2-etylheksyl)phosphate (TEHP)*	78-42-2	**EF-GFA**		20-200 ng/1 g fat	40 %	Internal method, under development	ISO 17025
tris-o-kresylphosphate (ToCrP)*	78-30-8	**EF-GFA**		20-200 ng/1 g fat	40 %	Internal method, under development	ISO 17025
trikresylphosphate (TCrP)	1330-78-5	**EF-GFA**		200-1000 ng/1 g fat	40 %	Internal method, under development	ISO 17025
Phthalates							
Dibutylphthalate (DBP)	84-74-2	**EF-Sofia**		500 µg/kg	40 %		Not accredited
Dibutyladipat (DBPA)		**EF-Sofia**		500 µg/kg	40 %		Not accredited
Diethylhexcyladipate (DEHA)		**EF-Sofia**		2000 µg/kg	40 %		Not accredited
Di(2-ethylhexyl)-phthalate (DEHP)	117-81-7	**EF-Sofia**		1000 µg/kg	40 %		Not accredited
Dietylphthalate (DEP)		**EF-Sofia**		500 µg/kg	40 %		Not accredited
Diethyladipat (DEPA)	85-68-7	**EF-Sofia**		500 µg/kg	40 %		Not accredited
Benzylbutylphthalate (BBP)		**EF-Sofia**		300 µg/kg	40 %		Not accredited
Diisobutylphthalate (DIBP)	84-69-5	**EF-Sofia**		500 µg/kg	40 %		Not accredited
Diisodectylyphthalate (DIDP)		**EF-Sofia**		5000 µg/kg	40 %		Not accredited
Diisoheptylphthalate (DIHP)		**EF-Sofia**		5000 µg/kg	40 %		Not accredited
1,2-Cyclohexane dicarboxylic acid diisononyl ester (DINCH)		**EF-Sofia**		500 µg/kg	40 %		Not accredited
Diisobutyl adipate (DIPA)		**EF-Sofia**		300 µg/kg	40 %		Not accredited
Dimethylphthalate (DMP)		**EF-Sofia**		500 µg/kg	40 %		Not accredited
Di-n-octylphthalte (DNOP)		**EF-Sofia**		500 µg/kg	40 %		Not accredited
Diphenylphthalate (DPF)		**EF-Sofia**		500 µg/kg	40 %		Not accredited
Dinonylphthalte+diisononylphthalate (SDD)		**EF-Sofia**		n.a.	40 %		Not accredited
Tributyl-o-acetylcitrate (TOA)		**EF-Sofia**		n.a.	40 %		Not accredited
Triclosan	9012-63-9	**ALS**		10 µg/kg	40 %	Internal method, under development?	Accredited
Diuron	330-54-1	**NIVA/EFM**		10 µg/kg	50 %	Internal method	Accredited
Irgarol	28159-98-0	**NIVA/EFM**		10 µg/kg	50 %	LE / cleanup / LC/MS/MS	Accredited
BEM							
VDSI		**NIVA**			10-20%	ICES 1999	Not accredited
EROD		**NIVA**			10-20%	ICES 1991	Not accredited
CYP1A		**NIVA**			10-20%	ICES 1998	Not accredited
ALA-D		**NIVA**			20 %	ICES 2004	Not accredited

2.2.2 Laboratories and brief method descriptions

The 2014 samples were largely analysed by Eurofins Moss (EFM) and by one of the Eurofins laboratories in Germany (GFA). NIVA was responsible for the PFAS analyses. A brief description of the analytical methods used follows (from Green *et al*. 2008a).

Metals were analysed at Eurofins Moss according to NS EN ISO 17294-2. Metals were extracted using nitric acid and quantified using Inductively Coupled Plasma Mass Spectrometry (ICP-MS), except for chromium, which was determined using GAAS or ICP-Atomic Emission Spectroscopy (ICP-AES). Mercury (total) has been analysed using Cold-Vapour AAS (CVAAS).

Polychlorinated biphenyls (PCBs) and other chlororganic hazardous substances were analysed at Eurofins-Moss using GC-MS. Fat content was extracted using a mixture of cyclohexane and acetone or iso-propanol on the target tissue. Among the individual PCBs quantified, seven (ΣPCB-7) are commonly used for interpretation of the results[1] (***Table 4***).

Table 4. *Suggested PCB-congeners (PCB-7), which are to be quantified in biota (ICES 1986).*

IUPAC/CB no.	Structure
28	2 4-4'
52	2 5-2'5'
101	2 4 5-2'5'
118	2 4 5-3'4'
138	2 3 4-2'4'5'
153	2 4 5-2'4'5'
180	2 3 4 5-2'4'5'

Polycyclic aromatic hydrocarbons (PAH) were analysed at Eurofins Moss using a gas chromatograph (GC) coupled to a mass-selective detector (MSD). The individual PAHs are distinguished by the retention time and/or significant ions. All seven potential carcinogenic PAHs (IARC 1987) are included in the list of single components determined to constitute the total concentration of PAH. For this report the total is the sum of tri- to hexacyclic PAH compounds named in EPA protocol 8310 minus naphthalene (dicyclic)-totalling 15 compounds, so that the classification system of the Norwegian Environment Agency can be applied (see Appendix B).

Organic tin compounds were analysed at Eurofins GFA in 2014/2014 using GC-MS detection.

Analyses of polybrominated diphenylether (PBDE) in cod liver were done at Eurofins Moss in 2014/2015. Results are given based on the total extractable fat content of the target tissue using a GC-Negative Chemical Ionization (NCI)-MS.

Analysis of perfluorinated alkylated substances (PFAS) in cod liver 2014 were done at NIVA. The general procedures include extractions with solvents using ultrasonic bath before intensive clean up and LC/MS/MS-analysis (ESI negative mode). From 2013 LC-qTOF has been used for detection and quantification. The limit of detection and quantification has improved for analyses of the 2014-samples primarily due to a slight modification in the method and better access to internal standards.

[1] Several marine conventions (e.g. OSPAR and HELCOM[1]) use ΣPCB-7 to provide a common basis for PCB assessment.

Previously most of the analyses were performed at NIVA, using different procedures and instrumentation. In order to minimize methodical disturbance in time series, the transfer of analyses from NIVA to Eurofins Moss has also included several intercalibrations between the two labs.

The new analyses introduced in 2012/2013 were done by Eurofins. Chlorinated paraffins (SCCP (C10-C13), MCCP (C14-C17)), phosphorus flame retardants (PFRs) and nonyl- and octylphenols were determined by GC-MS at Eurofins GFA. Determination of bisphenol A (BPA) and tetrabromobisphenol A (TBBPA) were done at Eurofins GFA by GC-MS while hexabromocyclododecane (α, β, γ-HBCD) were determined by LC-MS-MS also by Eurofins GFA.

For fish, the target tissues for quantification of hazardous substances were; liver and fillet (***Table 2***), whereas for the biological effects methods (BEM) liver; blood and bile were used (cf. *Table 5*). In addition, the age, sex, and visual pathological state for each individual were determined. Other measurements include: fish weight and length, weight of liver, liver dry weight and fat content (% total extractable fat), the fillet dry weight and its % fat content. These measurements are stored in the database and published periodically (e.g. Shi *et al*. 2008).

The mussels are analysed for all contaminants including organotin. The shell length of each mussel is measured. On a bulk basis the total shell weight, total soft tissue weight, dry weight and % fat content is measured. These measurements are stored in the database and published periodically.

The dog whelk are analysed for organotin compounds and biological effects (imposex[1], see ***Table 3***).

2.3 Biological effects analysis

Five biological effects methods (BEM), including the measurement of OH-pyrene have been applied on an annual basis for this investigation. Each method is in theory generally indicative of one or a group of contaminants. For EROD and CYP1A however, some interaction effects are known. Analysis of OH-pyrene in bile is not a measurement of biological effects, per se. It is included here, however, since it is a result of biological transformation (biotransformation) of PAHs, and is thus a marker of PAH exposure. An overview of the methods, tissues sampled and contaminant specificity is shown in ***Table 5***. One of the major benefits of BEM used at the individual level (biomarkers) is the feasibility of integrating biological and chemical methods, as both analyses are done on the same individual.

Table 5*. The relevant contaminant-specific biological effects methods applied on an annual basis.*

Code	Name	Tissue sampled	Specificity
OH-pyrene	Pyrene metabolite	fish bile	PAH
ALA-D	δ-aminolevulinic acid dehydrase inhibition	fish red blood cells	Pb
EROD-activity	Cytochrome P4501A-activity (CYP1A/P4501A1, EROD)	fish liver	planar PCB/PCNs, PAHs, dioxins
CYP1A	Relative amount of cytochrome P450 1A-protein	fish liver	Supporting parameter for EROD-activity
TBT	Imposex	snail soft tissue	organotin

[1] Vas Deferens Sequence Index

BEM-sampling requires that the target fish is kept alive until just prior to sampling. Sampling for BEM-analyses is performed by trained personnel, most often under field conditions. Immediately after the fish are inactivated by a blow to the head. Samples are then collected and stored in liquid nitrogen. Analyses of a metabolite of pyrene (OH-pyrene) were done on bile samples stored at -20°C.

2.3.1 Rationale and overview

A thorough analysis and review of BEM-results has been performed twice since their inclusion in 1997 (Ruus *et al*. 2003; Hylland *et al*. 2009). Clear relationships were shown between tissue contaminants, physiological status, and responses in BEM parameters in cod (Hylland *et al*. 2009). Although metals contributed substantially to the models for ALA-D (and also for metallothionein - MT included in the programme 1997-2001) and organochlorines in the model for CYP1A activity, other factors were also shown to be important. Liver lipid and liver somatic index (LSI) contributed for all three BEM-parameters, presumably reflecting the general health of the fish. Size or age of the fish also exerted significant contributions to the regression models. It was concluded that the biological effect methods clearly reflected relevant processes in the fish even if they may not be used alone to indicate pollution status for specific locations at given times. Furthermore, the study showed that it is important to integrate a range of biological and chemical methods in any assessment of contaminant impacts. Through continuous monitoring within CEMP, a unique BEM time series/dataset are generated, that will also be of high value as a basis of comparison for future environmental surveys.

Biological effect methods were first included in the programme in 1997. There have been some modifications since then in accordance to the ICES guidelines (cf. ***Table 3***). In 2002, reductions were made in parameters and species analysed. There have also been improvements in the methods, such as discontinuation of single wavelength fluorescence and use of HPLC in the analysis of bile metabolites since 2000.

The MILKYS programme for 2014 included five biological effects methods (BEM) (cf. ***Table 5***). Measures of OH-pyrene, EROD-activity and CYP1A increase with increased exposure to their respective inducing contaminants. The activity of ALA-D on the other hand is inhibited by contamination (i.e., lead), thus lower activity means a response to higher exposure.

2.4 Passive sampling with silicone rubber passive samplers

2.4.1 Principle of passive sampling for hydrophobic contaminants

Passive sampling is based on the diffusive movement of substances from the environmental matrix being sampled into a polymeric device (initially free of the compounds of interest) in which contaminants absorb. For the passive sampling of hydrophobic compounds the best known sampler is the SemiPermeable Membrane Device (SPMD) comprising a low density polyethylene membrane containing a triolein lipid phase (Huckins *et al*. 2006). Currently, single phase polymeric samplers constructed from material such as low density polyethylene or silicone rubber are used as a result of their robustness (Allan *et al*. 2009a, b, Allan *et al*. 2010, Allan *et al*. 2011a, b). At equilibrium, the mass of a chemical absorbed in the sampling device can be used to calculate the freely dissolved contaminant concentration in the water that the device was exposed to through K_{sw}, the sampler-water partition coefficient. Passive sampling techniques that allow to derive freely dissolved contaminant concentrations have been the subject of much development over the last two decades (Vrana *et al*. 2005). For hydrophobic contaminants with $\log K_{ow} > 5$-6, polymeric

samplers have a large capacity. For typical deployment periods of a few weeks, equilibrium between the sampler and water will not be attained for these chemicals. Uptake in the linear mode (i.e. far from equilibrium) is therefore time-integrative for the deployment period in water. The resulting time-integrated freely dissolved concentration can be estimated if *in situ* sampling rates, R_s, equivalent amount of water sampled per unit of time (L d^{-1}) are known. Sampling rates can be estimated from the dissipation of performance reference compounds (PRC), analogues of compounds of interest (but not present in the environment) spiked into the samplers prior to exposure (Booij *et al.* 1998, Huckins *et al.* 2002).

Passive sampling based on silicone rubber is increasingly being used for routine monitoring of water and sediment. These have been used to monitor a range of contaminants at Andøya, Bjørnøya and Jan Mayen (*Tilførselprogrammet* 2009-2013). Deployments were in most cases at least 200 days. For the riverine input and discharge programme (RID, 2013-), silicone rubber passive samplers have also been chosen. The reason for this choice is that we have recently shown that there is a likely restriction of the sampling of voluminous molecules such as polybrominated diphenyl ethers when using polyethylene (Allan *et al.* 2013). This can affect the accurate estimation of sampling rates for these compounds from standard PRCs.

Passive samplers were deployed at three sites, Hvaler, Oslofjord and Ålesund for periods of just under one year and analysed for performance reference compounds (to estimate sampling rates), alkylphenols (octyl and nonylphenols), hexabromocyclododecane (HBCD) and polybrominated diphenyl ethers (PBDEs).

2.4.2 Methodology (field and lab)

Samplers used for this project include silicone rubber passive samplers (for analysis and for specimen banking), low density polyethylene (for specimen banking), and Polar Chemical Integrative Samplers (for specimen banking).

Samplers made of AlteSil silicone rubber (nominal size of 1000 cm^2 and 30 g, strips 100 cm long and 2.5 cm wide) were prepared in the NIVA laboratory following standard procedures. In short, the silicone rubber samplers were placed in a Soxhlet extractor for 24 hour cleaning using ethyl acetate. This step removes a significant amount of non-polymerized oligomers. Samplers were then left to dry before further cleaning with methanol. PRCs (deuterated PAHs and fluoroPCBs) were spiked into the samplers using a methanol-water solution (Booij *et al.* 2002). Onced spiked with PRCs, samplers were kept in the freezer at -20 °C until deployment. POCIS devices were purchased from Exposmeter AB (Sweden).

Two sets of replicate silicone samplers were deployed at each of the three sites (Oslofjord, Ålesund havn and Hvaler) using SPMD canisters and samplers mounted on spider holders. Two control samplers were used to assess potential contamination of the samplers during preparation and deployment procedures and to assess initial PRC concentrations. Triplicate POCIS devices were exposed at each of the three stations (one control sample per site was used). The sampling stations and deployment duration are shown in **Table 6**. Samplers were deployed for over 300 days at all three stations.

***Table 6**. Sampling stations, deployment and retrieval dates, and exposure times for samplers deployed at the three stations.*

Sampling station	Coordinates	Deployment date	Retrieval date	Exposure time (d)
Oslofjord (*304PP*)	N59.85527 E10.59527	22.07.2014	09.06.2015	322
Hvaler (*HPP*)	N59.09655 E11.05073	25.07.2014	25.06.2015	335
Ålesund harbour (*APP*)	N62.46322 E06.22077	05.08.2014	01.07.2015	330

Once back in the laboratory, all samplers were kept in the freezer at -20 °C until extraction and analysis.

Replicate samplers (~30 g each) and a control from each station were extracted. Additional preparation control samplers and QA spiked samplers were analysed together with exposed samplers. The initial step consisted in cleaning the surface of the samplers with milliQ water and drying before extraction. Samplers were placed in clean glass jars with recovery standards of substances of interest before extraction with pentane (200 mL) overnight. This extraction was repeated with fresh pentane and pentane extracts were combined. Extracts were reduced and split for the different analyses.

For PRCs and alkylphenols, the extract was cleaned up by gel permeation chromatography (GPC). One fraction of the extract was then analyzed by GC-MS to determine PRC concentrations. The other fraction of the extract was derivatised (with a solution of N,O-bis(trimethylsilyl) trifluoroacetamide and trimethylchlorosilane) before determination of alkyl phenolic substances by GC-MS.

For PBDEs and HBCD, the extract was cleaned up with concentrated sulphuric acid. The extract was then split into two. One fraction of the extract was cleaned up by acetonitrile partitioning before PBDEs determination by GC-MS. The solvent of the second fraction was changed to methanol before determination of HBCD isomers by LC-MS-MS.

2.4.3 Quality assurance: Spiked samplers

A set of silicone rubber passive sampling devices was prepared for QA purposes following a similar procedure to that used for standard samplers. Instead of spiking PRCs, target substances in known amounts were added to the samplers using the methanol-water solution (Booij *et al*. 2002). Substances added included alkylphenolic substances, polybrominated diphenyl ethers and hexabromocyclododecane isomers.

Once the batch was ready, six QA spiked samplers were randomly selected for extraction and analysis to determine the mean concentration and the reproducibility of the spiking of different samplers. The remaining QA spiked samplers were put into tins and stored in the freezer at -20 °C until use. The table in (Appendix G) shows mean concentrations (n = 6) obtained in QA spiked samplers for alkylphenolic substances, HBCD isomers and PBDE congeners. Mean concentrations measured are within 89-120 % of the nominal concentrations across the range of substances spiked into the samplers. Relative standard deviations of amounts spiked into the samplers vary from 4 to 19 % across the range of compounds (Appendix G).

2.4.4 Passive sampling data processing

Freely dissolved concentrations were calculated using the boundary-layer controlled uptake model given in Rusina *et al*. (2010) and using the non-linear least square method to estimate sampling rates as a function of $logK_{sw}$/MW (Booij & Smedes, 2010) from the performance reference compound data. Polymer-water partition coefficients for PRCs and for alkylphenols were not corrected for temperature or salt content of the water (but can be at a later stage if needs be). For PRCs (deuterated PAHs), K_{sw} values were from Smedes *et al*. (2009). For para-n-octylphenol and para-n-nonylphenol, $logK_{sw}$ values were 4.43 and 5.08, respectively (unpublished). Correlation of $logK_{sw}$ values with hexadecane-water partition coefficients (from Cosmotherm software), $logK_{hdw}$ were used to estimate $logK_{sw}$ for para-t-octylphenol and para-t-nonylphenol. Ultimately a measured value of K_{sw} for these compounds will be preferable. For PBDEs and HBCD, K_{sw} (not available for these substances) were estimated using the regression of $logK_{sw}$ with $logK_{ow}$ for PCBs for AlteSil silicone rubber.

2.5 Information on quality assurance

2.5.1 International intercalibrations

The laboratories have participated in the Quality Assurance of Information for Marine Environmental Monitoring in Europe (QUASIMEME) international intercalibration exercises and other proficiency testing relevant to chemical and imposex analyses. For chemical analyses, round 2014-2, FAPAS 202014 05100F and FAPAS 1264 apply to the 2014-samples. The results are acceptable. These QUASIMEME exercises included nearly all the contaminants as well as imposex analysed in this programme. The quality assurance programme is corresponding to the analyses of the 2013 samples (cf. Green *et al*. 2014).

NIVA participated in the last round of QUASIMEME Laboratory Performance Studies “imposex and intersex in Marine Snails BE1” in June-August 2012. Shell height, penis-length-male, penis-length-female, average-shell-height and female-male-ratio were measured. NIVA got the score satisfactory for all parameters except number of females for one sample, which got the score questionable. The score for VDSI was satisfactory for both samples tested.

2.5.2 Analyses of certified reference materials

In addition to the QUASIMEME exercises, certified reference materials (CRM) and in-house reference materials are analysed routinely with the MILKYS samples. It should be noted that for biota, the type of tissue used in the CRMs does not always match the target tissue for analysis. Uncertain values identified by the analytical laboratory or the reporting institute are flagged in the database. The results are also “screened” during the import to the database at NIVA and ICES.

The laboratories used for the chemical testing are accredited according to ISO 17025:2005, except for the PFCs.

2.6 Classification of environmental quality

There are several systems that can be used to classify the concentrations of contaminants observed. No system is complete in that it covers all the contaminants and target species-tissues investigated in this programme. The national classification system prepared by the Norwegian Environment Agency (*Miljødirektoratet*) has been the most used and in investigations similar to this programme and it is applied here. It is the most complete system and provides assessment criteria for five classes of contamination, where Class I is the best class (lowest concentration). This system is built on presumed background concentrations and the degree above this level. It is

currently under revision to accommodate the concern that elevated concentrations of contaminants can be harmful for the environment.

This risk-based approach is the basis for EU directives which have defined Environmental Quality Standards (EQS). Exceedances of EQS are interpreted as potentially harmful to the environment and remedial action should be implemented. Two main challenges with the EQS that prevent them from being easily applied are that they are generally not species or tissue specific and they can be in conflict with the national limits. The EQS apply to the whole organism whereas in fish monitoring analysis is generally done on a specific tissue[1]. The EQS can be considerably higher or lower than the national Class II (moderately polluted). For example for hexachlorobenzen (HCB) the EQS is 10 µg/kg w.w., whereas Class I and II are 0.1 and 0.3 µg/kg w.w. for blue mussel, respectively, and 0.2 and 0.5 µg/kg w.w. in cod fillet, respectively; or for mercury the EQS is 20 µg/kg w.w. whereas Class I and II are 40 and 100 µg/kg w.w. for blue mussel, respectively, and 100 and 300 µg/kg w.w. in cod fillet, respectively (cf. ***Table 7*** and Appendix C). These anomalies warrant the need to have clear guidance as to how the EQS should be applied and how to explain the difference in the two systems. Even so, the EQS have been discussed where possible when assessing the results from this programme.

Assessing the risk to human consumption that elevated concentrations of contaminants in seafood has not been the task of this programme and hence, the EU foodstuff limits have not been applied.

Focus for the 2014-investigation is on the principle cases where median concentrations exceeded the upper limit to Class I in the environmental quality classification system of the Norwegian Environment Agency (cf. Molvær *et al.* 1997)[2]. In addition to this, the EU directive 2013/39/EU where Environmental Quality Standards (EQS) for biota are defined are considered (***Table 7***, ***Table 10***). The Norwegian Environment Agency defines most classes on a wet weight basis, the exception being for metals in blue mussel which are on a dry weight basis. The EQS and OSPAR time trend methods of analyses are based on wet weight concentrations. To harmonize the presentation classification and trend analyses for these results the class limits for metals in blue mussel were unofficially converted to a wet weight basis where needed. The relevant part of the Norwegian Environment Agency system is shown in Appendix C.

The choice of base by OSPAR is aimed at meeting several considerations: scientific validity, uniformity for groups of contaminants for particular tissues and a minimum loss of data. As to the latter, the choice of base will affect the number of data that can be included in the assessment, depending on available information on dry weights, wet weights and lipid weights.

[1] The concentration of a contaminant can vary considerably from tissue to tissue. Hence, monitoring is usually based on tissues with high concentrations and that are of sufficient size to meet the constraints of the analyses. In this regard fish liver and fish fillet are the most commonly used tissues in monitoring.

[2] The Norwegian Environment Agency report M-241 (Arp *et al.* 2014) with updated standards was not finalized in time to be considered for this report. However, it can be noted these updated standards like EU directive 2013/39/EU concern biota in general, and, unlike Molvær *et al.* (1997), are not species or tissue specific.

***Table 7**. The Water Framework Directive (WFD) Environmental Quality Standards for "biota"* [1] *(cf. Environmental Quality Standard Directive-2013/39/EU) and the Class I and V (upper limit to insignificant and extreme degree of pollution, respectively) in the environmental classification system of the Norwegian Environment Agency (NEA) (Molvær et al. 1997). Concentrations are given in µg/kg wet weight. Note: EQS used for assessing water with passive sampling are treated separately (see Appendix G, Table 38).*

Hazardous substance	EQS biota [1]	NEA - blue mussel Class I - V	NEA - cod-liver Class I - V	NEA - cod-fillet Class I - V
Brominated diphenylether [2]	0.0085			
Fluoranthene	30 [3]			
Benzo(a)pyrene	5 [3]	1 - 30		
Benzo(b)fluoranthene	[3]			
Benzo(k)fluoranthene	[3]			
Benzo(g,h,i)perylene	[3]			
Indeno(1,2,3-cd)-pyrene	[3]			
Polyaromatic hydrocarbons (PAH) [4]		50 – 5000		
Hexachlorobenzene (HCB)	10	0.1 - 5	20 - 40	0.2 - 5
Hexachlorobutadiene (HCBD)	55			
Mercury and its compounds	20	40 – 800 [5]		100 – 1000
Dicofol	33			
Perfluorooctane sulfonic acid and its derivatives (PFOS)	9.1			
Dioxins and dioxin-like compounds	0.0065 [6]			
Hexabromocyclododecane (HBCD)	167			
Heptachlor and heptachlorexpoxide	0.0067			

1) Fish unless otherwise stated. An alternative biota taxon or another matrix may be monitored instead as long as the EQS applied provides an equivalent level of protection.
2) Sum of BDE congener numbers 28 (tri), 47 (tetra), 99 (penta), 100 (penta), 153 (hexa) and 154 (hexa)
3) Crustaceans and molluscs. (Monitoring of these PAHs not appropriate for fish)
4) The sum of tri- to hexacyclic PAH compounds named in EPA protocol 8310 minus naphthalene (dicyclic)-totalling 15 compounds, so that the classification system of the Norwegian Environment Agency can be applied.
5) Conversion assuming 20% dry weight.
6) Sum of PCDD+PCSF+PCB-DL TEQ

The system has five classes from Class I, insignificantly polluted, to Class V, extremely polluted. However, the system does not cover all the contaminants for the species and tissues used in MILKYS. To assess concentrations not included in the system provisional presumed high background values were used (cf. Appendix C). The factor by which this limit or the Class I limit is exceeded is calculated (cf. Appendix F). High background concentration corresponds to the upper limit to Class I; insignificantly polluted, which in this context has no statistical implications.

The median concentrations are assessed according to the system of the Norwegian Environment Agency, but where this is not possible, presumed high background levels are used. It should be noted that there is in general a need for periodic review and supplement of the list of limits used in the classification system in the light of results from reference localities and introduction of new analytical methods, and/or units. Because of changes in the limits, assessments of presumed high background levels over the years may not correspond.

Recommendations for changes to Class I (cf. Knutzen & Green 2001, Green & Knutzen 2003) have been taken into account in this report. Revisions to corresponding Classes II-V have not been done, but the Norwegian Environment Agency is currently reviewing their classification system.

The results can also be useful as part of the implementation of The Water Framework Directive (WFD) (2000/60/EC) ratified by Norway in 2009, and the Marine Strategy Directive (MSFD) (2008/56/EC), which by late 2015 has not yet been ratified by Norway. These two directives together concern all waters out to territorial borders. They are the main policies at the EU level designed to achieve good "ecological" (WFD) or "environmental and chemical" (MSFD) status, herein termed GES, in the European marine environment, by the year 2015 (2021 for Norway) and 2020 at the latest, respectively. The directives also set out to ensure the continued protection and preservation of the environment and the prevention of deterioration. The Norwegian framework regulation on water management (the Water Regulation) was adopted on December 15th 2006, and incorporates the WFD into Norwegian law. The Environmental Quality Standards (EQS) for 45 priority substances or groups of substances have been outlined in the EQS Directive (EQSD) (2013/39/EU replacing directive 2008/105/EC). Several of these substances are monitored by MILKYS. The EQS apply to concentrations in water, and for fifteen substances biota (***Table 7***, ***Table 10***). There is also a provision which allows a country to use other EQS in sediment and biota provided these offer the same level of protection as the EQS set for water. It should be noted that application of the EQS set may be in conflict with the best class by the Norwegian Environment Agency system for classification of environmental quality; e.g. lower than the Class I for mercury and higher for Class V for HCB in blue mussel. This has not been resolved and for this report, the system of the Norwegian Environment Agency provides the primary assessment criteria.

Proposed background assessment criteria (BAC) for EROD and OH-pyrene and VDSI (OSPAR 2013) were used to assess the results (***Table 8***).

Table 8. *Assessment criteria for biological effects measurements using background assessment concentration (BAC) and Environmental assessment criteria (EAC) (OSPAR 2013). Note that Assessment criteria have specifically been compiled for the assessment of CEMP monitoring data on hazardous substances. They do not represent target values or legal standards (OSPAR 2009).*

Biological effect	Applicable to:	BAC	EAC	Units, method
EROD	cod liver	145	-	pmol/min/ mg microsomal protein
OH-pyrene	cod liver	21*	-	ng/ml; HPLC-F
VDSI	dog whelk	0.3	2	

*) Values in this report are normalized and the unit of the assessment criterion is ng/ml, without normalization to absorbance at 380nm. Normalization in this investigation reduced the values by a factor of about 30.

2.7 Statistical time trends analysis

2.7.1 Treatment of values below the detection limit

Values below the limit of detection are set to half of the value of this limit for calculation for use in time trends or set to zero when included in a sum (e.g. PCB-7). This is in accordance to EU directive (2009/90/EC). The annual median is classified as less-than if over half of the values are below the limit of detection and is assigned the median value prefixed with a "<" sign in Appendix F, however when presented in tables of the main text on half of this value is shown. It should be noted that the detection limit can vary within and among sets of samples and comparisons of detection limits should be made with caution.

In calculating trends, a time series must have at most only one "less-than median" provided it is not the first in the series. The effect that a less-than value has on the trend analysis has not been

quantified; however, the results should be treated with caution because the dominance of values below the limit of quantification could invalidate the statistical assumption behind the analysis (Rob Fryer, pers. comm.).

2.7.2 The model approach

A simple model approach has been developed to study time trends for contaminants in biota based on median concentration (ASMO 1994). The method has been applied to Norwegian data and results are shown in Appendix E. The results can be presented as shown in ***Figure 5***. It should be noted that this robust method has been developed so that it could provide a rough guide to possible trends in the OSPAR region. Further investigation is necessary to better understand the factors affecting a particular trend. This may lead to different conclusions. As an exercise in this respect the times series for mercury in cod filet from the Inner Oslofjord was examined more closely (see section 3.2.1).

The model approach uses a Loess smoother based on a running six-year interval where a non-parametric curve is fitted to median log-concentration (Nicholson *et al*. 1991, 1994 and 1997 with revisions noted by Fryer & Nicholson 1999). The concentrations are on the preferred basis of wet weight as mentioned above. Supplementary analyses were performed on a dry weight basis for blue mussel data and lipid weight basis for chlororganic contaminants in blue mussel and fish liver (see Appendix F). For statistical tests based on the fitted smoother to be valid the contaminants indices should be independent to a constant level of variance and the residuals for the fitted model should be log-normally distributed (cf. Nicholson *et al*. 1998). A constant of +1 was added to VDSI data prior to log transformation to enable analysis of observations that were equal to zero.

An estimate was made of the power of the temporal trend series expressed as the percent change that the test is able to detect. The power is based on the percentage relative standard deviation (RLSD) estimated using the robust method described by ASMO (1994) and Nicholson *et al*. (1998). The estimate was made for series with at least five years of data.

The assessment method used up to and including the 2011 investigation have differed slightly from the method now employed by OSPAR in that a linear trend for the whole time series period was tested whereas OSPAR currently tests the difference in the smoothed annual concentration at the beginning of the time series compared the smoothed annual concentration at the end of the time series. This report presents an assessment in line with the current OSPAR approach.

The term “significant” refers to the results of a statistical analysis at 0.05 significance level used for detecting differences between the beginning and the end of the time series and can be found in the tables in Appendix F. In this appendix the statistical significance (p) is given as well as the annual detectable change (%) that can be detected with statistical probability of 90 % (Power) in two-sided testing with a 10 % significance level (alpha).

No attempt has been made to compensate for differences in size groups or number of individuals of blue mussel or fish in this study. However, investigations prior to 2007 showed significant differences between “small” and “large” fish. With respect to blue mussel, there is some evidence that concentrations do not vary significantly among the three size groups employed for this study (i.e. 2-3, 3-4 and 4-5 cm) (WGSAEM 1993).

The statistical analysis of time trends was carried out on all the results, including those for biological effects parameters.

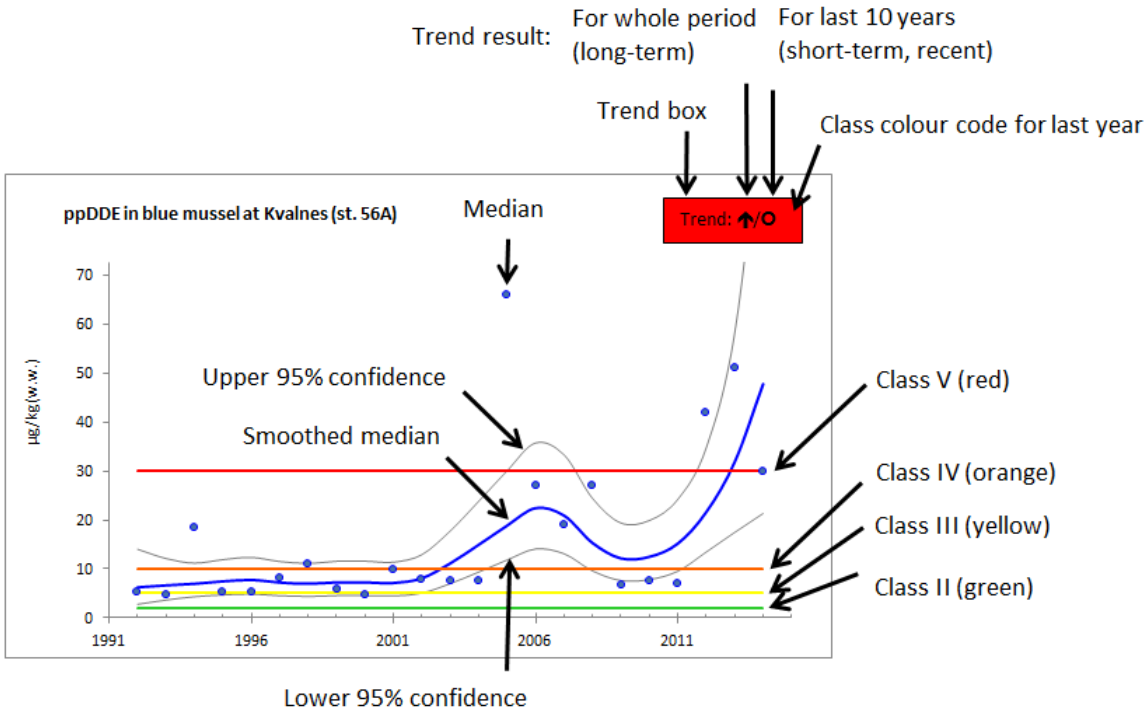

***Figure 5**. Example of time series that show the median concentration (blue dots), running mean of median values (Loess smoother - blue line) and 95 % confidence intervals (grey lines). The horizontal lines indicate the lower boundaries to the classes of pollution in the system of the Norwegian Environment Agency : Class II (green line, moderate=upper boundary to Class I (insignificantly polluted, also herein termed as "acceptable")), III (yellow line, marked), IV (orange line, severe) and V (red line, extreme) (cf. **Table 34**), or alternatively the Class II boundary is replaced by the upper boundary to provisional "high background level" as in which case no class-boundaries are shown. Further, if there are no classes the background concentration is indicated by a light grey line (see text and refer to Appendix C). For biota, trend analyses (shown in the trend box) were done on time series with five or more years and the results, before the slash "/" (i.e. long-term trend which means the entire time series), are indicated by an upward (↑) or downward (↓) arrow where significant trends were found, or a zero (O) if no trend was detected. Where there was sufficient data a time series analysis was performed for the last ten-year for the period 2005-2014 (short-term or recent trend) and the result is shown after the slash. A small filled square (▪) indicates that chemical analysis has been performed, but data either were insufficient to do a trend analysis or was not presented. The trend box is also coloured with respect to the Norwegian Environment Agency classification system as it applies to the final year: blue (Class I), green (Class II), yellow (Class III), orange (Class IV) or red (Class V). In addition, the box may be coloured dark grey or light grey. Dark grey indicates concentrations higher than estimated high background levels. Light grey indicates concentrations lower than background levels. Note that scales for the x axis and y axis can vary from figure to figure.*

3. Results and discussion

3.1 General information on measurements

A summary of the levels and trends in contaminants or their effects in Atlantic cod, blue mussel, dog whelk and periwinkle along the coast of Norway in 2014 is shown in ***Table 10***. More details on trend analyses for the entire monitored period that include results from either 2013 or 2014 are shown in Appendix F. The results from 2014 and some supplementary analyses of 2013-samples present data for a total of 2103 data sets (contaminant[1]-station-species) on over 136 different contaminants. Unless otherwise stated assessment of trends in the text below refer to long-term trends, i.e. for the whole sampling period, whereas a short-term trend refers to the analysis on data for the last 10 years, i.e. 2005-2014 and can also be referred to as recent trend.

Time trend analyses were performed on a selection of 30 representative contaminants or their effect (VDSI), and included data for 2014 and totalled 759 data series (***Table 9***). Of the 759 cases 53.1% could be classified and there were 37 cases where median concentrations were in Class II or higher in the Norwegian Environment Agency classification system (Molvær *et al*. 1997) or above what is expected in only diffusely contaminated areas (collectively termed: "over presumed high background concentrations"). Of the 759 data series recent and significant trends were registered in 104 cases: 86 (11.9 %) were downwards trends and 18 (2.4 %) were upwards (***Figure 6A***). Of the 403 cases that could be classified by the system of the Norwegian Environment Agency, 374 (92.8 %) were classified as insignificantly polluted (Class I), 26 (6.5 %) as moderately polluted (Class II), 2 (0.5 %) as markedly polluted (Class III), 0 as severely polluted (Class IV) and 1 (0.2 %) as extremely polluted (Class V, ***Figure 6B***). The downward trends were primarily associated with metals (55.6 %), tributyltin (TBT, 7.8 %) and Vas Deferens Sequence Index (VDSI) (the effect of TBT) (8.9 %) (***Figure 7A***). The upward trends were also mainly associated with metals (94.4 %), primarily Hg (27.8 %). There were 3 cases classified higher than Class II: PCBs and a PAH compound (Class III) and a DDT metabolite (Class V) (***Figure 7B***). The results are discussed in more detail below.

Primary focus were on those cases where median concentrations in 2014 were over presumed high backgrounds level (>Class I, insignificantly polluted, acceptable levels) and where significant upward trends were found, and to a lesser degree where no significant trends or significant downward trends were found. The evaluation focused secondarily on cases where median concentrations in 2014 were below presumed high background level (<Class I, insignificantly polluted) in combination with significant upward trends. An overview of trends, classifications and median concentrations is presented in Appendix F. The results are presented by classes and with results for observed trend analyses.

[1] In this regard «contaminants» include *inter alia* results from biological effects methods, stable isotopes and some biological co-variables.

***Table 9**. Selection of representative contaminants and number of time series assessed for each target species-tissue. Counts include supplementary investigations funded by the Ministry of Climate and Environment and are marked with an asterisk " * " [1].The specific results are shown in **Table 10**.*

Contaminant /BEM	Description	Blue mussel	Dog whelk, periwinkle	Cod, liver	Cod fillet	TOTAL
Ag	silver	32*		13		**45**
As	arsenic	32*		13		**45**
Cd	cadmium	32*		13		**45**
Co	cobalt	32*		13		**45**
Cr	chromium	32*		13		**45**
Cu	copper	32*		13		**45**
Hg	mercury	32*			14	**46**
Ni	nickel	32*		13		**45**
Pb	lead	32*		13		**45**
Zn	zinc	32*		13		**45**
PCB-7 (CB_S7)	sum of PCB congeners 28+52+101+118+138+153+180	29*		13		**42**
ppDDE (DDEpp)	p,p'-DDE (a DDT metabolite)	19*		7*		**26**
HBCDa	α—hexabromocyclododecane	10		10		**20**
SCCP	short chain chlorinated paraffin (C10-C13)	10		10		**20**
MCCP	medium chain chlorinated paraffin (C14-C17)	10		10		**20**
BDE47	tetrabromdiphenylether	10		8		**18**
BDE100	pentabromdiphenylether	10		8		**18**
BDE209	decabromdiphenylether	10		8		**18**
PAHs (P_S)	sum nondicyclic PAHs	11				**11**
KPAHs (PK_S)	sum carcinogen PAHs	11				**11**
BKF	benzo[k]fluoranthene	11				**11**
B[ghi]P	benzo[ghi]perylene	11				**11**
ICDP	Indeno[1,2,3-cd]pyrene	11				**11**
B[a]P	benzo[a]pyrene	11				**11**
FLU	Fluoranthene	11				**11**
PFOS	perfluorooctanoic sulfonate			8		**8**
PFOSA	perfluorooctylsulfonate acid amide			8		**8**
PFBS	Potassium perfluorobutanesulfonat			8		**8**
TBT	tributyltin (formulation basis)	8*	9			**17**
VDSI	Vas Deferens Sequence Index		8			**8**
TOTAL		**513**	**17**	**215**	**14**	**759**

1) Supplementary investigations funded by the Ministry of Climate and Environment involved additional analyses on samples from blue mussel stations 30A, I301, I304, 31A, 36A1, 71A, I712, 51A, 56A, 65A, 22A, 10A2 and 11X; cod stations 30B, 36B, 15B, 53B, 23B, 98B1 and 10B; as well as all analyses for blue mussel stations: 52A, 57A, 63A, 69A, I133, I306, I307.

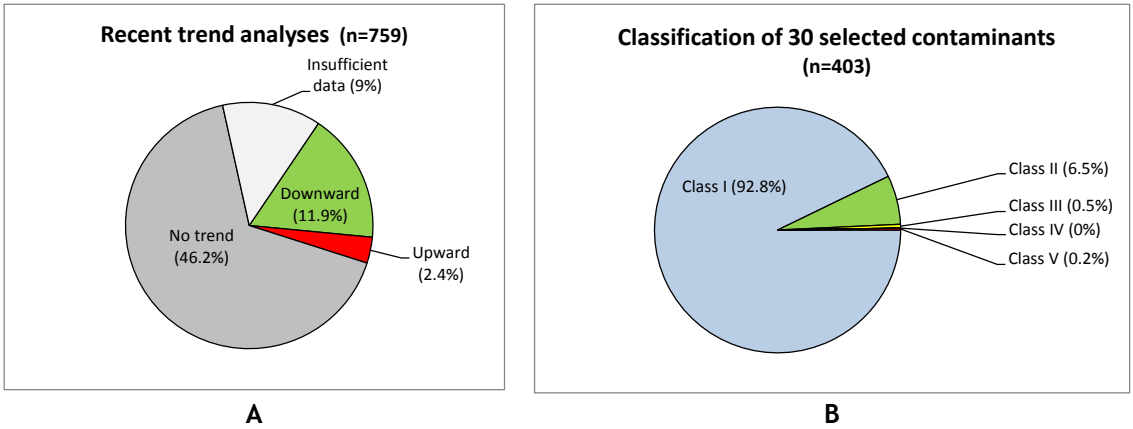

Figure 6. *Summary of the results from short-term trend analyses (**A**) and classification in Norwegian Environment Agency system (B) for 30 selected contaminants (cf. **Table 9**). Colour coding in Figure B refers to classification colours (cf. **Table 34**).*

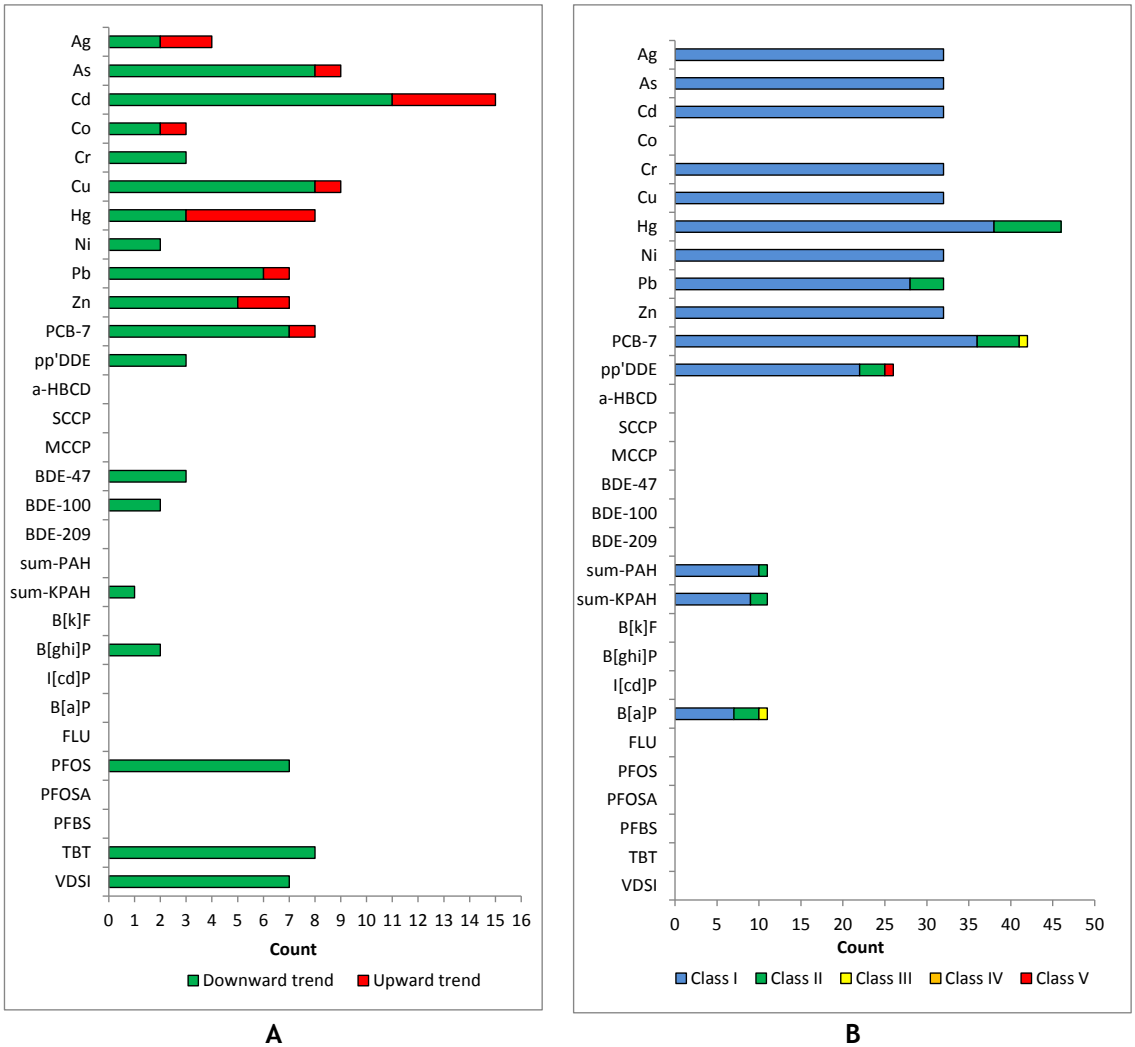

Figure 7. *Summary of short-term trends (**A**) and classification in Norwegian Environment Agency system (B) for each of the 30 selected contaminants (cf. **Table 9**, (see Appendix B for description of chemical codes). Colour coding in Figure B refers to classification colours (cf. **Table 34**).*

***Table 10**. Overview of samples collected in 2014 with indication of levels and trends in concentrations of contaminants monitored. Classification is based on observed concentrations in cod, blue mussel, dog whelk and periwinkle. Tissues: soft body (SB), muscle (MU, fillet), liver (LI) and whole organism (WO). The classification system of the Norwegian Environment Agency is used for biota (Molvær et al. 1997: Classes: I (blue), II (green), III (yellow), IV (orange) and V (red) (see Appendix C). For biota, trend analyses were done on time series with five or more years. An upward (↑) or downward (↓) arrow indicates statistically significant trends, whereas a zero (O) indicates no trend. A small filled square (▪) indicates that chemical analysis was performed, but either the results were insufficient to do a trend analysis. Results marked with a star (★) indicate that there is insufficient data above the detection limit to perform a trend analysis. The result from the trend analysis for the entire time series (long-term) is shown before the slash "/", and the result for the last 10 years (short-term) is shown after the slash. Dark grey indicates concentrations higher than estimated high background levels. Light grey indicates concentrations lower than high background levels. Note: Class limits for ΣDDT are used for ppDDE. (See Appendix B for description of chemical codes.)*

Station	Station name	Species	Tissue	Ag	As	Co	Cd	Cr	Cu	Hg	Ni	Pb	Zn	PCB7	DDEPP	HBCDA	SCCP	MCCP	BDE	BDE47	BDE100	BDE209	KPAH	BKF	BGHIP	ICDP	BAP	FLU	PFOS	PFOSA	PFBS	TBT	VDSI
10A2	Skallneset	Blue mussel	SB	O/O	↓/↓	O/O	O/O	O/O	O/O	↓/O	O/O	O/O	O/O	↓/O	★/O																		
11X	Brashavn	Blue mussel	SB	↓/↓	O/O	O/O	O/O	O/O	O/O	O/O	O/O	O/O	O/O	O/O	★/O																		
15A	Gåsøy	Blue mussel	SB	O/O	↓/↓	O/O	O/O	O/O	O/O	O/O	O/O	O/O	↓/O	O/O																			
22A	Espevær	Blue mussel	SB	★/★	O/O	O/O	O/↓	O/O	O/O	O/↓	O/O	↓/↓	↓/O	↓/O	O/O																	↓/↓	
26A2	Måløy	Blue mussel	SB	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪		▪/▪	▪/▪	▪/▪		▪/▪	▪/▪	▪/▪											
30A	Gressholmen	Blue mussel	SB	O/O	O/O	O/O	O/↓	O/O	O/O	O/O	O/O	O/O	O/O	O/O	↓/O	▪/▪	▪/▪	▪/▪		↓/↓	O/O	★/★	O/O	★/★	★/↓	★/★	★/★	O/O				↓/↓	
31A	Solbergstrand	Blue mussel	SB	O/O	O/O	O/O	↓/↓	O/O	O/O	↓/O	O/O	↓/O	O/O	O/O	O/O																	▪/▪	
35A	Mølen	Blue mussel	SB	★/★	O/O	O/O	↓/O	O/O	O/↓	O/O	O/O	↓/O	O/↓	↓/O									O/O	★/★			★/★	O/O				O/O	
36A1	Tjøme	Blue mussel	SB	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪		▪/▪	▪/▪	▪/▪										▪/▪	
51A	Byrkjenes	Blue mussel	SB							O/O				O/O	O/O																		
52A	Eitrheimsneset	Blue mussel	SB	O/O	O/O	O/O	↓/↓	O/O	O/O	↓/O	O/O	O/O	↓/O	↓/O	O/O																		
56A	Kvalnes	Blue mussel	SB							O/O				↓/O	↑/O																		
57A	Krossanes	Blue mussel	SB	▪/▪	O/O	O/O	↓/↓	▪/▪	↓/↓	O/O	O/O	↓/↓	↓/↓	↓/O	O/O																		
63A	Ranaskjær	Blue mussel	SB	★/★	O/O	O/O	↓/↓	O/O	↓/O	↓/O	O/O	↓/O	↓/O	↓/O	O/O																		
64A	Utne	Blue mussel	SB	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪																		
65A	Vikingneset	Blue mussel	SB	★/★	O/O	O/O	↓/↓	O/O	O/O	O/O	O/O	↓/↓	↓/O	↓/O	O/O																		
69A	Lille Terøy	Blue mussel	SB	▪/▪	O/O	O/O	↓/↓	O/O	O/O	↓/↓	O/O	↓/↓	↓/↓	O/O																			
71A	Bjørkøya	Blue mussel	SB	★/★	↓/↓	O/O	↓/O	O/O	O/↓	↓/↑	O/O	O/↑	O/O		↓/O	▪/▪	▪/▪	▪/▪		O/O	O/O		▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪					
76A	Risøy	Blue mussel	SB	★/★	↓/↓	↓/↓	O/O	O/O	↓/↓	O/O	O/O	O/O	↓/O	↓/O	O/O																		
91A2	Outer Trondheimsfjord	Blue mussel	SB	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪		▪/▪	▪/▪	▪/▪		▪/▪	▪/▪	▪/▪											
97A2	Bodø harbour	Blue mussel	SB	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪		▪/▪	▪/▪	▪/▪		▪/▪	▪/▪	★/★											
98A2	Lofoten, Svolvær	Blue mussel	SB	O/O	O/O	O/O	O/O	O/O	O/O	O/O	O/O	O/O	O/O	↓/O		▪/▪	▪/▪	▪/▪		O/O	▪/▪	▪/▪	▪/▪	▪/▪	★/★	▪/▪	▪/▪	▪/▪					
I023	Singlekalven	Blue mussel	SB	★/★	O/O	O/O	O/O	O/O	O/O	O/O	O/O	O/O	O/O	↓/O		▪/▪	▪/▪	▪/▪		▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪					
I024	Kirkøy	Blue mussel	SB	▪/▪	▪/▪	▪/▪	O/↑	▪/▪	O/O	O/O	▪/▪	O/O	O/O	↓/O																			
I131A	Lastad	Blue mussel	SB	▪/▪	O/O	↓/↓	O/O	O/O	O/O	O/O	O/O	O/O	O/O										O/O	★/★	O/O	★/★	★/★	O/O					
I133	Odderøy	Blue mussel	SB	▪/▪	O/O	O/O	O/O	O/O	O/O	O/O	O/O	O/O	O/O	↓/↑	O/O																	↓/↓	
I241	Nordnes	Blue mussel	SB	★/★	O/O	O/O	↓/↓	O/O	O/O	↓/↓	O/O	↓/↓	O/O	O/O		▪/▪	▪/▪	▪/▪		▪/▪	▪/▪	▪/▪											
I301	Akershuskaia	Blue mussel	SB	★/★	O/O	O/O	↓/O	O/O	O/O	O/O	O/O	↓/O	↓/O	↓/O	O/O								O/O	O/O	O/O	★/★	★/★	O/O				↓/↓	
I304	Gåsøya	Blue mussel	SB	★/★	O/O	O/O	↑/↑	O/O	O/O	↑/↑	O/O	O/O	↓/O	↓/↓	O/O								O/O	★/★	★/★	★/★	★/★	O/O				▪/▪	
I306	Håøya	Blue mussel	SB	★/★	O/O	O/O	O/O	▪/▪	↓/↓	O/O	▪/▪	O/O	↓/O	↓/O																			
I307	Ramtonholmen	Blue mussel	SB	★/★	O/O	O/O	O/O	O/O	↓/↓	O/O	O/O	O/O	↓/↓	↓/↓																			
I712	Croftholmen	Blue mussel	SB	★/★	O/O	O/O	O/O	↓/↓	O/O	O/O	O/O	O/O	O/O		O/O	▪/▪	▪/▪	▪/▪		▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪					
I965	Moholmen	Blue mussel	SB	★/★	O/O	O/O	O/↓	↓/↓	O/O		↓/↓	↓/↓	O/O										O/O	O/O	O/O	O/O	O/O	O/O					
I969	Bjørnebærviken	Blue mussel	SB	O/O	O/O	O/O	O/O	O/O	O/O		O/O	O/O	O/O										O/↓	O/O	O/↓	O/O	O/O	O/O					

Station	Station name	Species	Tissue	Ag	As	Co	Cd	Cr	Cu	Hg	Ni	Pb	Zn	PCB7	DDEPP	HBCDA	SCCP	MCCP	BDE	BDE47	BDE100	BDE209	KPAH	BKF	BGHIP	ICDP	BAP	FLU	PFOS	PFOSA	PFBS	TBT	VDSI
02B	Hvaler	Cod	LI	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪		▪/▪	▪/▪	▪/▪	▪/▪		▪/▪	▪/▪	▪/▪															
10B	Varangerfjord	Cod	LI	O/O	O/O	O/O	🡻/O	★/★	🡻/🡻		★/★	★/★	🡻/O	🡻/🡻	🡻/🡻																	
13B	Kristiansand harbour	Cod	LI	O/O	🡻/🡻	O/O	O/O	O/O	O/O		🡻/🡻	★/★	O/O	O/O		▪/▪	▪/▪	▪/▪		O/O	O/O	★/★							🡻/🡻	★/★	★/★		
15B	Farsund area	Cod	LI	O/O	O/O	O/O	O/🡹	★/★	O/🡹		O/O	★/★	O/🡹	O/O	🡻/🡻																	
23B	Bømlo north	Cod	LI	O/O	🡻/🡻	O/O	O/O	★/★	O/O		★/★	★/★	O/O	🡻/O	🡻/O	▪/▪	O/O	O/O		🡻/O	🡻/O	★/★							O/🡻	★/★	★/★		
30B	Inner Oslofjord	Cod	LI	O/O	🡻/🡻	O/O	🡹/O	O/O	🡻/O		O/O	O/O	O/O	O/O	🡻/O	▪/▪	▪/▪	▪/▪		O/🡻	🡻/🡻	🡻/O							O/🡻	O/O	🡻/🡻		
36B	Færder area	Cod	LI	O/O	O/O	O/O	🡻/O	★/★	🡻/🡻		O/O	★/★	🡻/O	O/🡻	🡻/O	O/O	▪/▪	▪/▪		🡻/O	O/O	★/★							🡻/🡻	O/O	★/★		
43B2	Tromsø harbour	Cod	LI	🡹/🡹	🡻/🡻	O/O	🡻/🡻	★/★	O/O		O/O	★/★	🡻/🡻	🡻/🡻		▪/▪	▪/▪	▪/▪		🡻/🡻	🡻/🡻	★/★							🡻/🡻	★/★	★/★		
45B2	Hammerfest (havn)	Cod	LI	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪		▪/▪	▪/▪	▪/▪	▪/▪																		
53B	Inner Sørfjord	Cod	LI											O/O	O/O	▪/▪	🡻/O	O/O		O/O	O/O	★/★							🡻/🡻	★/★	★/★		
71B	Grenslandsfjord	Cod	LI	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪		▪/▪		▪/▪			▪/▪	▪/▪	▪/▪														
80B	Inner Trondheimsfjord	Cod	LI	🡻/🡻	O/O	O/O	O/O	O/★	O/O		O/O	★/★	O/O	🡻/🡻		▪/▪	▪/▪	▪/▪		O/O	O/O	▪/▪							🡻/🡻	O/O	★/★		
96B	Helgeland	Cod	LI	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪	▪/▪		▪/▪	▪/▪	▪/▪	▪/▪																		
98B1	Lofoten, Skrova	Cod	LI	🡹/🡹	🡹/🡹	🡹/🡹	O/🡹	★/★	O/O		O/O	★/★	🡹/🡹	🡻/🡻	🡻/🡻	▪/▪	▪/▪	▪/▪		🡻/O	O/O	★/★							O/O	★/★	★/★		
02B	Hvaler	Cod	MU							▪/▪																						
10B	Varangerfjord	Cod	MU							🡻/O																						
13B	Kristiansand harbour	Cod	MU							O/O																						
15B	Farsund area	Cod	MU							O/🡹																						
23B	Bømlo north	Cod	MU							🡹/🡹																						
30B	Inner Oslofjord	Cod	MU							🡹/🡹																						
36B	Færder area	Cod	MU							O/O																						
43B2	Tromsø harbour	Cod	MU							O/O																						
45B2	Hammerfest (havn)	Cod	MU							▪/▪																						
53B	Inner Sørfjord	Cod	MU							O/O																						
71B	Grenslandsfjord	Cod	MU							▪/▪																						
80B	Inner Trondheimsfjord	Cod	MU							O/O																						
96B	Helgeland	Cod	MU							▪/▪																						
98B1	Lofoten, Skrova	Cod	MU							O/O																						
71G	Fugløyskjær	Common periwinkle	SB																												O/O	
11G	Brashavn	Dog Whelk	SB																												★/★	
131G	Lastad	Dog Whelk	SB																												★/★	
15G	Gåsøy	Dog Whelk	SB																												★/★	
227G1	Melandsholmen	Dog Whelk	SB																												🡻/🡻	
22G	Espevær	Dog Whelk	SB																												🡻/🡻	
36G	Færder	Dog Whelk	SB																												🡻/🡻	
76G	Risøy	Dog Whelk	SB																												🡻/🡻	
98G	Lofoten, Svolvær	Dog Whelk	SB																												★/★	
11G	Brashavn	Dog Whelk	WO																													O/O
131G	Lastad	Dog Whelk	WO																													🡻/🡻
15G	Gåsøy	Dog Whelk	WO																													🡻/🡻
227G1	Melandsholmen	Dog Whelk	WO																													🡻/🡻
22G	Espevær	Dog Whelk	WO																													🡻/🡻
36G	Færder	Dog Whelk	WO																													🡻/🡻
76G	Risøy	Dog Whelk	WO																													🡻/🡻
98G	Lofoten, Svolvær	Dog Whelk	WO																													🡻/🡻

3.2 Levels and trends

3.2.1 Mercury (Hg)

Mercury (Hg) was analysed in cod fillet at 14 stations and in blue mussel at 32 stations.

Important levels exceeding Class I

Cod fillet were moderately polluted (Class II) with Hg in the Inner Oslofjord (st. 30B), Færder (st. 36B), Hvaler (st. 02B), Grenlandsfjord (st. 71B), Farsund (st. 15B), the Inner Sørfjord (st. 53B) and Bømlo (st. 23B). Blue mussel at Byrkjenes (st. 51A) in the Inner Sørfjord was also moderately polluted with Hg. All other blue mussel showed background levels (Class I) of Hg.

Class increased since 2013

The concentration of Hg in cod fillet from Hvaler (st. 02B) had increased from being insignificantly polluted (Class I, 0.089 mg/kg w.w.) in 2013 to being moderately polluted (Class II, 0.129 mg/kg w.w.) in 2014. Blue mussel from Byrkjenes (st. 51A) had also increased from being insignificantly polluted (0.038 mg/kg w.w.) with Hg in 2013 to being moderately polluted (0.071 mg/kg w.w.) in 2014. In addition, no trends were found in 2014. In 2013, a significant downward long-term trend was found at Byrkjenes.

Upward trends

Cod fillet from the Inner Oslofjord (st. 30B) was moderately polluted (Class II) with Hg and showed both significant upward long-term and short-term trends (***Table 10***, ***Figure 8***). The median concentration had decreased from 0.318 mg/kg w.w. in 2013 to 0.207 mg/kg w.w. in 2014.

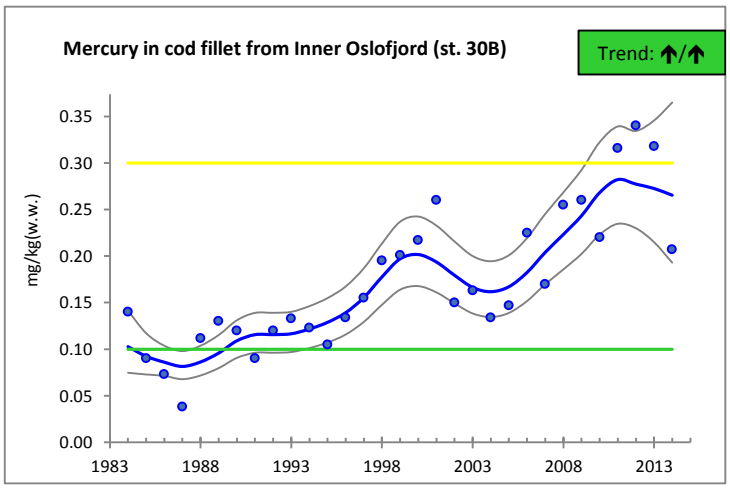

Figure 8. *Median concentrations (mg/kg w.w.) of Hg in cod fillet from 1984 to 2014 in the Inner Oslofjord (st. 30B).*

Cod fillet from Bømlo (st. 23B) was also moderately polluted (Class II) with Hg and showed both significant upward long-term and short-term trends. In 2013, no significant long-term trend was found. Cod fillet from Farsund area (st. 15B) was also moderately polluted (Class II) with Hg and showed significant upward short-term trends.

Blue mussel at Bjørkøya (st. 71A) in the Grenlandsfjord area had background levels (Class I) in 2014, but a significant upward short-term trend was observed. Mussels in the Inner Oslofjord at Gåsøya (st. I304) showed significant upward long-term and short-term trends but within background levels.

Class decreased since 2013

Hg-concentrations in cod fillet from the Inner Oslofjord (st. 30B) and Grenlandsfjord (st. 71B) had decreased from being markedly polluted (Class III) in 2013 to being moderately polluted (Class II) in 2014. Hg-concentrations in cod fillet from Kristiansand harbour (st. 13B) and blue mussel from Kvalnes (st. 56A) in the Mid Sørfjord had decreased from being moderately polluted (Class II) in 2013, to being insignificantly polluted (Class I) in 2014.

Downward trends/low levels

There was a significant downward long-term trend in cod fillet from the Varangerfjord in 2014 and the concentration was at background levels (Class I). In 2013, a significant upward short-term trend was observed at this station, but no short-term trend was found in 2014.

Significant downward long-term trends within background levels were observed in mussel from the Oslofjord at Solbergstrand (st. 31A), in the Grenlandsfjord at Bjørkøya (st. 71A) ***(Table 10, Figure 9)***, in the Sørfjord at Eitrheimsneset (st. 52A), in the Hardangerfjord at Ranaskjær (st. 63A) and Lille Terøy (st. 69A), at Nordnes (st. I241) close to Bergen and in the Varangerfjord at Skallneset (st. 10A2). Significant downward short-term trends were found at Lille Terøy (st. 69A), Espevær (st. 22A) and Nordnes (st. I241).

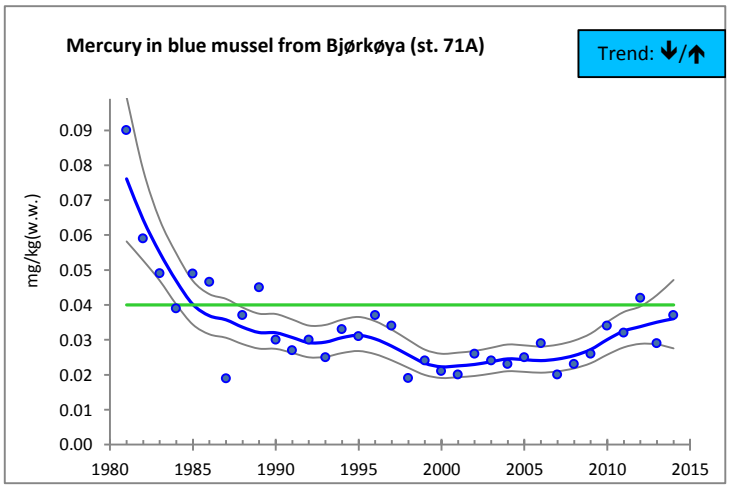

Figure 9. *Median concentrations (mg/kg w.w.) of mercury in blue mussel from 1981 to 2014 at Bjørkøya (st. 71A) in the Grenlandsfjord area.*

General, large scale trends

For the period 1990-2006, OSPAR (2010) found 70-75% reduction in riverine and direct discharges of Hg to the North Sea, and sediment from the North Sea showed a predominance of downward over upward significant trends. This reduction is not so evident for the Norwegian discharges. For MILKYS long-term trends, there is some evidence of downward trends. Seven downward trends and one upward trend were found in blue mussel. However, three trends were found in cod fillet; upwards in the Inner Oslofjord and at Bømlo and downwards in the Varangerfjord.

Total riverine inputs of Hg in Norway have been in the range of 139 to 259 kg in 2013 (Skarbøvik *et al*. 2014). The range of total riverine inputs of Hg were 60 to 97 kg to Skagerrak, 31 to 64 kg to the North sea, 43 to 77 kg to the Norwegian Sea and 4 to 21 kg to the Barents sea (Skarbøvik *et al*. 2014), indicating higher input in the southern part of Norway. In addition to riverine inputs was the contribution by direct discharges from sewage and industrial effluents amounting to 18 kg or about 7 % of the total (276 kg) (Skarbøvik *et al*. 2014). No trend analysis was made for the period 1990-2013.

When considering the total of 37 recent short-term (2005-2014) trends for both cod and blue mussel, significant trends are limited to upwards at five stations and downwards at three stations (***Table 10***, ***Figure 10***).

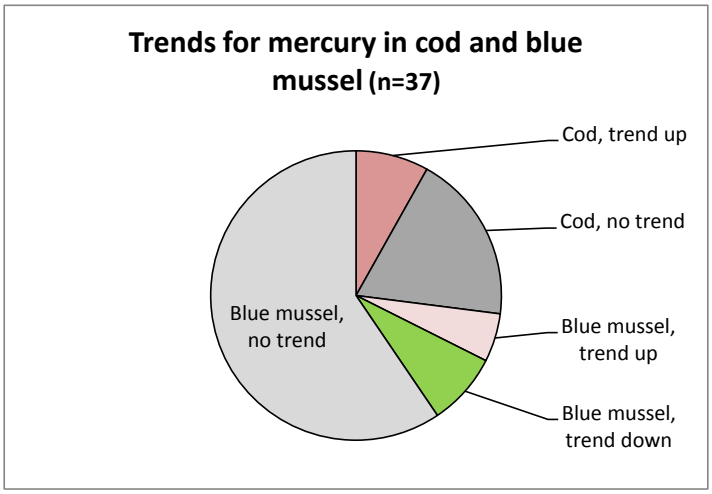

Figure 10. *Frequency of short-term (recent) trends (2005-2014) for Hg in cod fillet and blue mussel.*

Emissions of Hg to air from land-based industries showed essentially a decrease from 2002 (257 kg Hg/year) to 2009 (104 kg Hg/year), and the emission was 100 kg Hg/year in 2014 (***Figure 11***). Changes in emissions do not provided an explanation for the 2005-2014 increasing short-term trends. The emissions to air varied between 260 kg Hg/year in 2004 to 100 kg Hg/year in 2014 in the period 2002-2014.

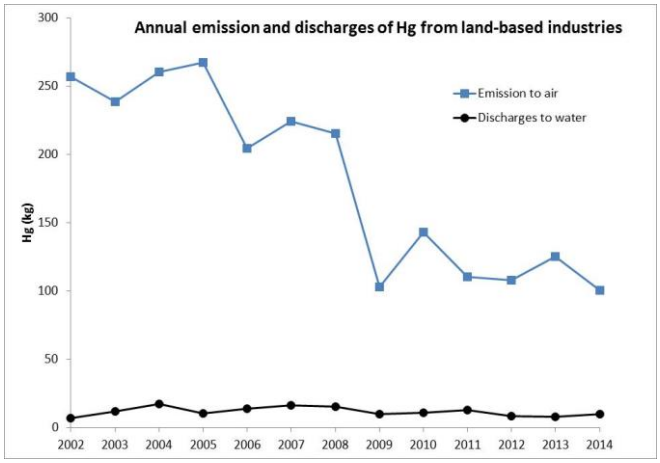

Figure 11. *Annual emissions of Hg to air and discharges to water from land-based industries for the period 2002-2014 (data from www.norskeutslipp.no). Note that this category excludes emissions and discharges from municipal treatment plants, land runoff, transportation and offshore industry.*

There is some indication that Norwegian atmospheric deposition in Southern Norway is decreasing for the period 1995-2006,but this was not statistically confirmed (Wängberg *et al*. 2010).. Still we see upward short-term and long-term trends in cod fillet from the Inner Oslofjord and Bømlo, and significant upward short-term trend at Farsund. Possible explanations of increasing trends could be related to factors such as; climate change, more favourable conditions for methyl mercury formation, increased bioavailability of Hg stored in the sediments, increased access of cod to contaminated feeding areas due to improved oxygen levels in deep water, changes in what the cod eat, etc. It has also been argued that the increasing trend in Inner Oslofjord might be a result of sediment remediation works in Oslo harbour in 2006-2008. Neither explanation can be ruled out based on existing knowledge, but the monitoring designed to reveal spreading of mercury during the dredging operations (Berge, 2014) gave little evidence to support the latter hypotheses. Neither can it explain why Hg is the only contaminant, with the exception Cd for long term trend, showing an upward trend in the in cod from the Inner Oslofjord. Before speculating to much in potential causes, the nature of the trend data will be further investigated below.

Other studies

Blue mussel at Gåsøya in the Inner Oslofjord showed both significant upward long-term and short-term trends of Hg, but within background levels (Class I). Blue mussel at Frognerkilen, the mouth of Alna and Bekkelaget were also at background levels for Hg in 2014 (Ruus *et al*. 2015) and 2013 (Ruus *et al*. 2014).

Other investigations in the Inner Oslofjord (Berge 2014) also found insignificantly polluted (Class I) blue mussel, and that cod fillet from Bekkelaget and Frognerkilen was moderately polluted (Class II) in 2010 and 2013. Atmospheric deposition is a major source to the seas surrounding Norway and considerably larger than other sources such as riverine discharges, shipping and offshore installations (Green *et al*. 2013). Historical data on entry of Hg to the Inner Oslofjord is not available. Bjerkeng *et al*. (2009) found that more than 60 % of the Hg input to Bunnefjorden was from atmospheric deposition. Present discharge of Hg to the Inner Oslofjord has been calculated to be around 7.3 kg/year (Berge *et al*. 2013b).

Blue mussel from Langøya in the Holmestrandfjord in 2014 was at background levels (Class I) for Hg (Gitmark *et al*. 2015). There are limited data for other recent surveys and data for 2014 was not found. Hg in cod fillet was still declining in the Grenlandsfjord during the period from 2008 to 2012, but the level in the Frierfjord was still higher than in 1999 (Ruus *et al*. 2013a). Blue mussel at seven stations in the Kristiansandsfjord in 2013 was insignificantly polluted (Class I) or slightly above (Schøyen *et al*. 2014). The concentrations of metals and Hg in blue mussel in the Sørfjord have decreased significantly during the last 25 years due to remedial actions performed by the local industry (Ruus *et al*. 2013b).

Concentrations of Hg in cod from the Barents sea collected during 1976, 1995 and 2000 did not seem to have increased in the period of 25 years (Ervik *et al*. 2003).

Most of the Hg-pollution in Norwegian lakes is now due to atmospherically deposited Hg originating from other parts of the world (Fjeld *et al*. 2015). The concentration of Hg in trout from Mjøsa showed a decreasing trend in the period 1980-2005, and had been more or less unchanged in the period 2006-2013 (Fjeld *et al*. 2015, Løvik *et al*. 2015). Surveys from 2008 suggests that the length adjusted average Hg-concentrations in ten perch populations from forest lakes, increased with 63 % since the early 1990s (Fjeld & Rognerud 2009).

Environmental Quality Standards (EQS)

EU has provided Environmental Quality Standard (EQS) of 0.02 mg/kg w.w. in biota for "fish" (cf. ***Table 7***) which is below the upper limit of insignificantly polluted (Class I) blue mussel (0.04 mg/kg w.w.). Applying this EQS for blue mussel, concentrations of Hg were above or at the EQS applied for biota at Singlekalven (st. I023, 0.022 mg/kg w.w.) and Kirkøy (st. I024, 0.028 mg/kg w.w.) in the Hvaler area, and Bjørkøya (st. 71A, 0.037 mg/kg w.w.) and Croftholmen (st. I712, 0.027 mg/kg w.w.) in the Grenlandsfjord-area. This was also the result at Odderøy (st. I133, 0.021 mg/kg w.w.) in the Kristiansandsfjord. This was also the case at Byrkjenes (st. 51A, 0.071 mg/kg w.w.), Eitrheimsneset (st. 52A, 0.033 mg/kg w.w.), Kvalnes (st. 56A, 0.039 mg/kg w.w.) and Krossanes (st. 57A, 0.032 mg/kg w.w.) in the Sørfjord, and in the Hardangerfjord at Vikingneset (st. 65A, 0.02 mg/kg w.w.).

The EQS for fish are based on analyses on whole fish. Therefore, the EQS cannot be directly compared to concentrations found in certain tissues of fish. We have in this study only measured Hg in fillet. Converting concentrations in fillet to concentrations in whole fish is uncertain, and would probably be an overestimate because Hg accumulates more in the fillet than in other tissues (Kwasniak & Falkowska 2012). If it is assumed, for this exercise, that the same concentration is found in all tissue types, then the results of Hg (in cod fillet) would have exceeded the EQS (0.020 mg/kg w.w.) for all 2014-samples (except for the Varangerfjord st. 10B where the concentration was at the EQS-limit 0.02 mg/kg w.w.), as it did for all 2013 and 2012-samples.

3.2.2 Cadmium (Cd)

Cadmium (Cd) was analysed in cod liver at 13 stations and in blue mussel at 32 stations.

Levels exceeding Class I

All cod liver was at background levels and all blue mussel was insignificantly polluted (Class I) in 2014, as in 2013.

Upward trends

For cod liver, there was significant upward long-term trend in the Inner Oslofjord (st. 30B) and significant upward short-term trends at Farsund (st. 15B) and in Lofoten (st. 98B1) (***Table 10***).

There were both significant upward long-term and short-term trends in blue mussel at Gåsøya (st. I304) and a short-term upward trend in blue mussel at Kirkøy (st. I024) in the Hvaler area.

Downward trends/low levels

All concentrations of Cd in cod liver and blue mussel were low, i.e. within background levels (Appendix C). There were significant downward long-term trends of Cd in cod liver from Færder (st. 36B), Tromsø harbour (st. 43B2) and in the Varangerfjord (st. 10B). There was also a significant downward short-term trend in cod liver from Tromsø harbour (st. 43B2).

In blue mussel, there were significant downward long-term and short-term trends at Solbergstrand (st. 31A) in the Mid Oslofjord and at Eitrheimsneset (st. 52A) in the Inner Sørfjord and at Krossanes (st. 57A) in the Outer Sørfjord. This was also the result at Ranaskjær (st. 63A), Vikingneset (st. 65A) and Lille Terøy (st. 69A) in the Hardangerfjord, and at Nordnes (st. I241) close to Bergen. There were significant downward long-term trends for blue mussel at Akershuskaia (st. I301) in the Inner Oslofjord, at Mølen (st. 35A) in the Mid Oslofjord and at Bjørkøya (st. 71A) in the Grenlandsfjord. There were significant downward short-term trends for blue mussel at Gressholmen (st. 30A) in The Inner Oslofjord, at Espevær (st. 22A) on the west coast and at Moholmen (st. I965) in the Ranfjord.

Other studies

Blue mussel at Frognerkilen, the mouth of Alna and Bekkelaget were also at background levels for Cd in 2014 (Ruus *et al*. 2015) and 2013 (Ruus *et al*. 2014). Other reports have also shown blue mussel insignificantly polluted with Cd in the Inner Oslofjord 2006-2013 (Berge 2014). Mussels were, however, up to moderately polluted with Cd at Langøya in the Holmestrandfjord in 2014 (Gitmark *et al*. 2015). Blue mussel at all seven stations in the Kristiansandsfjord was at background levels in the period 2010 to 2013 (Schøyen *et al*. 2014).

General, large scale

Discharges of Cd to water from land-based industries showed a decrease from 2007 (686 kg Cd/year) to 2014 (257 kg Cd/year) (***Figure 12***). The emission of Cd to air showed a gradually decrease from 2002 (352 kg Cd/year) to 2014 (51 kg Cd/year).

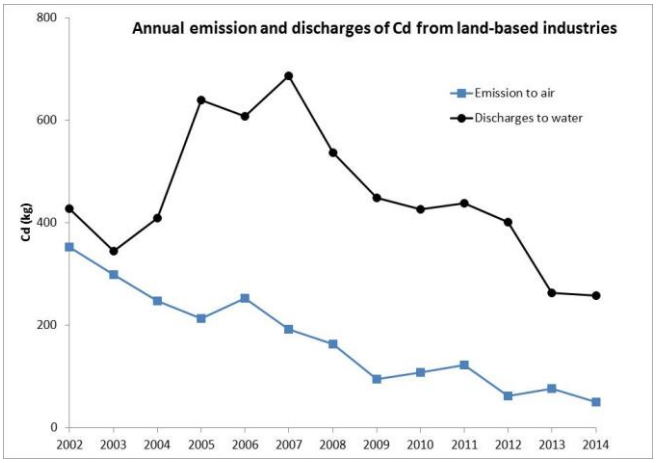

Figure 12. *Annual emissions of Cd to air and discharges to water from land-based industries in the period 2002-2014 (data from www.norskeutslipp.no). Note that this category excludes emissions and discharges from municipal treatment plants, land runoff, transportation and offshore industry.*

The discharge of Cd to water from local industry in Skien has gradually increased from 0.01 kg/year in 2004 to 0.12 kg/year in 2014 (www.norskeutslipp.no). The discharge of Cd to water from local industry in Odda in the Inner Sørfjord has decreased from 130 kg/year in 2007 to between 30 and 40 kg/year in the period 2008-2014 (www.norskeutslipp.no).

Total riverine inputs of Cd in Norway have been in the range of 1.92 to 2.13 tonnes in 2013 (Skarbøvik *et al*. 2014). The range of total riverine inputs were 0.93 to 0.94 tonnes Cd to Skagerrak, 0.44 to 0.46 tonnes Cd to the North sea, 0.32 to 0.44 tonnes Cd to the Norwegian Sea and 0.23 to 0.28 tonnes Cd to the Barents sea (Skarbøvik *et al*. 2014), indicating higher input in the southern part of Norway. In addition to riverine inputs, is the contribution by direct discharges from sewage and industrial effluents amounting to 0.10 tonnes or about 5 % of the total (2 tonnes) (Skarbøvik *et al*. 2014).

3.2.3 Lead (Pb)

Lead (Pb) was analysed in cod liver at 13 stations and in blue mussel at 32 stations.

Important levels exceeding background levels

Cod liver from the Inner Oslofjord (st. 30B) exceeded background levels of Pb.

Important levels exceeding Class I

The presence of Pb in blue mussel exceeded Class I (insignificantly polluted) at four of the blue mussel stations (***Table 10***). The highest level (2.7 mg Pb/kg w.w.) was found in blue mussel from Eitrheimsneset (st. 52A) in the Inner Sørfjord and they were moderately polluted (Class II). Blue mussel at Odderøy (st. I133) in the Kristiansandsfjord, Krossanes (st. 57A) in the Outer Sørfjord and Moholmen (st. I965) in the Ranfjord were also moderately polluted (Class II) with Pb.

At Odderøy, both significant upward long-term and short-term trends were found in 2013. In 2014, no significant trends were found. At Eitrheimsneset, there was a significant downward long-term trend in 2013. In 2014, no significant trends were found at this location.

Levels increased since 2013
Cod liver from the Inner Oslofjord (st. 30B) was at background levels in 2013, but exceeded background levels in 2014. In 2013, both significant downward long-term and short-term trends were found in the Inner Oslofjord. In 2014, no significant trends were found at this location.

Class increased since 2013
The Pb-concentrations in blue mussel at Krossanes (st. 30A) in the Outer Sørfjord had increased from being on background level (Class I) in 2013 to being moderately polluted (Class II) in 2014.

Upward trends
Blue mussel from Bjørkøya (st. 71A) in the Grenlandsfjord showed significant upward short-term trend.

Downward trends/low levels
Observed concentrations of Pb in cod liver were at background level at all stations, except for the Inner Oslofjord (st. 30B) where background level was exceeded (***Table 10***). No significant trends were found in cod liver from the Inner Oslofjord. At eight stations, data was inadequate for trend analysis in cod liver due to concerns about the limit of detections.

Of the trend series performed for blue mussel, 10 revealed significant downward long-term trends. Both significant downward long-term and short-term trends were observed for mussel at Krossanes (st. 57A) in the Outer Sørfjord and Moholmen (st. I965) in the Ranfjord, and both were moderately polluted (Class II). Both significant downward long-term and short-term trends were also observed for mussel at Vikingneset (st. 65A) and Lille Terøy (st. 69A) in the Hardangerfjord, at Espevær (st. 22A), and at Nordnes (st. I241) close to Bergen, all within background levels (Class I). Significant downward long-term trends were observed in blue mussel at Akershuskaia (st. I301) in the Inner Oslofjord, at Solbergstrand (st. 31A) and Mølen (st. 35A) in the Mid Oslofjord, and at Ranaskjær (st. 63A) in the Hardangerfjord, all within background concentrations.

Other studies
Blue mussel at Frognerkilen, the mouth of Alna and Bekkelaget were markedly polluted (Class III) for Pb in 2014 (Ruus *et al*. 2015) and at background levels (Class I) in 2013 (Ruus *et al*. 2014). Monitoring of mussels in the Inner Oslofjord in 2006 to 2013 showed that mussels were up to moderately polluted (Class II) with Pb (Berge 2014) and that mussels were up to moderately polluted with Pb from Langøya in the Holmestrandfjord in 2014 (Gitmark *et al*. 2015).

Blue mussel from Odderøy in the Kristiansandsfjord was markedly polluted with Pb in 2012 and 2013, while mussels in the inner fjord were insignificantly polluted and mussels in the outer fjord were moderately polluted (Schøyen *et al*. 2014).

General, large scale
There were low levels of Pb in cod liver and significant downward long-term trends from 10 areas (Oslo harbour, Mid Oslofjord, Outer Sørfjord, Inner/Mid/Outer Hardangerfjord, Bergen harbour and Ranfjord), even in the vicinity of highly populated areas such as Oslo. EU banned leaded-fuel in road vehicles 1 January 2000, but some countries had banned the fuel beforehand (e.g. Sweden, Germany, Portugal). The results indicate that the ban of Pb in gasoline has had a positive effect.

OSPAR (2010) found 50-80% reduction in riverine and direct discharges of Pb to the North Sea for the period 1990-2006. Total riverine inputs of Pb in Norway have been in the range of 36.23 to 36.29 tonnes in 2013 (Skarbøvik *et al*. 2014). The range of total riverine inputs were 17.56 tonnes Pb to

Skagerrak, 8.99 to 9.00 tonnes Pb to the North sea, 6.81 to 6.83 tonnes Pb to the Norwegian Sea and 2.87 to 2.90 tonnes Pb to the Barents sea (Skarbøvik *et al*. 2014), indicating higher input in the southern part of Norway. The upper estimate of the total Pb load dropped 35 % to 36 tonnes in 2013 compared to the mean for the period 1990-2012 (55 tonnes). In addition to riverine inputs, comes the contribution by direct discharges from sewage and industrial effluents amounting to 1.44 tonnes or about 4 % of the total (38 tonnes) (Skarbøvik *et al*. 2014).

Discharges of Pb to water from land-based industries showed a decrease from 2010 (6841 kg Pb/year) to 2014 (1359 kg Pb/year) (***Figure 13***).

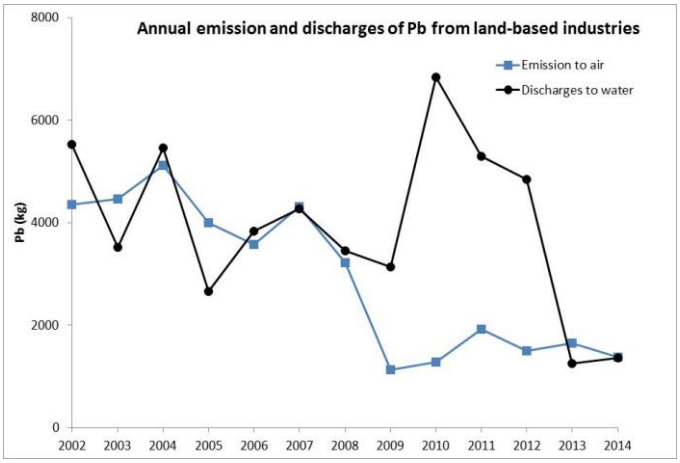

Figure 13. *Annual emissions of Pb to air and discharges to water from land-based industries in the period 2002-2014 (data from www.norskeutslipp.no). Note that this category excludes emissions and discharges from municipal treatment plants, land runoff, transportation and offshore industry.*

3.2.4 Copper (Cu)

Copper (Cu) was analysed in cod liver at 13 stations and in blue mussel at 32 stations.

Important levels exceeding Class I

All blue mussel was insignificantly polluted (Class I) with Cu. Cod liver from all stations had Cu-concentrations at background levels in 2014, as observed in 2013.

Class decreased since 2013

Concentrations of Cu in blue mussel at Gressholmen (st. 30A) in the Inner Oslofjord and Bodø harbour (st. 97A2) had decreased to background levels (Class I) in 2014 from being moderately polluted (Class II) in 2013.

Upward trends

In cod liver from the Farsund area (st. 15B), a significant upward short-time trend was found.

Downward trends/low levels

Cod liver from all stations had Cu-concentrations at background levels in 2014, as observed in 2013. A significant downward long-term trend was observed in cod liver from the Inner Oslofjord (st. 30B). In the Færder area (st. 36B) and in the Varangerfjord (st. 10B) there were both significant downward short-term and long-term trends in cod liver.

In blue mussel, both significant downward long-term and short-term trends were observed at Ramtonholmen (st. I307) and Håøya (st. I306) in the Inner Oslofjord, and at Risøy (st. 76A2). This was also the result at Krossanes (st. 57A) in the Outer Sørfjord. A significant downward long-term trend was found at Ranaskjær (st. 63A) in the Hardangerfjord and significant downward short-term trends were found at Mølen (st.35 A) in the Mid Oslofjord and at Bjørkøya (st. 71A) in the Grenlandsfjord. The concentrations of Cu at all seven blue mussel stations in the Kristiansandsfjord in 2013 were at background levels (Schøyen *et al*. 2014).

Other studies

Blue mussel at Frognerkilen, the mouth of Alna and Bekkelaget were at background levels for Cu in 2014 (Ruus *et al*. 2015) and 2013 (Ruus *et al*. 2014). Blue mussel from the Inner Oslofjord was up to moderately polluted with Cu in 2013 (Berge 2014). All blue mussel stations at Langøya in the Holmestrandfjord had background concentrations of Cu in 2014 (Gitmark *et al*. 2015). The concentrations of Cu at all seven blue mussel stations in the Kristiansandsfjord in 2013 were at background levels (Schøyen *et al*. 2014).

General, large scale

Discharges of Cu to water from land-based industries showed a gradually decrease from 2005 (90 186 kg Cu/year) to 2014 (42 656 kg Cu/year) (***Figure 14***).

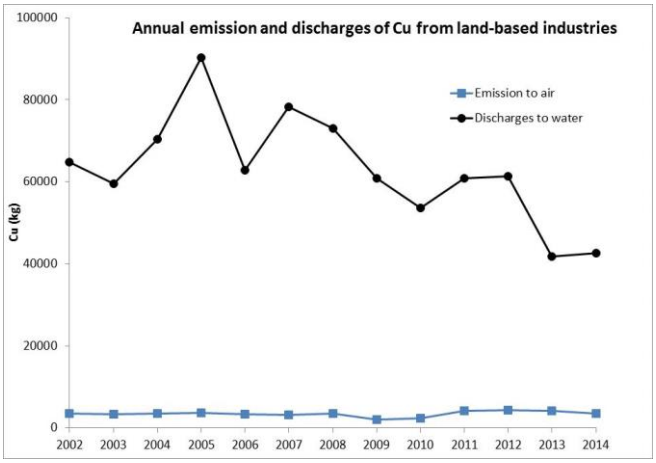

Figure 14. *Annual emissions of Cu to air and discharges to water from land-based industries in the period 2002-2014 (data from www.norskeutslipp.no). Note that this category excludes emissions and discharges from municipal treatment plants, land runoff, transportation and offshore industry.*

Upper average of total riverine inputs of Cu in Norway has been 236.97 tonnes in 2013 (Skarbøvik *et al*. 2014). The total riverine inputs of Cu were 94.82 tonnes to Skagerrak, 24.75 to 24.77 tonnes to the North sea, 47.89 to 47.90 tonnes to the Norwegian Sea and 69.48 tonnes to the Barents sea (Skarbøvik *et al*. 2014). The upper estimate of the total Cu load increased 6 % to 237 tonnes in 2013 compared to the mean for the period 1990-2012 (223 tonnes). In addition to riverine inputs, comes the contribution by direct discharges from sewage and industrial effluents and fish farming amounting to 926.42 tonnes or about 80 % of the total (1163 tonnes) (Skarbøvik *et al*. 2014).

3.2.5 Zinc (Zn)

Zinc (Zn) was analysed in cod liver at 13 stations and in blue mussel at 32 stations.

Important levels exceeding Class I or background level
Cod liver from Færder (st. 36B), Grenland (st. 71B), Kristiansand harbour (st. 13B), Farsund (st. 15B), Bømlo (st. 23B) and Lofoten (st. 98B1) had concentrations that exceeded background levels.

All blue mussel was at background levels (Class I).

Class increased since 2013
Concentrations of Zn in cod liver at Færder (st. 36B), Farsund (st. 15B), Bømlo (st. 23B) and Lofoten (st. 98B1) exceeded background levels in 2014, but not in 2013.

Upward trends
Both significant upward long-term and short-term trends for Zn were found in cod liver from Lofoten (st. 98B1). A significant upward short-term trend was observed in cod liver from Farsund (st. 15B). In 2013, no significant trends were found at this location.

No upward trends were found in blue mussel.

Class decreased since 2013
Observed concentrations of Zn in blue mussel from Moholmen (st. I965) in the Ranfjord revealed background levels (Class I) in 2014, but were moderately polluted (Class II) in 2013.

Downward trends/low levels
Cod liver from Tromsø harbour (st. 43B2) showed both significant downward long-term and short-term trends. Significant downward long-term trends were found in cod liver at Færder (st. 36B) and in the Varangerfjord (st. 10B).

Both significant long-term and short-term trends in blue mussel for Zn were found at Ramtonholmen (st. I307) in the Inner Oslofjord, Krossanes (st. 57A) in the Outer Sørfjord and at Lille Terøy (st. 69A) in the Hardangerfjord. Significant downward long-term trends were found in blue mussel from Akershuskaia (st. I301), Gåsøya (st. I304) and Håøya (st. I306) in the Inner Oslofjord. This was also the result at Risøy (st. 76A2) and Gåsøy (st. 15A) close to Farsund. This was also the case at Eitrheimsneset (st. 52A) in the Inner Sørfjord, and at Ranaskjær (st. 63A), Vikingneset (st. 65A) in the Hardangerfjord and at Espevær (st. 22A).

Other studies
Other studies also documented low levels of Zn in blue mussel. Blue mussel at Frognerkilen, the mouth of Alna and Bekkelaget were at background levels for Zn in 2014 (Ruus *et al*. 2015) and 2013 (Ruus *et al*. 2014). All blue mussels had background levels (Class I) of Zn at Langøya in the Holmestrandfjord in 2014 (Gitmark *et al*. 2015). All seven blue mussel stations in the Kristiansandsfjord were insignificantly polluted by Zn in the period 2010 to 2013 (Schøyen *et al*. 2014).

General, large scale
Discharges of Zn to water from land-based industries showed a gradually decrease from 2005 (200 785 kg Zn/year) to 2014 (78 223 kg Zn/year) (***Figure 15***).

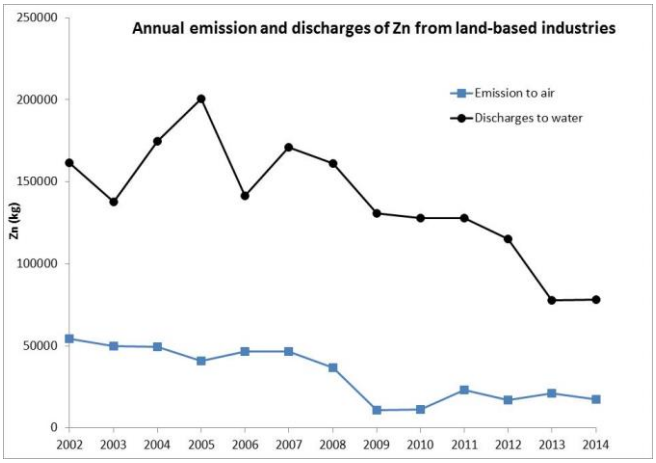

Figure 15. *Annual emissions of Zn to air and discharges to water from land-based industries in the period 2002-2014 (data from www.norskeutslipp.no). Note that this category excludes emissions and discharges from municipal treatment plants, land runoff, transportation and offshore industry.*

Upper average of total riverine inputs of Zn in Norway has been 925.18 tonnes in 2013 (Skarbøvik *et al*. 2014). Upper average of total riverine inputs of Zn were 662.89 tonnes to Skagerrak, 103.19 tonnes to the North sea, 127.52 tonnes to the Norwegian Sea and 31.57 tonnes to the Barents sea (Skarbøvik *et al*. 2014), indicating higher input in the southern part of Norway. The upper estimate of the total Zn load increased 21 % to 925 tonnes in 2013 compared to the mean for the period 1990-2012 (727 tonnes). In addition to riverine inputs, comes the contribution by direct discharges from sewage and industrial effluents amounting to 26.52 tonnes or about 3 % of the total (952 tonnes) (Skarbøvik *et al*. 2014).

3.2.6 Silver (Ag)

Silver (Ag) was analysed in cod liver at 13 stations and in blue mussel at 32 stations.

Levels

There were no changes in classes for Ag in blue mussel from 2013 to 2014, and only background levels (Class I) were observed. The environmental classifications system does not include Ag in cod. The highest concentration (6.7 mg/kg w.w.) in cod liver was found in cod from the Inner Oslofjord, as in 2013 (4.2 mg/kg w.w.). The second highest concentration (1.05 mg/kg w.w.) was found in cod liver from Lofoten (st. 98B1), as in 2013. The lowest concentration (0.061 mg/kg w.w.) was found in the Inner Trondheimsfjord (st. 80B), as in 2013.

Upward trends

There were both significant upward long-term and short-term-trends in cod liver from Lofoten (st. 98B1) and Tromsø harbour (st. 43B2).

Downward trends

There were both significant downward long-term and short-term-trends in cod liver from the Inner Trondheimsfjord (st. 80B) and in blue mussel from Brashavn (st. 11X).

Other studies

The highest Ag-concentrations were found in cod from the Inner Oslofjord in both 2014 and 2013. Equivalent concentration in the gills of Atlantic salmon was found to be lethal (Farmen *et al*. 2012), which indicates the need for a classification system to assess the possible effects in cod. There are no historical data on the amounts of Ag entering the Inner Oslofjord.

Blue mussel at Frognerkilen, the mouth of Alna and Bekkelaget in 2014 (Ruus *et al*. 2015) and 2013 (Ruus *et al*. 2014) were all at background levels (Class I).

Another investigation showed that blue mussel from seven stations in the Kristiansandsfjord was insignificantly polluted (Class I) by Ag in 2013 (Schøyen *et al*. 2014).

Discharges of wastewater treatment plants and discharges from mine tailings are considered major and important sources of silver to the aquatic environment (Tappin *et al*. 2010). The incorporation of silver nanoparticles into consumer products is of clear concern in terms of inputs to wastewater treatment plants (Nowack 2010). Silver has very low toxicity to humans; however this is not the case for microbe and invertebrate communities. There is increasing focus on the occurrence of Ag in both wastewater treatment plant effluent and sludge due to the increasing use of nanosilver in consumer products. Recent studies have shown that much of the silver entering wastewater treatment plants is incorporated into sludge as silver sulphide nanoparticles (Ag_2S), although little is known about the species that occurs in discharged effluent (Kim *et al*. 2010, Nowack 2010). From a study of eight Norwegian wastewater treatment plants, concentrations of silver in effluent ranged from 0.01 to 0.49 µg/L, and concentrations in sludge ranged from <0.01 to 9.55 µg/g (Thomas *et al*. 2011).

General, large scale

Discharges of Ag to water from land-based industries showed a decrease from 2005 (2.36 kg Ag/year) to 2009 (0.1 kg Ag/year), and then a gradually increase to 2014 (0.6 kg Ag/year) (***Figure 16***).

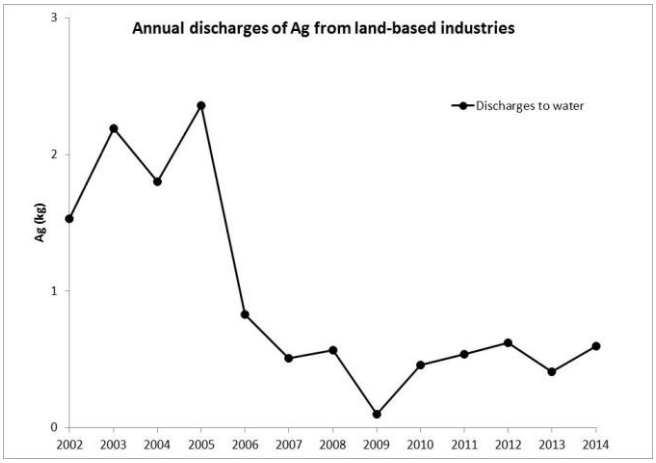

Figure 16. *Annual discharges of Ag to water from land-based industries in the period 2002-2014 (data from www.norskeutslipp.no). Note that this category excludes discharges from municipal treatment plants, land runoff, transportation and offshore industry.*

Total riverine inputs of Ag in Norway have been in the range of 0.02 to 7.09 tonnes in 2013 (Skarbøvik *et al*. 2014). Upper average of total riverine inputs of Ag were 3.22 tonnes to Skagerrak, 1.38 tonnes to the North sea, 1.82 tonnes to the Norwegian Sea and 0.66 tonnes to the Barents sea (Skarbøvik *et al*. 2014), indicating higher input in the southern part of Norway.

3.2.7 Arsenic (As)

Arsenic (As) was analysed in cod liver at 13 stations and in blue mussel at 32 stations.

Levels

Relevant values for background levels of As are not available for cod. The highest concentration was found in cod liver from the Inner Oslofjord (st. 30B, 11 mg As/kg w.w.) and the lowest value was found in Helgeland (st. 96B, 2 mg As/kg w.w.).

There were no changes in classes for As in blue mussel from 2013 to 2014, and all mussels were insignificantly polluted (Class I).

Upward trends

There were both significant upward long-term and short-term trends in the cod liver from Lofoten (st. 98B1).

Downward trends

There were both significant downward long-term and short-term trends in the cod liver from the Inner Oslofjord (st. 30B), Kristiansand harbour (st. 13B), Bømlo (st. 23B) and Tromsø harbour (st. 43B2).

In blue mussel, there were both significant downward long-term and short-term trends at Bjørkøya (st. 71A) in the Grenlandsfjord, Risøy (st. 76A2), Gåsøy (st. 15A) close to Mandal, and Skallneset (st. 10A2) in the Varangerfjord.

Other studies

Blue mussel at Frognerkilen, the mouth of Alna and Bekkelaget were at background levels for As in 2014 (Ruus *et al*. 2015) and 2013 (Ruus *et al*. 2014). Blue mussel in the Inner Oslofjord was up to moderately polluted with As from 2006 to 2013 (Berge 2014). Mussel was also up to moderately polluted with As at Langøya in the Holmestrandfjord in 2014 (Gitmark *et al*. 2015). Most blue mussel stations in the Kristiansandsfjord were moderately polluted by As (Schøyen *et al*. 2014).

General, large scale trends

Discharges of As to water from land-based industries showed an increase from 2008 (516 kg As/year) to 2010 (2587 kg As/year) and from 2013 (1504 kg As/year) to 2014 (1883 kg As/year) (***Figure 17***). No explanation was provided at this site (i.e. www.norskeutslipp.no) to explain this large increase. Emission to air had gradually decreased from 2002 (1240 kg As/year) to 2014 (537 kg As/year).

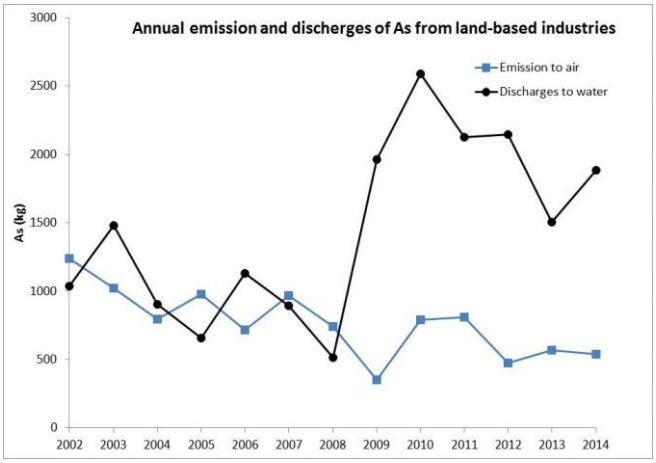

Figure 17. *Annual emissions of As to air and discharges to water from land-based industries in the period 2002-2014 (data from www.norskeutslipp.no). Note that this category excludes emissions and discharges from municipal treatment plants, land runoff, transportation and offshore industry.*

Upper average of total riverine inputs of As in Norway has been in the range of 26.76 tonnes in 2013 (Skarbøvik *et al*. 2014). Upper average of total riverine inputs of As were 11.47 tonnes to Skagerrak, 4.90 tonnes to the North sea, 5.48 tonnes to the Norwegian Sea and 4.91 tonnes to the Barents sea (Skarbøvik *et al*. 2014), indicating higher input in the southern part of Norway. In addition to riverine inputs, comes the contribution by direct discharges from sewage and industrial effluents amounting to 1.67 tonnes or about 6 % of the total (28 tonnes) (Skarbøvik *et al*. 2014).

3.2.8 Nickel (Ni)

Nickel (Ni) was analysed in cod liver at 13 stations and in blue mussel at 32 stations.

Levels

The national environmental classifications system does not include Ni in cod. The highest concentration was found in cod liver from the Inner Trondheimsfjord (st. 80B, 1.3 mg Ni/kg w.w.). At the two stations Bømlo (st. 23B) and Varangerfjord (st. 10B), data on cod liver was inadequate to perform trend analysis due to concerns about the limit of detections.

There were no changes in classes from 2013 to 2014 for Ni in blue mussel, and only background levels (Class I) were observed.

Upward trends

No upward trends were found.

Downward trends

Cod from Kristiansand harbour (st. 13B) and blue mussel from Moholmen (st. I965) in the Ranfjord had both significant downward long-term and short-term trends.

Other studies

All blue mussel stations in the Inner and Outer Oslofjord showed acceptable (background) levels of Ni. Blue mussel at Frognerkilen, the mouth of Alna and Bekkelaget were moderately polluted (Class II) for Ni in 2014 (Ruus *et al*. 2015) and at background levels (Class I) in 2013 (Ruus *et al*. 2014). Other investigations found that mussel was at background levels (Class I) for Ni at Langøya in the

Holmestrandfjord in 2014 (Gitmark *et al*. 2015). Blue mussel was insignificantly polluted by Ni in the Kristiansandsfjord in 2013 (Schøyen *et al*. 2014).

General, large scale
Discharges of Ni to water from land-based industries had decreased gradually from 2002 (15955 kg Ni/year) to (6990 kg Ni/year) 2014 (***Figure 18***).

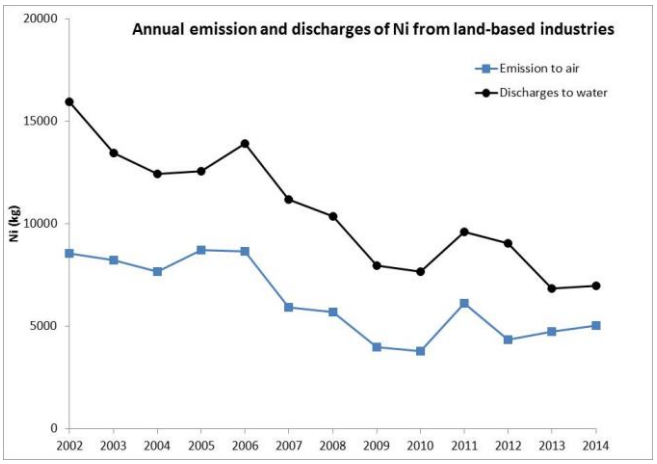

Figure 18. *Annual emissions of Ni to air and discharges to water from land-based industries in the period 2002-2014 (data from www.norskeutslipp.no). Note that this category excludes emissions and discharges from municipal treatment plants, land runoff, transportation and offshore industry.*

Upper average of total riverine inputs of Ni in Norway was 132.86 tonnes in 2013 (Skarbøvik *et al*. 2014). Upper average of total riverine inputs of Ni were 40.22 tonnes to Skagerrak, 19.19 tonnes to the North sea, 37.21 tonnes to the Norwegian Sea and 36.23 tonnes to the Barents sea (Skarbøvik *et al*. 2014). In addition to riverine inputs, comes the contribution by direct discharges from sewage and industrial effluents amounting to 8.63 tonnes or about 6 % of the total (141 tonnes) (Skarbøvik *et al*. 2014).

3.2.9 Chromium (Cr)

Chromium (Cr) was analysed in cod liver at 13 stations and in blue mussel at 32 stations.

Levels
Relevant values for background levels of Cr are not available for cod. The highest concentration in cod liver was found in cod liver from Bømlo (st. 23B, 0.3 mg Cr/kg w.w.).

There were no changes in classes from 2012 to 2013 for Cr in blue mussel. All mussels were insignificantly polluted (Class I) with Cr.

Upward trends
No upward trends were found.

Downward trends

Significant downward long-term and short-term trends were found in cod liver from Kristiansand harbour (st. 13B) and in blue mussel from Croftholmen (st. I712) in the Grenlandsfjord and in Moholmen (st. I965) in the Ranfjord.

Other studies

Blue mussel at Frognerkilen, the mouth of Alna and Bekkelaget were markedly to severely polluted (Class III-IV) for Cr in 2014 (Ruus *et al*. 2015) and at background levels (Class I) in 2013 (Ruus *et al*. 2014). Blue mussel from the Inner Oslofjord was insignificantly polluted with Cr in 2006 to 2013 (Berge 2014). Mussel was up to moderately polluted (Class II) with Cr at some stations at Langøya in the Holmestrandfjord in 2014 (Gitmark *et al*. 2015). There are limited data for other recent surveys and data for 2014 does not occur. Blue mussel at all seven stations in the Kristiansandsfjord had background levels of Cr in 2013 (Schøyen *et al*. 2014).

General, large scale trends

Emissions of Cr to air and discharges to water from land-based industries are shown in ***Figure 19***.

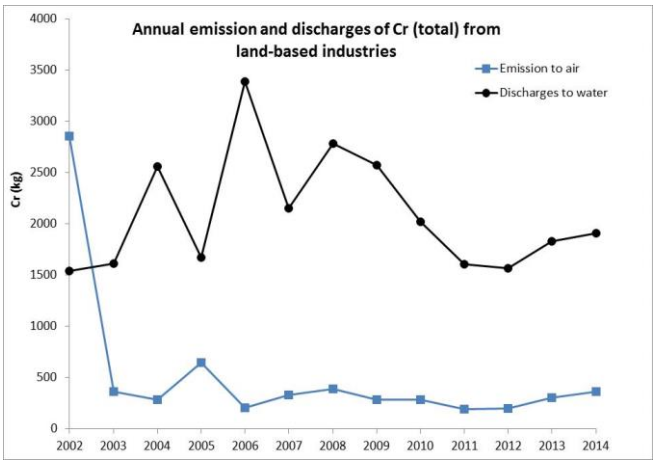

Figure 19. *Annual emissions of Cr to air and discharges to water from land-based industries in the period 2002-2014 (data from www.norskeutslipp.no). Note that this category excludes emissions and discharges from municipal treatment plants, land runoff, transportation and offshore industry.*

Total riverine inputs of Cr in Norway have been in the range of 74.21 to 77.98 tonnes in 2013 (Skarbøvik *et al*. 2014). The range of total riverine inputs were 20.53 to 22.20 tonnes Cr to Skagerrak, 15.26 to 16.68 tonnes Cr to the North sea, 32.36 to 32.91 tonnes Cr to the Norwegian Sea and 6.06 to 6.18 tonnes Cr to the Barents sea (Skarbøvik *et al*. 2014). In addition to riverine inputs, comes the contribution by direct discharges from industrial effluents amounting to 1.82 tonnes or about 2 % of the total (80 tonnes) (Skarbøvik *et al*. 2014).

3.2.10 Cobalt (Co)

Cobalt (Co) was analysed in cod liver at 13 stations and in blue mussel at 32 stations.

Levels

There is no national classification for Co in blue mussel or cod.

Upward trends

Both significant upward long-term and short-term trends were observed in cod liver at Lofoten (st. 98B1), as in 2013.

Downward trends

Both significant downward long-term and short-term trends were observed in blue mussel at Risøy (st. 76A2) and Lastad (st. I131A).

General, large scale trends

Discharges of Co to water from land-based industries showed decreasing values from 2011 (754 kg Co/year) to 2014 (468 kg Co/year) (***Figure 20***).

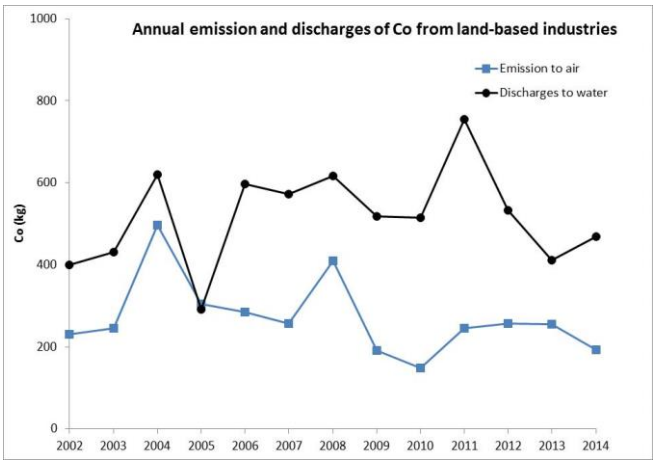

Figure 20. *Annual emissions of Co to air and discharges to water from land-based industries in the period 2002-2014 (data from www.norskeutslipp.no). Note that this category excludes emissions and discharges from municipal treatment plants, land runoff, transportation and offshore industry.*

3.2.11 Tributyltin (TBT)

Tributyltin (TBT) was analysed in blue mussel at eight stations, in dog whelk (*Nucella lapillus*) at eight stations and in common periwinkle (*Littorina littorea*) at one station. Imposex (VDSI) was investigated in *Nucella lapillus* at all eight stations.

Levels and trends

There were no changes in trends from 2013 to 2014. At Gåsøy (st. 15G), Lastad (st. 131G), Svolvær (st. 98G) and Brashavn (st. 11G), TBT-data was inadequate for trend analysis due to concerns about the limit of detections.

Concentrations of TBT in dog whelk (Nucella lapillus)

There is no national classification for TBT-concentrations in dog whelk. At Færder (st. 36G), Risøy (st. 76G), Melandsholmen (st. 227G1) and Espevær (st. 22G), significant downward long- and short-term TBT-trends were observed in 2014, as in 2013. The highest organotin level was found at Melandsholmen close to Haugesund (6.09 µg/kg w.w.) on the west coast of Norway, and the lowest values was observed at Gåsøy (<0.826 µg/kg w.w.).

Concentrations of TBT in common periwinkle (Littorina littorea)

There were no significant trends of TBT at Fugløyskjær in the Grenland area in 2014, as in 2013. The TBT-concentration was 1.76 µg/kg w.w.

Biological effects of TBT (imposex/VDSI) in dog whelk

The effects from TBT were low (VDSI<0.448) at all stations investigated in 2014. There were significant downward trends at all stations, except for at Brashavn where no trends were observed. It can be noted that VDSI-values at this location have been low during the whole monitoring period since 2002. No effects (VDSI=0) were found at Færder, Risøy, Lastad and Gåsøy. These results, including Espevær (VDSI=0.08) and Lofoten (VDSI=0.034) were below the OSPARs Background Assessment Criteria (BAC=0.3, OSPAR 2009). VDSI at Lofoten had decreased to 0.034 in 2014 from 0.464 in 2013. The VDSI was 0.448 at Melandsholmen. These results were over BAC but below the OSPARs Ecotoxicological Assessment Criteria (EAC=2, OSPAR 2013).

General, large scale trends

The results show that the Norwegian legislation banning application of organotins on ships shorter than 25 meters in 1990 and longer than 25 meters in 2003/2008, has been effective in reducing imposex in dog whelk populations. Some of the previously effected snail populations have also re-established. The international convention that was initiated by the International Maritime Organization (IMO) did not only ban application of organotins on ships after 2003 but also stated that organotins after 2008 could not be part of the system for preventing fouling on ships. VDSI in dog whelk was around level 4 in all dog whelk stations before the ban in 2003, except for the Varangerfjord where the VDSI had been low in the whole monitoring period. It was a clear decline in VDSI as well as TBT at nearly all stations between 2003 and the total ban in 2008 (***Figure 21*** and ***Figure 22***). The exceptions being for VDSI for snails from Varangerfjord where the VDSI has remained low (<0.3) for the entire investigation period. After 2008, the VDSI has been close to zero at many of the stations. A typical example of decreasing trends is shown for Færder in ***Figure 23***.

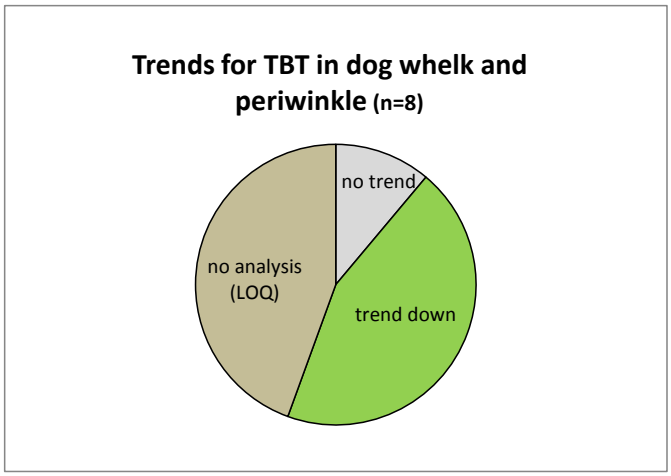

Figure 21. *Frequency of trends for TBT in dog whelk and periwinkle (1991-2014). No upward trends were detected. Concerns about LOQ prevented some trend analyses.*

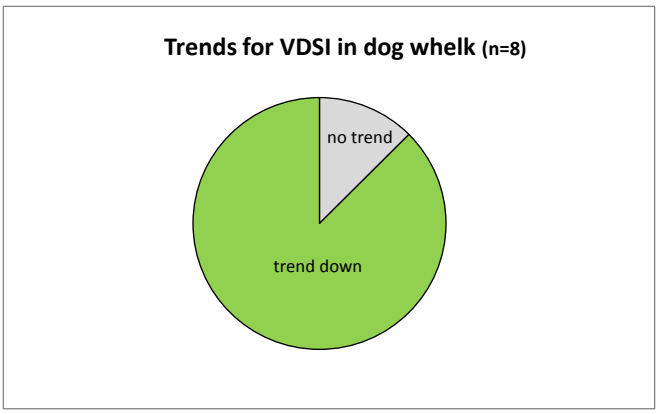

Figure 22. *Frequency of trends for VDSI in dog whelk (1991-2014). No upward trends were detected.*

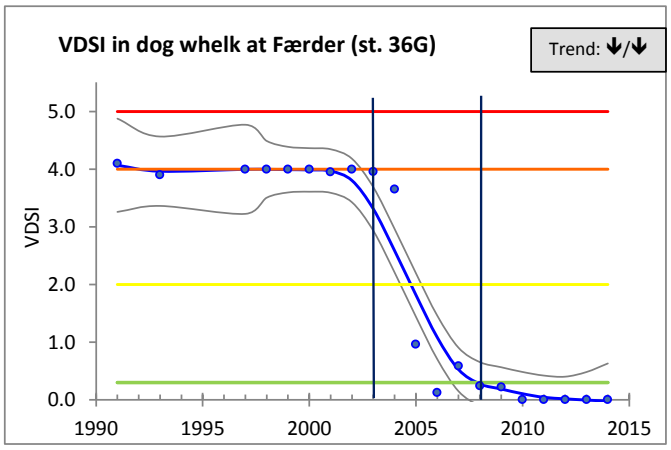

Figure 23. *Change in VDSI for dog whelk from Færder (st. 36G). The vertical black lines indicate the initial ban of TBT in 2003 and total ban in 2008. The horizontal lines indicate OSPAR classes (see* ***Table 35*** *in* Appendix C*). The green line indicates OSPAR Background Assessment Criteria (BAC = 0.3) and the yellow line indicates the OSPAR Ecotoxicological Assessment Criteria (EAC = 2).*

Discharges of tributyltin and trifenyltin to water from land-based industries from 2002 to 2014 (***Figure 24***), but they do not adequately reflect loads to the marine environment in that this does not include discharges from shipping for this period. The values were high in 2003 (487 g tributyltin and trifenyltin/year) and 2009 (504 g tributyltin and trifenyltin/year).

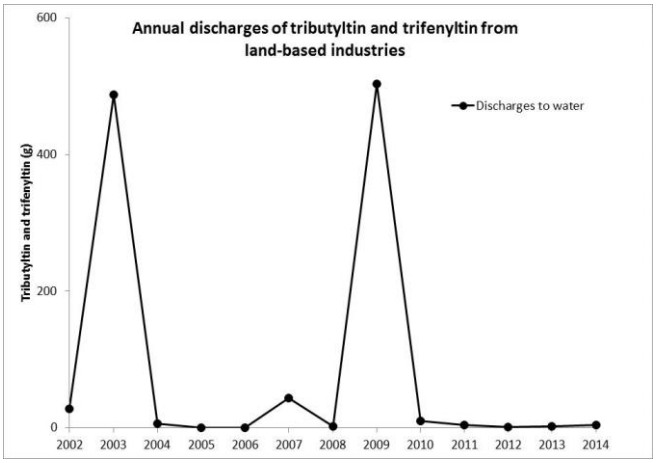

Figure 24. *Annual discharges of tributyltin and trifenyltin to water from land-based industries in the period 2002-2014 (data from www.norskeutslipp.no). Note that this category excludes discharges from municipal treatment plants, land runoff, transportation and offshore industry.*

3.2.12 Polychlorinated biphenyls (PCB-7)

Polychlorinated biphenyls (defined here as PCB-7, see ***Table 4***) are a group of chloriniated organic compounds that previously had a broad industrial and commercial applicaton. PCB-7 was analysed in cod liver at 13 stations and in blue mussel at 29 stations.

Important levels exceeding Class I

Cod liver from the Inner Oslofjord (st. 30B) (***Figure 25***) was markedly polluted (Class III) with PCB-7, while cod liver from Kristiansand harbour (st. 13B) and the Inner Sørfjord (st. 53B) were moderately polluted (Class II).

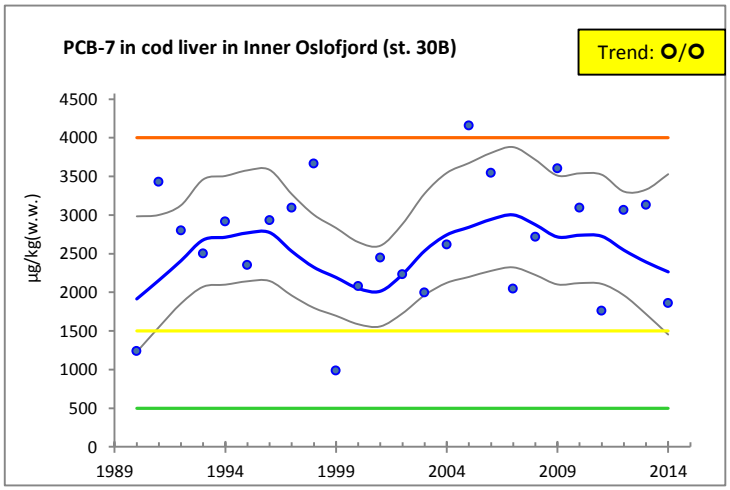

Figure 25. *Median concentrations (mg/kg w.w.) of PCB-7 in cod liver from 1990 to 2014 in the Inner Oslofjord (st. 30B).*

Mussels from Akershuskaia (st. I301) in the Inner Oslofjord was still moderately polluted (Class II) in 2014, as in 2013. Mussels at Nordnes (st. I241) and Outer Trondheimsfjord (st. 91A2) were also moderately polluted (Class II).

Class increased since 2013

Blue mussel in the Outer Trondheimsfjord (st. 91A2) was moderately polluted (Class II) in 2014 but was insignificantly polluted (Class I) in 2013.

Upward trends

No upward trends for PCB-7 were found in cod liver. There was upward short-term trend for PCB-7 in blue mussel at Odderøy (st. I133A).

Class decreased since 2013

Cod liver from Kristiansand harbour (st. 13B) was markedly polluted (Class III) with PCBs in 2013 and moderately polluted (Class II) in 2014, as for the period 2009 to 2012.

PCB-concentrations in blue mussel at Gressholmen (st. 30A) and Gåsøy (st. 15A) were at background levels (Class I) in 2014, but were moderately polluted (Class II) in 2013.

Downward trends/low levels

There were both significant downward long-term and short-term trends for PCB-7 in cod liver from the Inner Trondheimsfjord (st. 80B), Lofoten (st. 98B1), Tromsø harbour (st. 43B2) and in the Varangerfjord (st. 10B), and for blue mussel at Gåsøya (st. I304) and Ramtonholmen (st.I307) in the Inner Oslofjord.

A significant downward long-term trend was observed in cod at Bømlo (st. 23B) and a short-time trend were found at Færder (st. 36B). In addition, there were significant downward long-term trends (and no short-term trends) for PCB-7 in blue mussel at 14 stations; five from the Oslofjord area (Akershuskaia (st. I301), Håøya (st. I301), Singlekalven (st. I023), Kirkøy (st. I024), Mølen (st. 35A)) and five from the Sørfjord/Hardanger region (Eitrheimsneset (st. 52A), Kvalnes (st. 56A), Krossanes (st. 57A), Ranaskjær (st. 63A), Vikingneset (st. 65A)). This was also the case at Risøy (st. 76A) close to Risør, Espevær (st. 22A) on the west coast, and Lofoten (st. 98A2) and Skallneset (st. 10A2) in the northern part of Norway. All were at background concentrations except for moderate levels (Class II) at Akershuskaia (st. I301) in the Inner Oslofjord.

Inner Oslofjord

Cod liver caught at 100 m depth in the Inner Oslofjord (st. 30B) was markedly polluted while blue mussel from Akershuskaia (st. I301) was moderately polluted with ΣPCB-7. Mussel at other stations in the Oslofjord like Gressholmen, Gåsøya, Ramtonholmen, Håøya, Solbergstrand, Mølen and Tjøme were insignificantly polluted with ΣPCB-7.

Other studies

The high concentrations of ΣPCB-7 in cod liver from the Inner Oslofjord during the last years have been confirmed in another study which showed that cod liver from Bekkelaget and Frognerkilen was markedly to severely polluted (Class III-IV) with PCBs in 2006 to 2013 (Berge 2014). A certain decrease in concentration of PCBs in cod from Bekkelagsbassenget based on wet weight could be observed, but the decrease was not significant and not evident when results were normalised to lipid content. Monitoring of blue mussel in the Inner Oslofjord showed that mussels were up to markedly polluted with ΣPCB-7 in the period 2006 to 2013 (Berge 2014). A study of flounder liver from the Inner Oslofjord in 2013 showed apparently lower (a factor of~7) median concentration of ΣPCB-7 than in cod in 2012 (Ruus *et al*. 2014). Blue mussel at all seven stations in the Kristiansandsfjord was insignificantly polluted with PCB-7 in the period 2010 to 2013.

Historical data on entry of PCBs to the Inner Oslofjord is not available. Present entry of PCBs to the fjord has however been calculated to be around 3.3 kg/year (Berge *et al*. 2013b). Run-off from urban surfaces is the most important contributor (2.1 kg/year). It is also anticipated that sediments in the fjord store much of the historic inputs of PCB, but their role as a current source of PCBs for uptake in biota is unclear. Parts of the Inner Oslofjord are densely populated with much urban activities. The high concentrations of PCBs observed in cod liver are probably related to these activities both in past and present, as well as reduced water exchange with the Outer fjord.

PCB-concentrations in trout from Mjøsa have been relatively stable since 2000, and no trend was detected (Løvik *et al*. 2015).

General, large scale trends

On a national level, the results show that the concentrations of PCBs in general have decreased in both cod and blue mussel over the whole monitoring period and no significant upward trends for PCBs in mussels and cod were observed except for a upward short-term trend for PCB-7 in blue mussel at Odderøy (st. I133A).

In Norway PCBs has been prohibited since 1980, but leakage from old products as well as landfills and natural deposits may still be a source of contamination. Production and new use of PCBs is also prohibited internationally through the ECE-POPs protocol and the Stockholm Convention.

Emissions of PCBs to air and discharges to water from land-based industries are shown in (***Figure 26***). Before 2009 occasional high emissions and discharges were reported, but throughout 2009-2013 the levels have been low. Investigations by Schuster *et al.* (2010) indicate that emissions in the northern Europe have declined during the period 1994-2008 by about 50 %.

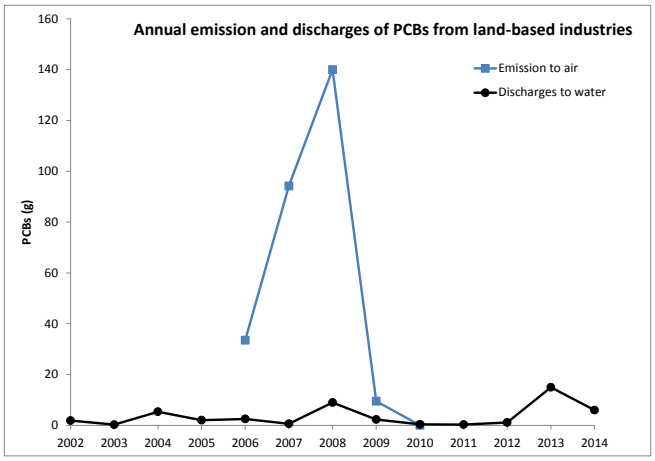

Figure 26. *Annual emissions of PCBs to air and discharges to water from land-based industries in the period 2002-2014 (data from* www.norskeutslipp.no*). No data for emissions to air are reported for 2002-2005 and 2011-2014. No data for discharges to water are reported for 2010-2011. Note that this category excludes emissions and discharges from municipal treatment plants, land runoff, transportation and offshore industry.*

3.2.13 Dichlorodiphenyldichloroethylene (ppDDE)

Dichlorodiphenyldichloroethylene (ppDDE) was analysed in cod liver at seven stations and in blue mussel at 19 stations.

Important levels exceeding Class I

Blue mussel at Kvalnes (st. 56A) in the Mid Sørfjord was extremely polluted (Class V) in 2014, 2013 and 2012, but the concentration had decreased from 51 µg/kg w.w. in 2013 to 30 µg/kg w.w. in 2014 (***Figure 27***).

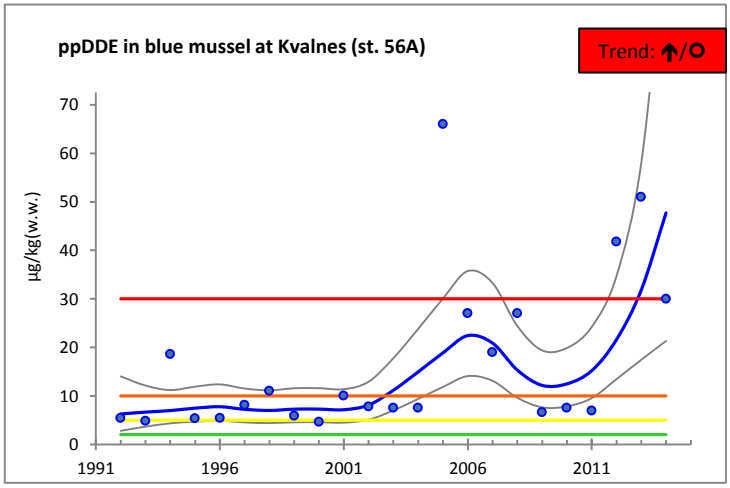

Figure 27. *Median concentrations (mg/kg w.w.) of ppDDE in blue mussel from 1992 to 2014 in the Mid Sørfjord at Kvalnes (st. 56A).*

Cod liver from the Inner Sørfjord (st. 53B) and blue mussel from Krossanes (st. 57A) and Utne (st. 64A) in the Outer Sørfjord were moderately polluted (Class II) with ppDDE.

Upward trends

A significant upward long-term trend was found in blue mussel at Kvalnes (st. 56A) in the Mid Sørfjord in 2014, as in 2013.

Class decreased since 2013

Mussels at Utne (st. 64A) in the Outer Sørfjord were severely polluted (Class IV) with ppDDE in 2013 while they were moderately polluted (Class II) in 2014. Cod liver in the Inner Sørfjord (st. 53B) was markedly polluted (Class III) in 2013, but moderately polluted (Class II) in 2014. Cod liver from the Inner Oslofjord (st. 30B) and blue mussel from Eitrheimsneset (st. 52A) in the Inner Sørfjord were both moderately polluted (Class II) in 2013, while they were at background level (Class I) in 2014.

Downward trends/low levels

Both significant downward long-term and short-term trends for ppDDE were found in cod liver from Farsund (st. 15B), Lofoten (st. 98B1) and in the Varangerfjord (st. 10B). Significant downward long-term trends were observed in cod liver from the Inner Oslofjord (st. 30B), Færder (st. 36B) and Bømlo (st. 23B) on the west coast. There were significant downward long-term trends in blue mussel at Gressholmen (st. 30A) in the Inner Oslofjord and at Bjørkøya (st. 71A).

At Skallneset (st. 10A2) and Brashavn (st. 11X) in the Varangerfjord, data was inadequate for long-term trend analysis due to concerns about the limit of detections in 2014.

Inner Oslofjord

Liver from Bekkelaget and Frognerkilen in the Inner Oslofjord had low levels of DDT in 2006, 2009 and 2010, and background levels (Class I) were observed in 2013 (Berge 2014). Monitoring in the Inner Oslofjord showed that blue mussel was up to moderately polluted (Class II) with ΣDDE+DDD in 2013 (Berge 2014).

The Sørfjord

The Sørfjord area has a considerable number of orchards. Earlier use and the persistence of DDT and leaching from contaminated soil is probably the main reason for the observed high concentrations of ppDDE in the Sørfjord area. It must however be noted that the use of DDT products have been prohibited in Norway since 1970. Green *et al*. (2004) concluded that the source of ppDDE in the Sørfjord was uncertain. Analyses of supplementary stations between Kvalnes and Krossanes in 1999 indicated that there could be local sources at several locations (Green *et al*. 2001).

A more intensive investigation in 2002 with seven sampling stations confirmed that there were two main areas with high concentrations north of Kvalnes and near Urdheim south of Krossanes (Green *et al*. 2004). Skei *et al*. (2005) concluded that the variations in concentrations of ΣDDT and the ratio between p,p'-DDT/p,p'DDE (insecticide vs. metabolite) in blue mussel from Byrkjenes and Krossanes corresponds with periods with much precipitation and is most likely a result of wash-out from sources on shore. Botnen & Johansen (2006) deployed passive samplers (SPMD- and PCC-18 samplers) at 12 locations along the Sørfjord to sample for DDT and its derivates in sea water. Blue mussel and sediments were also taken at some stations. The results indicated that further and more detailed surveys should be undertaken along the west side of the Sørfjord between Måge and Jåstad, and that replanting of old orchards might release DDT through erosion. Concentrations of ΣDDT in blue mussel in the Sørfjord in 2008-2011 showed up to Class V (extremely polluted) at Utne (Ruus *et al*. 2009, 2010a, 2011, 2012). There was high variability in the concentrations of ΣDDT in replicate samples from Utne, indicating that the station is affected by DDT-compounds in varying degree, dependent on local conditions. The highest concentrations of ppDDE in sediment were observed in Mid Sørfjord (Green *et al*. 2010b).

Increased ΣDDT-concentrations in blue mussel from the Sørfjord were discussed by Ruus *et al*. (2010b). Possible explanations were increased transport and wash-out to the fjord of DDT sorbed to dissolved humus substances.

General, large scale trends

DDT is banned in all countries in Europe, USA and Canada. In Norway, the use of DDT was restricted in 1969 and the last approved use of DDT was discontinued in 1988. However, DDT from landfills and orchards can still be a problem.

3.2.14 Polycyclic aromatic hydrocarbons (PAHs)

Polycyclic aromatic hydrocarbons (PAHs)[9] was analysed in blue mussel at 11 stations. The main sources of PAH in coastal waters include discharges from smelting industry.

Important levels exceeding Class I

All 11 blue mussel stations except Moholmen in the Ranfjord (st. I965), had concentrations of PAHs at background levels. Mussels from Moholmen were moderately polluted (Class II).

Class decreased since 2013

Blue mussel at Moholmen (st. I965) was markedly polluted (Class III) with PAHs in 2013, and moderately polluted (Class II) in 2014. Mussels at Akershuskaia (st. I301) in the Inner Oslofjord and Bjørnebærviken (st. I969) in the Ranfjord were both moderately polluted (Class II) in 2013, while they were insignificantly polluted (Class I) in 2014.

Trends

No significant trends were observed.

Other studies

Monitoring of blue mussel in another study in the Inner Oslofjord showed that mussels were up to markedly polluted with PAH-16 at Rådhuskaia/Pipervika in 2013 (Berge 2014). Mussels at all other stations were up to moderately polluted in the period from 2006 to 2013 (Berge 2014). Another investigation documented that mussels were up to moderately polluted with PAHs at Langøya in the Holmestrandfjord in 2013 (Gitmark *et al*. 2014). Blue mussel at two stations in Kristiansandsfjord was moderately polluted with PAHs in 2013 (Schøyen *et al*. 2014). Remedial action has been implemented to reduce the impact of PAHs in the Kristiansandsfjord. The Ranfjord has received discharges of PAHs from local industry for a number of years. No trends were detected for PAHs in blue mussel in the Ranfjord for the period 1995 (Bjørnbærviken) or 2001 (Moholmen) to 2013.

General, large scale trends

Emissions of PAHs to air and discharges to water from land-based industries from 2012 to 2014 can be seen in ***Figure 28***. The emission to air has decreased gradually from 2005 (178 013 kg PAHs/year) to 2014 (34 816 kg PAHs/year).

[9] For this report the total is the sum of tri- to hexacyclic PAH compounds named in EPA protocol 8310 minus naphthalene (dicyclic)-totalling 15 compounds, so that the classification system of the Norwegian Environment Agency can be applied (see Appendix B).

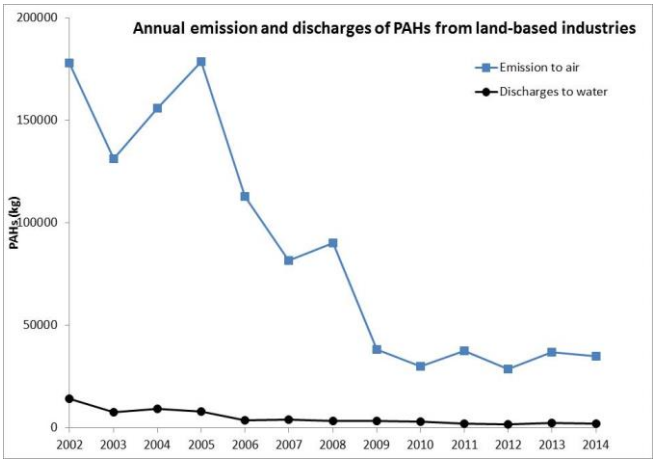

Figure 28. *Annual emissions of PAHs to air and discharges to water from land-based industries in the period 2002-2014 (data from* www.norskeutslipp.no*). Note that this category excludes emissions and discharges from municipal treatment plants, land runoff, transportation and offshore industry.*

Environmental Quality Standards (EQS)

The EQS (2013/39/EC) for fluoranthene (30 µg/kg w.w.) in biota for "molluscs" was exceeded at Akershuskaia (st. I301) (41.43 µg/kg w.w.) in the Inner Oslofjord, at Mølen (st. 35A) (36.2 µg/kg w.w.) in the Mid Oslofjord, and at Moholmen (st. I965) (61.05 µg/kg w.w.) in the Ranfjord.

3.2.15 Sum carcinogenic polycyclic aromatic hydrocarbons (KPAHs)

Sum carcinogenic polycyclic aromatic hydrocarbons (KPAHs) was analysed in blue mussel at 11 stations.

Important levels exceeding Class I

The concentrations of the potentially most carcinogenic PAHs (KPAHs, cf. Appendix B) in blue mussel exceeded Class I (insignificantly polluted) from two of 11 stations. The two stations were Mølen (st. 35A) in the Mid Oslofjord and Moholmen (st. I965) in the Ranfjord. Mussels from both stations were moderately polluted (Class II). Mussels at other stations were at background levels (Class I).

Class decreased since 2013

Blue mussel from Moholmen (st. I301) was markedly polluted (Class III) in 2013, but moderately polluted (Class II) in 2014. Mussels from Bjørnebærviken (st. I969) in the Ranfjord were moderately polluted (Class II) in 2013, while they were at background concentrations (Class I) in 2014.

Trends

A significant downward short-term trend was observed in blue mussel from Bjørnebærviken (st. I969).

Other studies

Blue mussel from the Inner Oslofjord was found to be severely polluted with KPAH at Rådhuskaia/Pipervika in 2013, and mussels from all other stations were up to moderately polluted in the period 2006 to 2013 (Berge 2014). Mussels from Langøya in the Holmestrandfjord in 2013 were up to markedly polluted with KPAH (Gitmark *et al.* 2014). Blue mussel at Odderøy and Svensholmen

in the Kristiansandsfjord were markedly polluted with KPAH in 2013, as in 2012 (Schøyen *et al.* 2014).

3.2.16 Benzo[a]pyrene (B[a]P)

Benzo[a]pyrene (B[a]P) was analysed in blue mussel at 11 stations.

Important levels exceeding Class I

The highest concentration (3.6 µg/kg w.w.) was found at Mølen (st. 35A) where the mussels were markedly polluted (Class III) with B[a]P. The mussels were moderately polluted (Class II) with B[a]P at Gåsøya (st. I304) in the Inner Oslofjord, at Singlekalven (st. I023) in the Hvaler area, and at Moholmen (st. I965) in the Ranfjord where. Other mussels were at background levels (Class I).

Class increased since 2013

Blue mussel at Gåsøya (st. I304) in the Inner Oslofjord and at Singlekalven (st. I023) in the Hvaler area were at background levels (Class I) in 2013, while they were moderately polluted (Class II) in 2014.

Class decreased since 2013

The concentration of B[a]P in blue mussel from Moholmen (st. I965) in the Ranfjord had decreased from being markedly polluted (Class III) in 2013 to moderately polluted (Class II) in 2014. Mussel at Bjørnebærviken (st. I969) was moderately polluted (Class II) in 2013, while they was at background level (Class I) in 2014.

Trends

No trends were observed. At five stations, data was inadequate for trend analysis.

Other studies

A previous study of blue mussel in the Inner Oslofjord showed that mussels were severely polluted with B[a]P at Rådhuskaia/Pipervika in 2013 (Berge 2014).

High concentrations in the Ranfjord are most likely related to diffuse influence from activities related to local harbours and industry.

Environmental Quality Standards (EQS)

The EQS (2013/39/EC) for B[a]P is 5 µg/kg w.w. in biota for “fish”. Applying this EQS for blue mussel, all concentrations of B[a]P were below the EQS applied for biota.

3.2.17 Polybrominated diphenyl ethers (PBDEs)

Polybrominated diphenyl ethers (PBDEs) are a group of brominated flame retardants used in a variety of products. PBDEs were in 2014 analysed in cod liver at eight stations and in blue mussel at 10 stations.

Levels of cod liver

The tetrabromodiphenyl ether BDE47 was the dominant congener in cod liver and the concentration was highest in the Inner Oslofjord (st. 30B, 21 µg/kg w.w.) (***Figure 29***). The lowest BDE47-concentration in liver was found in cod from Lofoten (st. 98B1, 1.1 µg/kg w.w.).

Trends

At eight cod stations, data for BDE153, -183 and -196 was inadequate for trend analysis due to concerns about the limit of detections in 2014. This was also the case regarding BDE99 at five stations and for BDE209 at six stations. At the two blue mussel stations Gressholmen (st. 30A) in the Inner Oslofjord and Bjørkøya (st. 71A) in the Grenlandsfjord area, data for BDE153, -154, and -183 was inadequate for trend analysis due to concerns about the limit of detections in 2014. This was also the case regarding BDE209 at Gressholmen (st. 30A) and Bodø harbour (st. 97A2).

A significant upward short-term trend was observed for BDE154 in cod liver from Bømlo (st. 23B). No upward trends were observed for the other congeners.

Significant downward long-term and short-term trends where observed for BDE47, -99, -100 and -154 in cod liver caught in Tromsø harbour (st. 43B2) in 2014. This was also the case for BDE100 in cod liver from the Inner Oslofjord (st. 30B).

Significant downward long-term trends of BDE47 were found in cod liver at Færder (st. 36B), Bømlo (st. 23B) and Lofoten (st. 98B1). Significant downward long-term trends for BDE28 and -100 were also observed in cod liver at Bømlo (st. 23B).

A significant downward short-term trend for cod liver from the Inner Oslofjord (st. 30B) was found for BDE47.

Significant downward long-term and short-term trends where observed for BDE47 in blue mussel from Gressholmen (st. 30A) in the Inner Oslofjord.

The standard deviation varied considerably among stations, also for other PBDEs. The highest deviations were found in the Inner Trondheimsfjord (st. 80B) for BDE47 and -100 (***Table 11***). In 2013, the highest deviations were found at Ålesund (st. 28B) for BDE47, -100 and -154. In 2014, cod from this site were missing. It seems like the deviations were highest in affected areas.

In the urban areas like Oslo and Trondheim harbour, some of the BDE-congeners in cod liver had significantly higher levels than in remote areas like Færder and Bømlo (Tukey-Kramer HSD test).

PBDEs have been investigated annually in cod liver since 2005. In the Inner Oslofjord (st. 30B), cod have also been analysed for PBDEs in 1993, 1996 and 2001 (***Figure 30***). Samples for similar analyses were also collected from the Færder area (st. 36B) in 1993 and 1996, and from Bømlo (st. 23B) on the west coast in 1996 and 2001. In 2014, PBDEs were analysed in cod from eight stations (***Table 11***). Of the PBDEs, only congeners BDE28, -47, -100 and -154 were over the detection limit in at least half of the samples from each station.

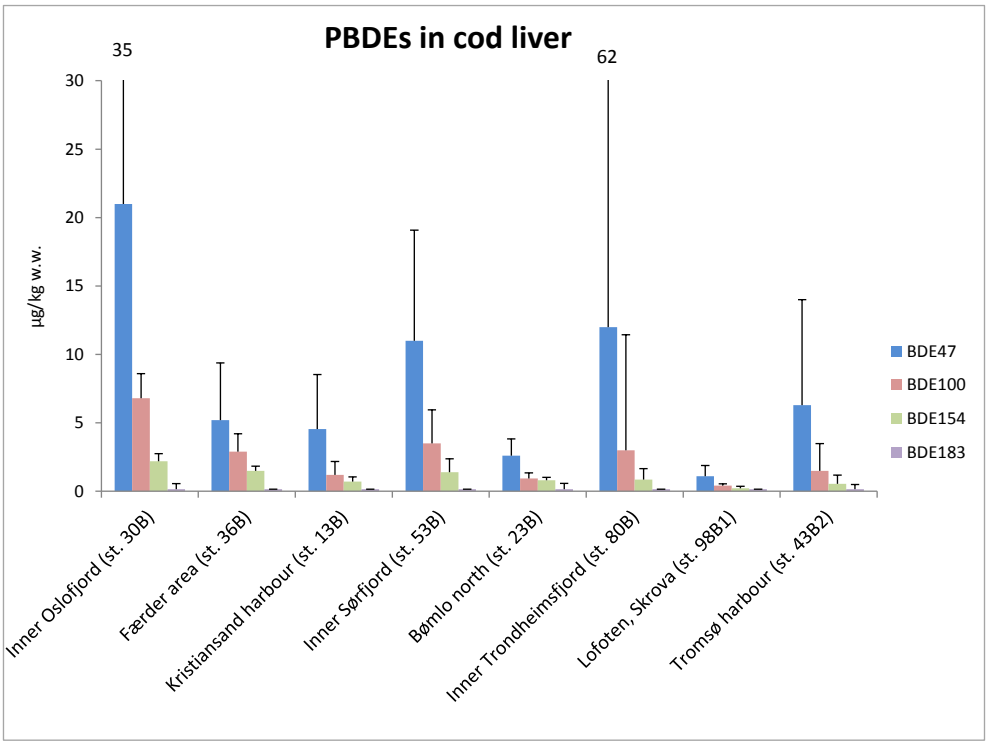

***Figure 29**. Median concentrations (µg/kg w.w.) of PBDEs in cod liver in 2014. Only the results are shown where concentrations were above the detection limit for half or more of the samples. The error bar indicates one standard deviation above the median.*

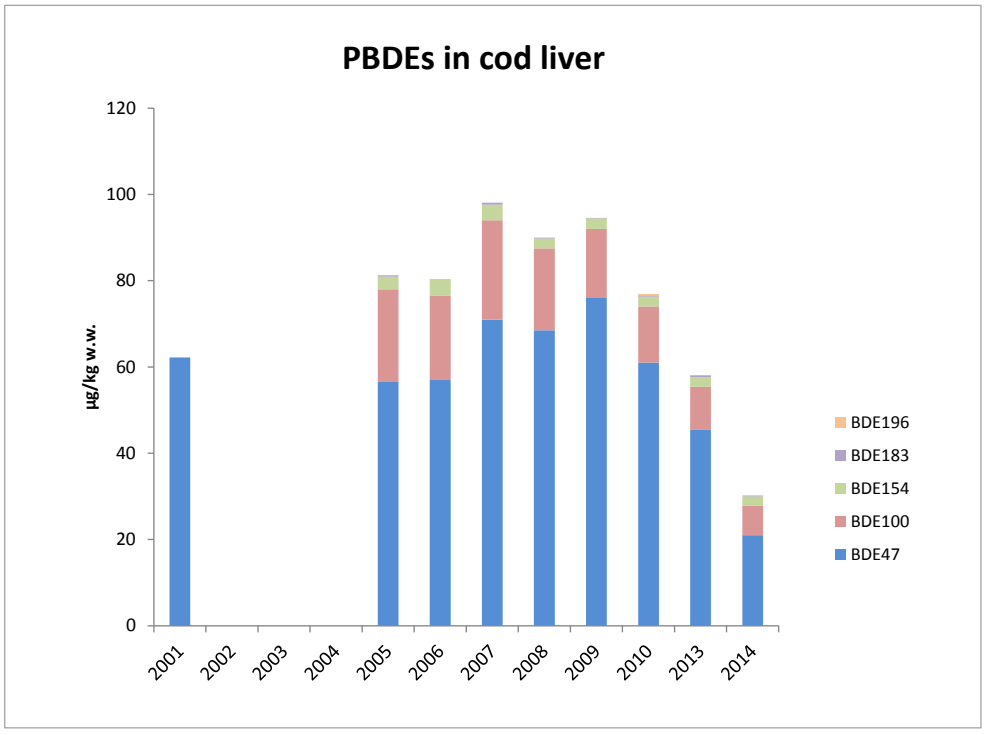

***Figure 30**. Median concentrations (µg/kg w.w.) of PBDEs in cod liver from 2001 to 2014 in the Inner Oslofjord (st. 30B).*

***Table 11**. Median concentrations (µg/kg w.w.) and standard deviations for PBDE congeners in blue mussel and cod liver in 2014. BDESS indicates sum of all BDEs. Count indicates number of samples analysed. The first number within the parentheses indicates the number of pooled samples included. The second number within the parentheses indicates for mussels the total number of individuals used in all pooled samples and for cod the number individuals in each pooled sample. Shaded values are below one half the detection limit, and indicate that over half of the values are below this limit. The standard deviation is based on all values. Caution should be used when comparing such values because the limit of detection can vary both within and among samples. Detectable data information (D.d.i.) indicates the number of data above the limit of detection (if any) and the numbers within the square brackets indicate the minimum and maximum values in this category. (See Appendix B for description of chemical codes.)*

Component	Count	BDE28			BDE47			BDE99			BDE100			BDE126			BDE153		
Species and sampling locality	2014	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i
Blue mussel																			
Gressholmen (st. 30A)	3(3-50)	0.005	0.000	3[0.005 - 0.005]	0.027	0.002	3[0.024 - 0.029]	0.010	0.000	3[0.01 - 0.01]	0.010	0.000	3[0.01 - 0.01]	0.001	0.000		0.001	0.000	
Tjøme (st. 36A1)	3(3-50)	0.005	0.000	3[0.005 - 0.005]	0.021	0.005	3[0.018 - 0.029]	0.010	0.000	3[0.01 - 0.01]	0.010	0.000	3[0.01 - 0.01]	0.001	0.000		0.001	0.000	
Singlekalven (st. I023)	3(3-50)	0.031	0.019	3[0.005 - 0.042]	0.054	0.023	3[0.03 - 0.075]	0.016	0.001	3[0.015 - 0.017]	0.010	0.000	3[0.01 - 0.01]	0.001	0.000		0.001	0.000	
Bjørkøya (st. 71A)	3(2-50)	0.005	0.002	2[0.005 - 0.005]	0.033	0.004	3[0.028 - 0.037]	0.023	0.003	3[0.018 - 0.024]	0.010	0.000	3[0.01 - 0.01]	0.001	0.000		0.001	0.000	
Croftholmen (st. I712)	1(1-50)	0.005	0.000	1[0.005]	0.036	0.000	1[0.036]	0.028	0.000	1[0.028]	0.010	0.000	1[0.01]	0.001	0.000		0.020	0.000	1[0.02]
Nordnes (st. I241)	1(1-50)	0.011	0.000	1[0.011]	0.244	0.000	1[0.244]	0.150	0.000	1[0.15]	0.052	0.000	1[0.052]	0.001	0.000		0.020	0.000	1[0.02]
Måløy (st. 26A2)	3(3-50)	0.014	0.002	3[0.01 - 0.014]	0.052	0.002	3[0.05 - 0.053]	0.024	0.003	3[0.02 - 0.026]	0.013	0.000	3[0.013 - 0.013]	0.001	0.000		0.002	0.000	
Outer Trondheimsfjord (st. 91A2)	3(3-50)	0.008	0.020	3[0.008 - 0.043]	0.050	0.030	3[0.048 - 0.102]	0.023	0.001	3[0.021 - 0.024]	0.012	0.001	3[0.012 - 0.015]	0.001	0.000		0.002	0.001	
Bodø harbour (st. 97A2)	3(3-50)	0.013	0.030	3[0.008 - 0.061]	0.066	0.048	3[0.061 - 0.146]	0.041	0.002	3[0.041 - 0.044]	0.024	0.003	3[0.019 - 0.024]	0.001	0.000		0.002	0.000	
Lofoten, Svolvær (st. 98A2)	3(3-50)	0.001	0.000		0.015	0.003	3[0.012 - 0.018]	0.010	0.000	3[0.01 - 0.01]	0.010	0.000	3[0.01 - 0.01]	0.001	0.000		0.002	0.000	
Cod, liver																			
Inner Oslofjord (st. 30B)	13(8-3)	0.265	0.000	12[0.21 - 0.93]	21.000	13.930	13[5.3 - 65]	0.280	0.405	11[0.14 - 1.5]	6.800	1.791	13[4.4 - 11]	0.050	0.000		0.050	0.014	1[0.15]
Færder area (st. 36B)	15(12-3)	0.091	0.000	15[0.14 - 0.47]	5.200	4.179	15[2 - 17]	0.050	0.150	2[0.2 - 0.68]	2.900	1.307	15[2 - 6.9]	0.050	0.000		0.050	0.000	
Kristiansand harbour (st. 13B)	14(7-2)	0.170	0.000	10[0.11 - 0.66]	4.550	3.989	14[1.8 - 18]	0.050	0.050	6[0.11 - 0.24]	1.200	0.979	14[0.58 - 4.5]	0.050	0.000		0.050	0.000	
Inner Sørfjord (st. 53B)	9(8-4)	0.148	0.000	9[0.14 - 0.65]	11.000	8.088	9[6.6 - 33]	0.050	0.132	2[0.13 - 0.5]	3.500	2.446	9[1.6 - 8.1]	0.050	0.000		0.050	0.000	
Bømlo north (st. 23B)	14(4-2)	0.062	0.000	8[0.12 - 0.27]	2.600	1.226	14[1.6 - 5.7]	0.050	0.122	5[0.12 - 0.56]	0.930	0.416	14[0.49 - 2]	0.050	0.029	1[0.21]	0.050	0.000	
Inner Trondheimsfjord (st. 80B)	15	1.210	0.000	15[0.2 - 4.8]	12.000	49.964	15[7.2 - 200]	0.610	0.279	15[0.15 - 1]	3.000	8.439	15[1.6 - 31]	0.050	0.000		0.050	0.000	
Lofoten, Skrova (st. 98B1)	8(7-2)	0.004	0.050	1[0.11]	1.100	0.785	8[0.87 - 3]	0.050	0.052	2[0.17 - 0.24]	0.410	0.127	8[0.29 - 0.65]	0.050	0.000		0.050	0.000	
Tromsø harbour (st. 43B2)	15	0.509	0.000	15[0.11 - 2.2]	6.300	7.704	15[4.1 - 35]	0.260	0.181	11[0.12 - 0.61]	1.500	1.983	15[0.87 - 8.8]	0.050	0.000		0.050	0.000	

Table 11. *(cont.)*

Component	Count	BDE154			BDE183			BDE196			BDE209			BDESS		
Species and sampling locality	2014	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i
Blue mussel																
Gressholmen (st. 30A)	3(3-50)	0.001	0.000		0.002	0.000		0.004	0.000		0.041	0.002		0.154	0.004	3[0.148 - 0.156]
Tjøme (st. 36A1)	3(3-50)	0.001	0.000		0.002	0.000		0.004	0.000		0.045	0.004		0.154	0.010	3[0.144 - 0.165]
Singlekalven (st. I023)	3(3-50)	0.001	0.000		0.002	0.000		0.004	0.001		0.040	0.005		0.202	0.044	3[0.157 - 0.245]
Bjørkøya (st. 71A)	3(2-50)	0.001	0.000		0.002	0.000		0.004	0.000		0.042	0.002		0.172	0.008	3[0.157 - 0.172]
Croftholmen (st. I712)	1(1-50)	0.020	0.000	1[0.02]	0.002	0.000		0.004	0.000		0.058	0.000		0.246	0.000	1[0.246 - 0.246]
Nordnes (st. I241)	1(1-50)	0.020	0.000	1[0.02]	0.004	0.000		0.007	0.000		0.500	0.000	1[0.5]	1.018	0.000	1[1.018 - 1.018]
Måløy (st. 26A2)	3(3-50)	0.002	0.000		0.003	0.000		0.007	0.000		1.880	0.221	3[1.53 - 1.94]	2.004	0.221	3[1.659 - 2.071]
Outer Trondheimsfjord (st. 91A2)	3(3-50)	0.002	0.000		0.003	0.000		0.007	0.001		0.069	0.007		0.255	0.061	3[0.25 - 0.357]
Bodø harbour (st. 97A2)	3(3-50)	0.002	0.000		0.003	0.000		0.005	0.001		0.053	0.016		0.266	0.096	3[0.247 - 0.422]
Lofoten, Svolvær (st. 98A2)	3(3-50)	0.002	0.000		0.003	0.000		0.005	0.000		0.052	0.003		0.163	0.002	3[0.161 - 0.165]
Cod, liver																
Inner Oslofjord (st. 30B)	13(8-3)	2.200	0.553	13[1.5 - 3.3]	0.150	0.417		0.150	0.000		0.250	0.000		31.720	15.558	13[16.42 - 81.43]
Færder area (st. 36B)	15(12-3)	1.500	0.328	15[0.59 - 1.7]	0.150	0.000		0.150	0.000		0.250	0.000		10.650	5.728	15[6.17 - 27.55]
Kristiansand harbour (st. 13B)	14(7-2)	0.715	0.332	14[0.34 - 1.3]	0.150	0.000		0.150	0.000		0.250	0.000		7.765	5.296	14[4.12 - 25.76]
Inner Sørfjord (st. 53B)	9(8-4)	1.400	0.971	9[0.65 - 3.8]	0.150	0.000		0.150	0.000		0.250	0.000		18.520	11.110	9[10.29 - 45.68]
Bømlo north (st. 23B)	14(4-2)	0.810	0.205	14[0.57 - 1.2]	0.150	0.426		0.150	0.000		0.250	0.000		6.075	1.867	14[4.38 - 10.43]
Inner Trondheimsfjord (st. 80B)	15	0.860	0.791	15[0.43 - 3.2]	0.150	0.000		0.150	0.000		0.250	0.000		18.830	59.914	15[11.27 - 240.35]
Lofoten, Skrova (st. 98B1)	8(7-2)	0.225	0.141	7[0.1 - 0.51]	0.150	0.000		0.150	0.000		0.250	0.000		3.245	0.990	8[2.66 - 5.35]
Tromsø harbour (st. 43B2)	15	0.550	0.639	15[0.27 - 2.6]	0.150	0.347		0.150	0.000		0.250	0.000		10.460	10.740	15[6.93 - 50.28]

***Table 12**. Median concentrations (µg/kg w.w.) and standard deviations for PBDE congeners in blue mussel and cod liver in supplementary analsyses of 2013 samples (indicated with an asterisk *). BDESS indicates sum of all BDEs. Count indicates number of samples analysed. The first number within the parentheses indicates the number of pooled samples included. The second number within the parentheses indicates for mussels the total number of individuals used in all pooled samples and for cod the number individuals in each pooled sample. Shaded values are below one half the detection limit, and indicate that over half of the values are below this limit. The standard deviation is based on all values. Caution should be used when comparing such values because the limit of detection can vary both within and among samples. Detectable data information (D.d.i.) indicates the number of data above the limit of detection (if any) and the numbers within the square brackets indicate the minimum and maximum values in this category. (See Appendix B for description of chemical codes.)*

Component	Count	BDE28			BDE47			BDE99			BDE100			BDE126			BDE153		
Species and sampling locality	2013	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i
Blue mussel																			
Akershuskaia (st. I301)*	2(2-50)	0.005	0.000		0.278	0.138	2[0.18-0.376]	0.206	0.101	2[0.135-0.277]	0.063	0.026	2[0.045-0.082]	0.005	0.000		0.012	0.004	1[0.015]
Gressholmen (st. 30A)	3(3-204)				0.058	0.007	3[0.054-0.067]	0.037	0.003	3[0.033-0.04]	0.014	0.001	3[0.014-0.016]	0.001	0.000		0.001	0.000	
Gåsøya (st. I304)*	2(2-15)	0.005	0.000		0.041	0.014	2[0.031-0.051]	0.014	0.005	2[0.011-0.018]	0.011	0.001	1[0.012]	0.005	0.000		0.005	0.000	
Solbergstrand (st. 31A)*	1(1-48)	0.005	0.000		0.027	0.000	2[0.027-0.029]	0.005	0.000		0.005	0.000		0.005	0.000		0.005	0.000	
Mølen (st. 35A)*	3(3-88)	0.005	0.000		0.050	0.001	3[0.048-0.05]	0.028	0.005	3[0.028-0.036]	0.013	0.001	3[0.013-0.014]	0.005	0.000		0.005	0.000	
Tjøme (st. 36A1)	3(3-50)				0.029	0.003	3[0.025-0.031]	0.013	0.001	3[0.011-0.013]	0.008	0.000	3[0.007-0.008]	0.001	0.000		0.001	0.000	
Singlekalven (st. I023)	3(3-33)				0.016	0.002	3[0.015-0.018]	0.008	0.001	3[0.008-0.009]	0.003	0.000	3[0.003-0.004]	0.001	0.000		0.001	0.000	
Bjørkøya (st. 71A)	3(3-20)				0.029	0.003	3[0.025-0.031]	0.019	0.001	3[0.018-0.02]	0.009	0.001	3[0.009-0.011]	0.000	0.001		0.001	0.001	
Croftholmen (st. I712)	2(2-9)				0.030	0.009	2[0.023-0.036]	0.023	0.010	2[0.016-0.03]	0.012	0.004	2[0.009-0.015]	0.000	0.000		0.001	0.000	
Nordnes (st. I241)	2(2-20)				0.216	0.001	2[0.215-0.217]	0.130	0.012	2[0.122-0.139]	0.045	0.002	2[0.044-0.046]	0.001	0.000		0.009	0.000	
Måløy (st. 26A2)	3(3-53)				0.026	0.001	3[0.024-0.026]	0.011	0.001	3[0.011-0.013]	0.007	0.000	3[0.007-0.007]	0.001	0.000		0.001	0.000	
Outer Trondheimsfjord (st. 91A2)	3(3-72)				0.041	0.005	3[0.037-0.046]	0.013	0.002	3[0.012-0.016]	0.011	0.001	3[0.01-0.012]	0.000	0.000		0.001	0.000	
Bodø harbour (st. 97A2)	3(3-230)				0.052	0.020	3[0.031-0.07]	0.029	0.008	3[0.019-0.036]	0.018	0.008	3[0.011-0.027]	0.001	0.000		0.001	0.000	
Lofoten, Svolvær (st. 98A2)	3(3-99)				0.012	0.001	3[0.01-0.012]	0.004	0.000	3[0.003-0.004]	0.004	0.000	3[0.004-0.005]	0.001	0.000		0.001	0.000	

Table 12. *(cont.)*

Component	Count	BDE154			BDE183			BDE196			BDE209			BDESS		
Species and sampling locality	2013	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i
Blue mussel																
Akershuskaia (st. I301)*	2(2-50)	0.013	0.004	1[0.015]	0.005	0.000		0.005	0.000		0.080	0.028	2[0.06-0.1]	0.740	0.255	n.a.
Gressholmen (st. 30A)	3(3-204)	0.001	0.000		0.002	0.003		0.004	0.000		0.039	0.002		0.202	0.009	n.a.
Gåsøya (st. I304)*	2(2-15)	0.005	0.000		0.005	0.000		0.005	0.000		0.014	0.006	2[0.01-0.019]	0.180	0.026	n.a.
Solbergstrand (st. 31A)*	1(1-48)	0.005	0.000		0.005	0.000		0.005	0.000		0.033	0.000	2[0.033-0.034]	0.180	0.000	n.a.
Mølen (st. 35A)*	3(3-88)	0.005	0.000		0.005	0.000		0.005	0.000		0.024	0.004	3[0.022-0.029]	0.216	0.010	n.a.
Tjøme (st. 36A1)	3(3-50)	0.001	0.000		0.002	0.000		0.003	0.001		0.032	0.008		0.119	0.008	n.a.
Singlekalven (st. I023)	3(3-33)	0.001	0.000		0.001	0.000		0.003	0.000		0.027	0.004		0.092	0.006	n.a.
Bjørkøya (st. 71A)	3(3-20)	0.001	0.001		0.001	0.002		0.002	0.003		0.040	0.030		0.143	0.034	n.a.
Croftholmen (st. I712)	2(2-9)	0.001	0.000		0.001	0.000		0.002	0.000		0.018	0.001		0.109	0.022	n.a.
Nordnes (st. I241)	2(2-20)	0.007	0.000	2[0.006-0.007]	0.006	0.001		0.003	0.000		0.030	0.003		0.477	0.011	n.a.
Måløy (st. 26A2)	3(3-53)	0.001	0.000		0.001	0.000		0.003	0.000		0.030	0.001		0.115	0.004	n.a.
Outer Trondheimsfjord (st. 91A2)	3(3-72)	0.001	0.000		0.001	0.000		0.002	0.001		0.028	0.052		0.129	0.061	n.a.
Bodø harbour (st. 97A2)	3(3-230)	0.001	0.000		0.001	0.000		0.003	0.000		0.029	0.005		0.169	0.041	n.a.
Lofoten, Svolvær (st. 98A2)	3(3-99)	0.001	0.000		0.002	0.000		0.003	0.000		0.034	0.002		0.102	0.003	n.a.

Levels in blue mussel

Only congeners BDE28, -47, -99 and -100 showed concentrations above the detection limit for half or more of the samples at a station (***Table 11***, ***Figure 31***, ***Table 10***).

The most dominant congener in 2014 was BDE47, as was also the case in the previous year. BDE47, -99 and -100 were detected at all 10 stations in 2014, as in 2013. For these congeners, the highest median concentrations were found in mussels from Nordnes (st. I241) in Bergen harbour (0.244 µg BDE47/kg w.w., 0.15 µg BDE99/kg w.w. and 0.052 µg BDE100/kg w.w.). The highest concentrations of BDE153 (0.02 µg/kg w.w.) and BDE154 (0.02 µg/kg w.w.) were also found at Nordnes close to the center of Bergen and at Croftholmen (st. I712) in the Outer Grenlandsfjord. There were insufficient data to do temporal trend analysis except for BDE47 at Gressholmen (st. 30A), Bjørkøya (st. 71A) and Lofoten (st. 98A2), and for BDE99 and -100 at Gressholmen and Bjørkøya. Both significant downward long-term and short-term trends were found for BDE47 at Gressholmen (st. 30A). No trends were observed for BDE47 at Bjørkøya (st. 71A) and Lofoten (st. 98A2), and for BDE99 and -100 at Gressholmen and Bjørkøya.

Blue mussel from Nordnes in the Bergen harbour area showed significantly higher concentrations of BDE47, -99, and -100 than mussels from all the other stations (Tukey-Kramer HSD test).

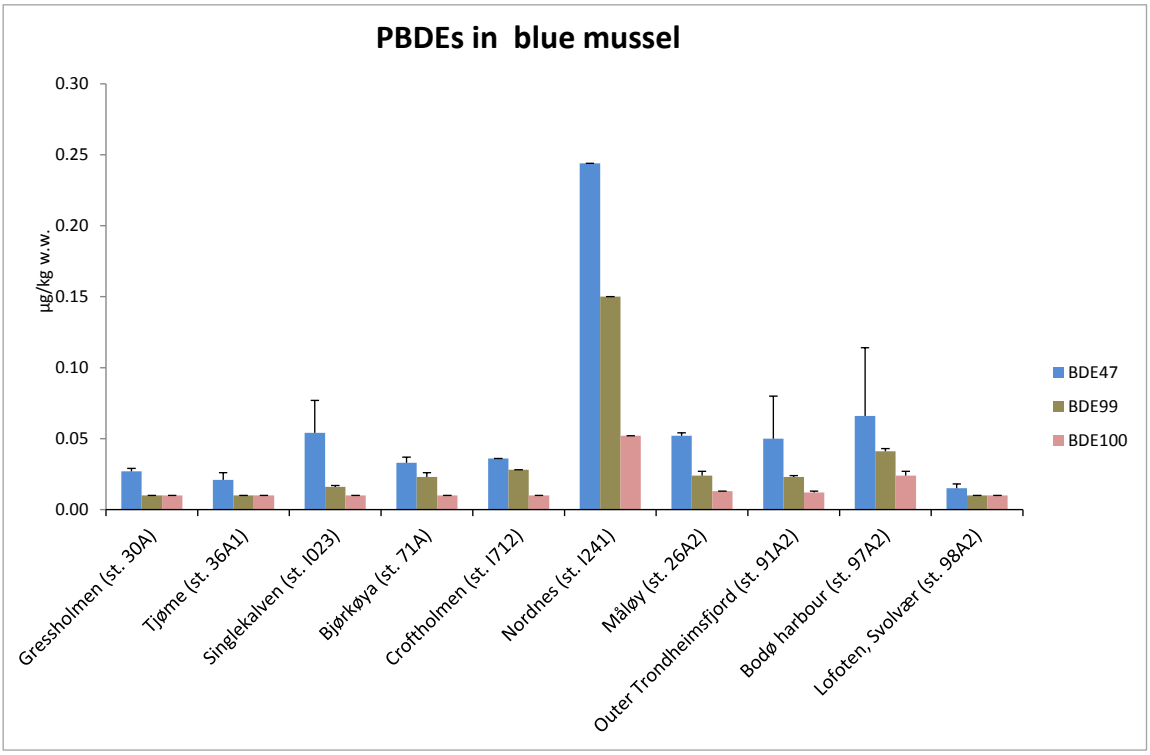

Figure 31*. Median concentrations (µg/kg w.w.) of PBDEs in blue mussel in 2014. Only the results are shown where concentrations were above the detection limit for half or more of the samples. The error bar indicates one standard deviation above the median.*

Environmental Quality Standards (EQS)

The EQS (2013/39/EC) for polybrominated diphenylethers (0.0085 µg/kg w.w.) in biota for "fish" is the sum of the concentrations of congener numbers 28, 47, 99, 100, 153 and 154. This EQS applies to whole fish. Therefore, the EQS cannot be directly compared to concentrations found in different tissues of fish. The median concentration of BDE47 alone in cod liver would have exceeded this EQS value at all stations. These results indicate that the EQS might not be a useful criterion to judge the condition of the environment with respect to this contaminant in biota.

Inner Oslofjord

Parts of the Inner Oslofjord are densely populated with much urban activities and accompanying PBDEs in certain products. The high concentrations of PBDEs observed in cod are probably related to these activities, as well as reduced water exchange with the Outer fjord. A study of flounder liver from the Inner Oslofjord in 2013 showed generally substantially lower (e.g. a factor of~35 for BDE47) than the median concentration measured in cod in 2012 (Ruus *et al*. 2014). The congener BDE47 was also dominating at three blue mussel stations (Frognerkilen, Alna and Bekkelaget) in the Inner Oslofjord in 2013 (Ruus *et al*. 2014).

Other studies

Median concentrations for the sum of PBDEs found at presumed reference stations like Lofoten, Færder, Utsira and Bømlo-Sotra indicate that a high background level in diffusely contaminated areas might be about 30 µg/kg w.w. for cod liver (Fjeld *et al*. 2005). This is higher than the sum of the medians BDE47, -100, -154, -183, and -196 found at MILKYS cod stations in the Inner Oslofjord (cf. ***Figure 29***) and higher than the average concentrations found at two cod stations in the North Sea (14.6 and 15.4 µg/kg w.w.) (Green *et al*. 2011a) and three cod stations in the Norwegian Sea (5.89, 12.9 and 19 µg/kg w.w.) (Green *et al*. 2012b). It cannot be disregarded that this high background concentration might be too high. The median found in the Inner Oslofjord for sum PBDE/BDEs (31.72 µg/kg w.w.) was for the first time below the interval for sum PBDEs of 37-112 µg/kg w.w. found in other contaminated areas (Fjeld *et al*. 2005, Berge *et al*. 2006). Bakke *et al*. (2007b) found mean concentrations of sum of PBDEs in remote areas to be within the range 3.4-29.0 µg/kg w.w.

The congeners BDE47 and -100 were observed to be most dominant in 2014, as in 2013. The low concentrations of BDE99 could be due to the debromination to BDE47, because BDE99 is more suseptable to biotransformation than other common PBDE such as BDE47 (Streets *et al*. 2006). Furthermore, BDE47 is a more stable congener than BDE99, (Benedict *et al*. 2007). Investigations of brown trout (*Salmo trutta)*, smelt (*Osmerus eperlanus*) and vendace (*Coregonus albula*) in lake Mjøsa showed that the decrease was greatest for BDE99, which probably is due to a biotransformation (debromination) to BDE47 (Fjeld *et al*. 2012). In recent years, there has been a clear reduction of PBDE-concentrations in freshwater fish from Mjøsa (Løvik *et al*. 2015).

Supplementary analyses of blue mussel samples 2013

Supplementary analyses of PBDEs were carried out on 2013 samples at the four blue mussel stations in the Inner Oslofjord; Akershuskaia (st. I301), Gåsøya (st. 30A), Solbergstrand (st. 31A) and Mølen (st. 34A) (***Table 12***). The highest concentrations of BDE47, -99, -100, -153, -154 and -209 of all national stations were found in mussels from the harbour area at Akershuskaia. All BDEs, except for PDE-47 and-209, were below detection limits in the Outer Oslo fjord at Solbergstrand. Blue mussel from Akershuskaia and from Nordnes in the Bergen harbour area showed significantly higher concentrations of BDE47, -99, and -100 than mussels from all the other stations (Tukey-Kramer HSD test).

General, large scale trends

There were seven recent significant downward trends; BDE47, -99, -100 and -154 in cod liver at Tromsø harbour (st. 43B2), BDE-47 and -100 in cod liver in the Inner Oslofjord (st. 30B) and BDE47 in blue mussel at Gressholmen (st. 30A).

One recent significant trend was upward; BDE154 in cod liver at Bømlo (st. 23B).

There were 11 significant downward long-term trends in cod liver. This was found for BDE99 and -100 in the Inner Oslofjord (st. 30B), for BDE47 from Færder (st. 36B), for BDE28, -47 and -100 from Bømlo (st. 23B), for BDE47 from Lofoten (st. 98B1), and for BDE47, -99, -100 and -154 from Tromsø harbour (st. 43B2). There was also one significant downward long-term trend in blue mussel for BDE47 at Gressholmen (st. 30A).

These results are more in line with the general decreasing trend of penta-mix PBDEs (that includes BDE100, Law *et al*. 2014), PBDEs in European emissions (Schuster *et al*. 2010) and in marine mammals in the Arctic and North Atlantic since 2000 (Rotander *et al*. 2012). It can be noted that after 2002 a sharp decline in concentrations of PBDEs (as well as PFCs) was observed in blood from newborns in New York state (Ma *et al*. 2013).

3.2.18 Perfluorinated alkylated substances (PFAS)

Perfluorinated alkylated substances (PFAS) are organofluorine compounds used as oil-, stain- and water-repellant surfactants and a number of other products. PFAS were analysed in cod liver at eight stations (***Table 10*** and ***Figure 32***). PFAS have been analysed annually in cod liver since 2005. Samples collected in the Inner Oslofjord (st. 30B) and Bømlo (st. 23B) in 1993 have also been analysed for PFAS.

Levels and trends

PFOS and PFOSA at all stations revealed assumed background concentrations. Significant downward trends for PFOS were dominating in 2014, as in 2013.

PFOS

The median concentration of perfluoroctonoic sulphonate (PFOS) was highest in the Inner Oslofjord (st. 30B, 5.5 µg/kg w.w.) and lowest in the Inner Trondheimsfjord (st. 80B, 0.6 µg/kg w.w.) (***Table 13***). The concentration found in the Inner Oslofjord had increased since 2013 from 3.24 µg/kg w.w.) in 2013 to 5.5 µg/kg w.w. in 2014, and at Færder (st. 36B) the concentrations had increased from 0.775 µg/kg w.w. in 2013 to 3.2 µg/kg w.w. in 2014. Significant downward trends were identified at seven of the eight stations. There were both significant downward long-term and short-term trends for PFOS at Færder (st. 36B), Kristiansand harbour (st. 13B), the Inner Sørfjord (st. 53B), the Inner Trondheimsfjord (st. 80B) and Tromsø harbour (st. 43B2). Significant downward short-term trends were observed in the Inner Oslofjord (st. 30B) and at Bømlo (st. 23B).

PFOSA

Perfluorooctane sulphonamide (PFOSA) had a maximum median concentration of 9.2 µg/kg w.w. in the Inner Oslofjord, and a minimum level at Tromsø harbour (0.4 µg/kg w.w.). The concentration of PFOSA was higher than PFOS in the Inner Oslofjord and Færder (***Figure 32***, ***Figure 33)***. The median concentrations of the remaining PFAS were below the detection limits at Færder, Inner Sørfjord, Inner Trondheimsfjord and Tromsø harbour (***Table 10***, ***Table 13***).

Cod from the Inner Oslofjord and Outer Oslofjord had significant higher levels of PFOS and PFOSA in liver than all other stations (Tukey-Kramer HSD test, see also ***Figure 32***).

Environmental Quality Standards (EQS)
The EQS (2013/39/EC) for PFOS in biota (fish) is 9.1 µg/kg w.w. which applies to whole fish. Therefore, the EQS cannot be directly compared to concentrations found in different tissues of fish. We have in this study only measured PFOS in liver and have not considered converting liver to whole fish because this conversion is uncertain. If it is assumed, for this exercise, that the same concentration is found in cod liver as in the whole fish, then the results of PFOS would not be exceeded at any station (maximum concentration 5.5 µg/kg w.w. in the Inner Oslofjord).

Inner Oslofjord
Parts of the Inner Oslofjord are densely populated with much urban activities including presence of PFOSA in certain products. PFOSA is a precursor compounds in the production of fluorinated polymers but may also add to the exposure due to their degradation into PFOS. The high concentrations of PFOSA observed in cod are probably related to these activities, as well as reduced water exchange with the Outer fjord. PFOS was the dominant PFAS in cod liver in the Inner Oslofjord in 2009 (median 48 µg/kg w.w.) compared with PFOSA (41.5 µg/kg w.w.). In 2010, 2011, 2012, 2013 and 2014, PFOSA was the dominating substance (18, 19, 10, 7 and 9 µg/kg w.w., respectively) compared to PFOS (16, 5, 7, 3 and 6 µg/kg w.w., respectively). Schøyen & Kringstad (2011) analysed PFAS in cod blood samples from the same individuals which were analysed in the MILKYS programme in 2009 from the Inner Oslofjord (Green *et al*. 2010b). They found that PFOSA was the most dominant PFAS-compound with a median level 6 times higher than for PFOS. The median level of PFOSA in cod blood was about 5 times higher than in liver. The median level of PFOS in cod liver was about 1.5 times higher than in blood. Further, PFNA was also detected in cod blood. Rundberget *et al*. (2014) investigated cod from Inner Oslofjord (st. 30B) in the period 2009 to 2013 and found that blood was the preferred matrix for analysing PFAS. The levels of PFOS were roughly the same in blood as in liver and bile, but levels of other PFAS were higher in blood and therefor easier to detect. A study of flounder liver from the Inner Oslofjord in 2013 showed higher median concentration of PFOS than in cod in 2012, while the median concentration of PFOSA was lower in cod from 2012 (Ruus *et al*. 2014).

Other studies
Median concentrations of PFOS in cod liver from presumed reference stations like Lofoten, Kvænangen/Olderfjord north of Skjervøy and the Varangerfjord indicated that high background concentrations in only diffusely contaminated areas might be around 10 µg/kg w.w. (Bakke *et al*. 2007). All concentrations observed in this study were lower. The average concentration of PFOS in cod liver from two stations in the North Sea was 1.55 and 0.95 µg/kg w.w. (Green *et al*. 2011a) and from three stations in the Norwegian Sea was 0.75, 0.82 and 11 µg/kg w.w. (Green *et al*. 2012b).

PFAS in freshwater fish was investigated in 2014 (Fjeld *et al*. 2015). The concentrations of long-chained compounds, like PFOS and PFOSA, increased with trophic levels with the highest levels in brown trout (*Salmo trutta*) liver. The mean PFOS-concentrations in fish liver from the three main lakes (Mjøsa, Randsfjorden and Femunden) were in the range of 2 – 11 µg/kg w.w. The PFOS-levels were considerably elevated in perch (*Perca fluviatilis*) liver from Tyrifjorden and Vansjø with mean concentrations of 130 and 173 µg/kg w.w., respectively. Concentrations of PFOS in liver varied considerably but were on the average about 30 times higher than in fillet. The differences between fillet and liver concentrations seemed to increase with decreasing carbon chain length.

PFOA has been strictly regulated nationally in consumer products from June 2014[10]. PFOA-data at all stations was inadequate for trend analysis due to concerns about the limit of detections.

General, large scale trends

Seven of the eight stations showed significant downward trends in PFOS for the period 2005 to 2014. Significant downward trends for PFOS were dominating in 2013 and 2014, unlike the previous year (2012) when no trends were observed. The observed downwars trends could reflect the overall reduction in production and use of PFAS for the past 30 years (Nost *et al*. 2014, Axmon *et al*. 2014). It is however unclear why downward trends was not seen in 2012. A decrease in concentrations of PFAS in Sweden has been reported for food items (Johansson *et al*. 2014) and herring (Ullah *et al*. 2014). A sharp decline in concentrations of PFAS (as well as PBDEs) after 2002 was found in dried blood spots from newborns in New York state (Ma *et al*. 2013).

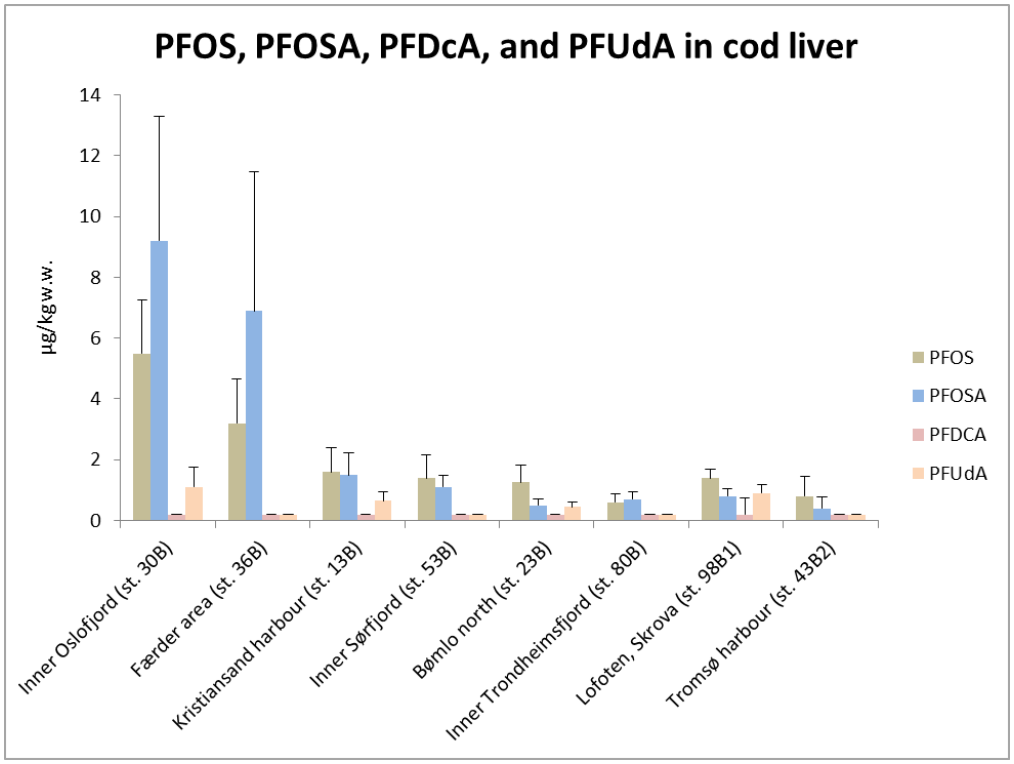

***Figure 32**. Median concentrations (µg/kg w.w.) of four PFAS compounds in cod liver in 2014. The error bar indicates one standard deviation above the median. PFDcA and PFUdA values for some stations are below the limit of detection - see **Table 13**).*

[10] http://www.miljodirektoratet.no/no/Nyheter/Nyheter/2014/Mars-2014/Overgangsordning-for-miljogiften-PFOA-i-forbrukerprodukter/

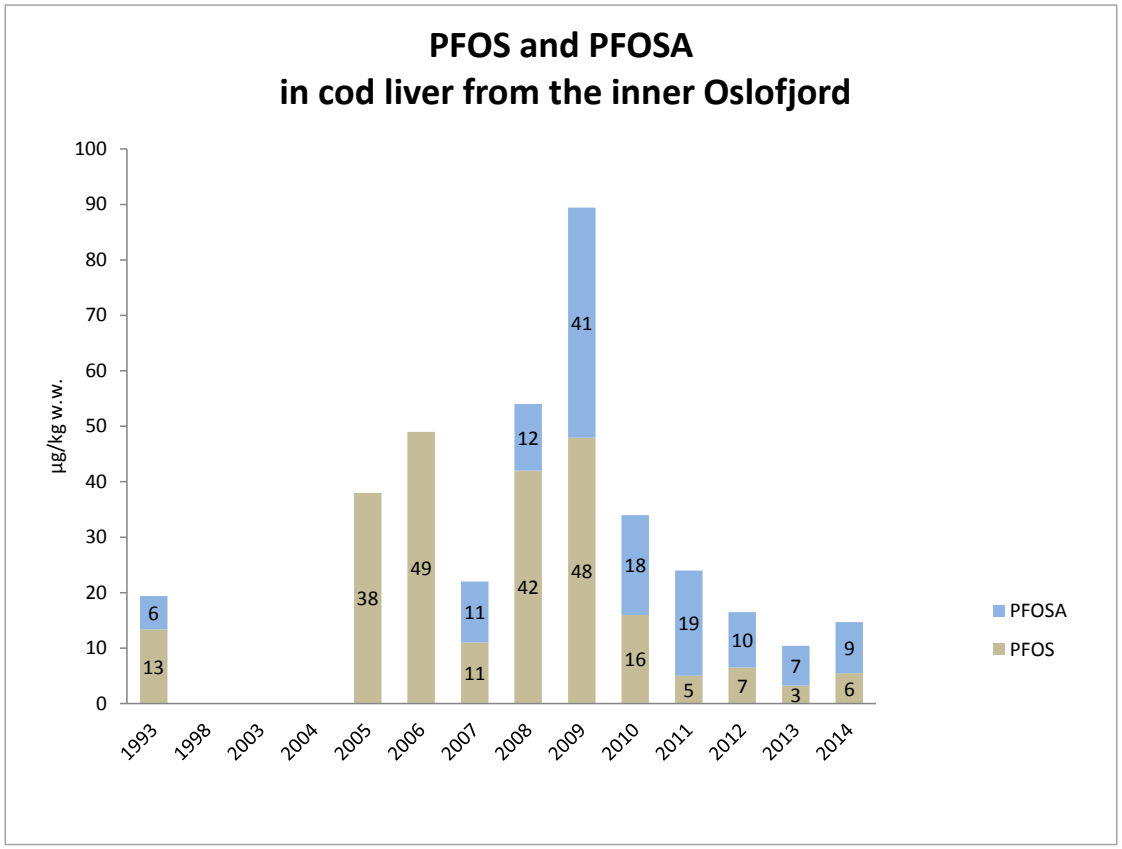

***Figure 33**. Median concentrations (µg/kg w.w.) of PFOS and PFOSA in cod liver from 1993 to 2014 in the Inner Oslofjord (st. 30B).*

***Table 13**. Median concentrations (µg/kg w.w.) and standard deviations of the PFAS-compounds analysed in cod liver in 2014. The first number within the parentheses indicates the number of pooled samples included. The second number within the parentheses indicates for cod the number individuals in each pooled sample. Shaded values are below one half the detection limit, and indicate that over half of the values are below this limit. The standard deviation is based on all values. Caution should be used when comparing such values because the limit of detection can vary both within and among samples. Detectable data information (D.d.i.) indicates the number of data above the limit of detection (if any) and the numbers within the square brackets indicate the minimum and maximum values in this category. (See Appendix B for description of chemical codes.)*

Component	Count	PFBS			PFDCA			PFDCS			PFHPA			PFHXA			PFHXS		
Species and sampling locality	2014	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.
Cod, liver																			
Inner Oslofjord (st. 30B)	13(8-3)	0.05	0		0.2	0		0.6	0.158	13[0.3 - 0.9]	0.2	0		0.2	0		0.05	0	
Færder area (st. 36B)	15(12-3)	0.05	0		0.2	0		0.05	0		0.2	0		0.2	0		0.05	0	
Kristiansand harbour (st. 13B)	14(7-2)	0.05	0		0.2	0		0.05	0		0.2	0		0.2	0		0.05	0	
Inner Sørfjord (st. 53B)	9(8-4)	0.05	0		0.2	0		0.05	0		0.2	0		0.2	0		0.05	0	
Bømlo north (st. 23B)	14(4-2)	0.05	0		0.2	0		0.05	0		0.2	0		0.2	0		0.05	0	
Inner Trondheimsfjord (st. 80B)	15	0.05	0		0.2	0		0.05	0		0.2	0		0.2	0		0.05	0	
Lofoten, Skrova (st. 98B1)	8(7-2)	0.05	0		0.2	0.53	1[1.9]	0.05	0		0.2	0		0.2	0		0.05	0	
Tromsø harbour (st. 43B2)	15	0.05	0		0.2	0		0.05	0		0.2	0		0.2	0		0.05	0	

***Table 13**.* (cont.)

Component	Count	PFNA			PFOA			PFOS			PFOSA			PFUDA		
Species and sampling locality	2014	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.
Cod, liver																
Inner Oslofjord (st. 30B)	13(8-3)	0.2	0		0.2	0		5.5	1.747	13[3.1 - 8.9]	9.2	4.104	13[3.1 - 18]	1.1	0.663	11[0.8 - 2.4]
Færder area (st. 36B)	15(12-3)	0.2	0		0.2	0		3.2	1.454	15[1.6 - 6.3]	6.9	4.57	15[3.2 - 20.9]	0.2	0	
Kristiansand harbour (st. 13B)	14(7-2)	0.2	0		0.2	0		1.6	0.784	14[0.9 - 3.9]	1.5	0.72	14[0.6 - 3.2]	0.65	0.289	8[0.6 - 1.2]
Inner Sørfjord (st. 53B)	9(8-4)	0.2	0		0.2	0		1.4	0.75	9[1 - 3.3]	1.1	0.388	9[0.7 - 1.8]	0.2	0	
Bømlo north (st. 23B)	14(4-2)	0.2	0		0.2	0		1.25	0.565	14[0.7 - 2.6]	0.5	0.198	14[0.3 - 1]	0.45	0.174	7[0.5 - 0.9]
Inner Trondheimsfjord (st. 80B)	15	0.2	0		0.2	0		0.6	0.28	15[0.3 - 1.3]	0.7	0.246	15[0.4 - 1.3]	0.2	0	
Lofoten, Skrova (st. 98B1)	8(7-2)	0.2	0		0.2	0		1.4	0.287	8[1 - 1.9]	0.8	0.238	8[0.5 - 1.2]	0.9	0.296	8[0.5 - 1.5]
Tromsø harbour (st. 43B2)	15	0.2	0		0.2	0		0.8	0.645	15[0.3 - 2.5]	0.4	0.376	15[0.2 - 1.3]	0.2	0	

3.3 New contaminants

3.3.1 Hexabromcyclododecanes (HBCD)

Hexabromcyclododecanes (HBCD) was analysed in cod liver at 10 stations and in blue mussel at 10 stations.

HBCD is a persistent pollutant with a high potential for bioaccumulation. HBCD is one of the substances identified as priority hazardous substances (2013/39/EU). The EQS (167 µg/kg w.w.) refers to fish and this threshold was not exceeded by any median concentration if it is assumed that this median applies to the whole organism and not just the liver. Cod from the Inner Oslofjord had the highest concentration of HBCD in the liver (***Figure 34***). HBCD is here the sum of the α–, β–, and γ–diastereomers. The median concentration of HBCD in cod liver from the Inner Oslofjord (st. 30B) was 14.82 µg/kg w.w. (***Table 14***). Parts of the Inner Oslofjord are densely populated facilitateing urban activities which could bring about use of products containing HBCD. The high concentrations of HBCD observed in cod are probably related to leaching of HBCD from such products, as well as to reduced water exchange with the outer fjord.

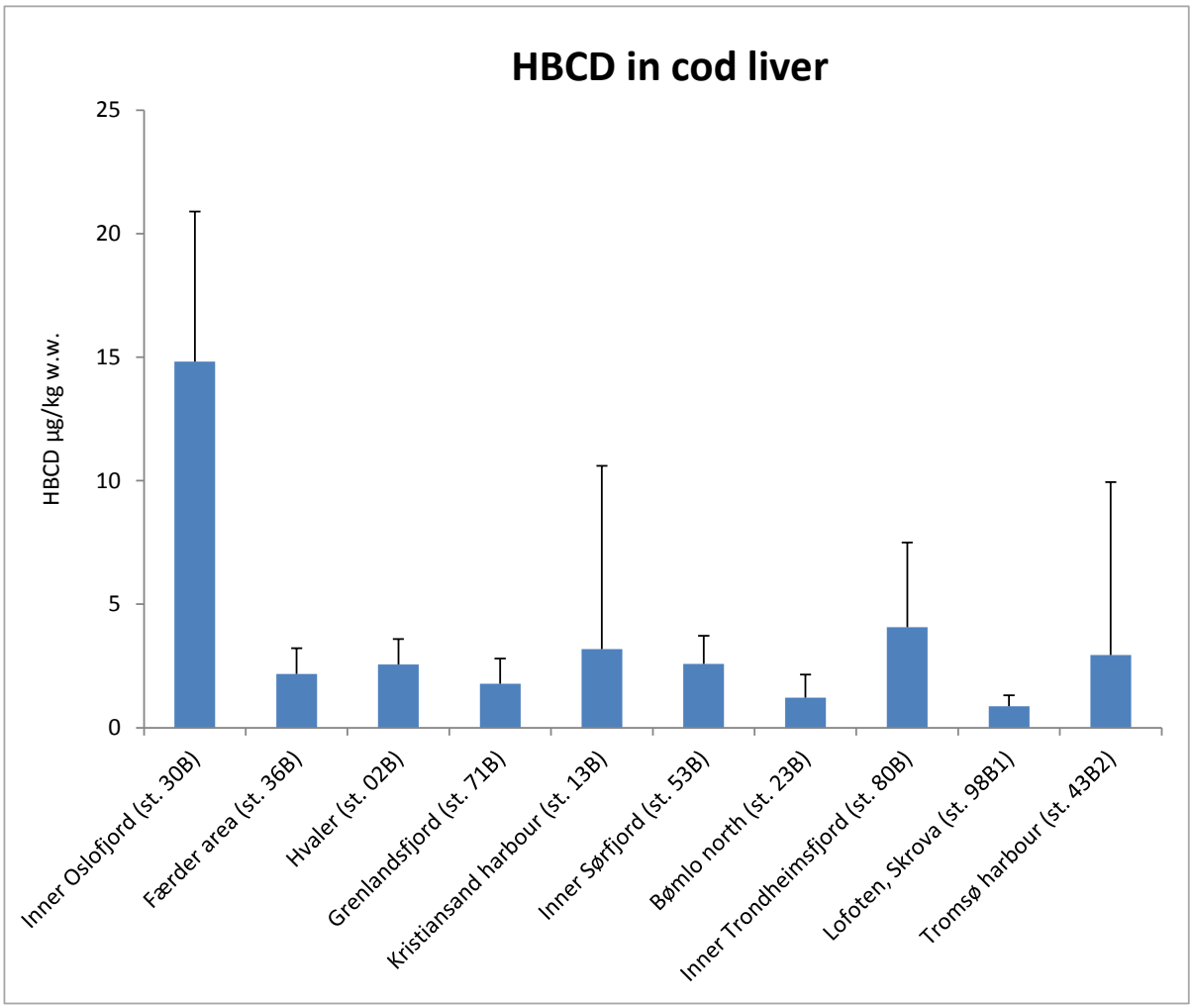

Figure 34*. Median concentration (µg/kg w.w.) of HBCD (sum of the α–, β–, and γ–diastereomers) in cod liver in 2014. The error bar indicates one standard deviation above the median.*

***Table 14**. Median concentration (µg/kg w.w.) with standard deviation of HBCD (sum of the α–, β–, and γ–diastereomers) in cod liver and blue mussel in 2014. Count indicates number of samples analysed. The first number within the parentheses indicates the number of pooled samples included. The second number within the parentheses indicates for mussels the total number of individuals used in all pooled samples and for cod the number individuals in each pooled sample. Shaded values are below one half the detection limit and indicate that over half of the values are below this limit. The standard deviation is based on all values. Caution should be used when comparing such values because the limit of detection can vary both within and among samples. Detectable data information (D.d.i.) indicates the number of data above the limit of detection (if any) and the numbers within the square brackets indicate the minimum and maximum values in this category.* (See Appendix B for description of chemical codes.)

Component	Count	α–HBCD			γ–HBCD			β–HBCD			HBCD		
Species and sampling locality	2014	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.
Blue mussel													
Gressholmen (st. 30A)	3(3-50)	0.110	0.006	3[0.11 - 0.12]	0.050	0.021	3[0.04 - 0.08]	0.004	0.001	1[0.01]	0.170	0.026	3[0.158 - 0.208]
Tjøme (st. 36A1)	3(3-50)	0.060	0.000	3[0.06 - 0.06]	0.040	0.010	3[0.03 - 0.05]	0.005	0.001		0.108	0.010	3[0.099 - 0.119]
Singlekalven (st. I023)	3(3-50)	0.050	0.010	3[0.04 - 0.06]	0.004	0.000		0.004	0.000		0.066	0.010	3[0.056 - 0.076]
Bjørkøya (st. 71A)	3(2-50)	0.040	0.010	3[0.03 - 0.05]	0.004	0.001		0.004	0.000	1[0.008]	0.056	0.011	3[0.045 - 0.066]
Croftholmen (st. I712)	1(1-50)	0.040	0.000	1[0.04]	0.008	0.000	1[0.008]	0.004	0.000		0.055	0.000	1[0.055]
Nordnes (st. I241)	1(1-50)	0.260	0.000	1[0.26]	0.060	0.000	1[0.06]	0.060	0.000	1[0.06]	0.380	0.000	1[0.38]
Måløy (st. 26A2)	3(3-50)	0.080	0.006	3[0.08 - 0.09]	0.005	0.000	1[0.01]	0.010	0.006	2[0.01 - 0.02]	0.100	0.012	3[0.1 - 0.12]
Outer Trondheimsfjord (st. 91A2)	3(3-50)	0.06	0.015	3[0.05 - 0.08]	0.005	0.000		0.005	0.006	1[0.02]	0.090	0.015	3[0.07 - 0.1]
Bodø harbour (st. 97A2)	3(3-50)	0.06	0.01	3[0.05 - 0.07]	0.01	0.001	2[0.01 - 0.01]	0.005	0.006	1[0.02]	0.090	0.013	3[0.068 - 0.09]
Lofoten, Svolvær (st. 98A2)	3(3-50)	0.02	0.006	3[0.02 - 0.03]	0.005	0.012	1[0.03]	0.005	0.000		0.040	0.017	3[0.04 - 0.07]
Cod, liver													
Inner Oslofjord (st. 30B)	13(8-3)	14.600	6.111	13[5.11 - 32]	0.120	0.225	10[0.08 - 0.71]	0.190	0.318	11[0.06 - 1.06]	14.820	6.083	13[6.23 - 32.87]
Færder area (st. 36B)	15(12-3)	2.020	0.995	15[0.44 - 3.7]	0.030	0.006		0.100	0.110	9[0.08 - 0.42]	2.180	1.043	15[0.56 - 3.86]
Hvaler (st. 02B)	3(3-3)	2.410	1.104	3[0.77 - 2.87]	0.035	0.012		0.080	0.061	2[0.08 - 0.18]	2.560	1.032	3[1.04 - 3.01]
Grenslandsfjord (st. 71B)	13(10-3)	1.600	0.941	13[0.16 - 3.71]	0.035	0.018	3[0.08 - 0.13]	0.040	0.131	6[0.09 - 0.43]	1.780	1.015	13[0.3 - 4.21]
Kristiansand harbour (st. 13B)	14(7-2)	2.965	7.271	14[0.73 - 29.7]	0.105	0.190	8[0.08 - 0.68]	0.033	0.080	3[0.12 - 0.31]	3.180	7.421	14[0.87 - 30.45]
Inner Sørfjord (st. 53B)	9(8-4)	2.420	0.888	9[1.32 - 3.94]	0.040	0.278	5[0.02 - 0.87]	0.100	0.058	9[0.08 - 0.22]	2.590	1.128	9[1.43 - 5.03]
Bømlo north (st. 23B)	14(4-2)	0.980	0.928	14[0.17 - 3.07]	0.035	0.026		0.053	0.051	4[0.08 - 0.25]	1.220	0.938	14[0.29 - 3.22]
Inner Trondheimsfjord (st. 80B)	15	3.630	3.257	15[1.16 - 11.7]	0.100	0.094	10[0.08 - 0.4]	0.200	0.168	10[0.08 - 0.54]	4.070	3.424	15[1.38 - 12.4]
Lofoten, Skrova (st. 98B1)	8(7-2)	0.465	0.398	8[0.18 - 1.45]	0.035	0.140	2[0.13 - 0.47]	0.135	0.119	5[0.11 - 0.35]	0.870	0.440	8[0.38 - 1.68]
Tromsø harbour (st. 43B2)	15	2.620	5.295	15[0.9 - 22.7]	0.100	1.749	11[0.08 - 6.65]	0.040	0.129	4[0.13 - 0.5]	2.940	7.003	15[1.06 - 29.59]

Considering only α–HBCD, which was the most dominant diastereomer (see ***Table 14***), concentrations in cod liver were significantly higher in the Inner Oslofjord than for nine of the other areas (Tukey-Kramer HSD test) (***Figure 35***). Individual variation was high in cod from the Kristiansand, Tromsø and Oslofjord harbour areas (stations 13B, 43B2 and 30B, respectively). Furthermore, cod liver showed about-100 times higher concentrations than in blue mussel on a wet weight basis (compare ***Figure 35*** and ***Figure 36***). The difference was smaller on a lipid basis. There are some indications of biomagnification for specific diastereomers of HBCD (Haukås 2009).

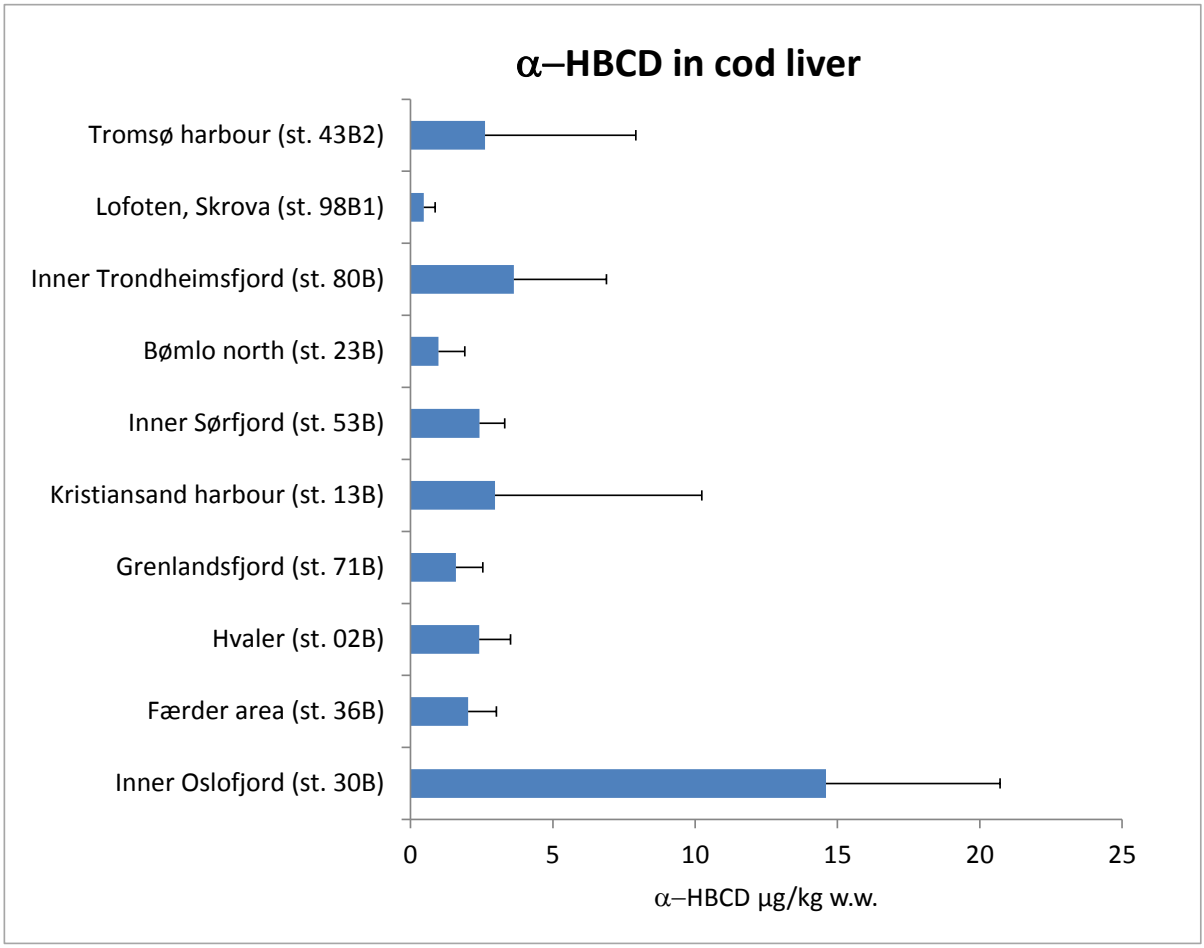

Figure 35*. Mean concentration (µg/kg w.w.) of α–HBCD in cod liver in 2014. The error bar indicates one standard deviation above the mean.*

In 2014 the concentration of α–HBCD in liver of flounder from Bestumkilen in the Inner Oslofjord ranged from <0.011 to 0.86 µg/kg w.w. (Ruus *et al*. 2015).

Blue mussel from Bergen harbour (Nordnes, st. I241) had concentrations of α–HBCD that were significantly higher than for all the other stations (Tukey-Kramer HSD test, see also **Figure 36**). The levels found in blue mussel from Gressholmen (st. 30A) in the Inner Oslofjord were significantly higher than for the eight other stations (Tukey-Kramer HSD test, see also **Figure 36**). The same level of contamination was found on two other stations in the Inner Oslofjord in 2013 (Ruus *et al*. 2014). In recent years, there has been a clear reduction of HBCD-concentrations in freshwater fish from Mjøsa (Løvik *et al*. 2015).

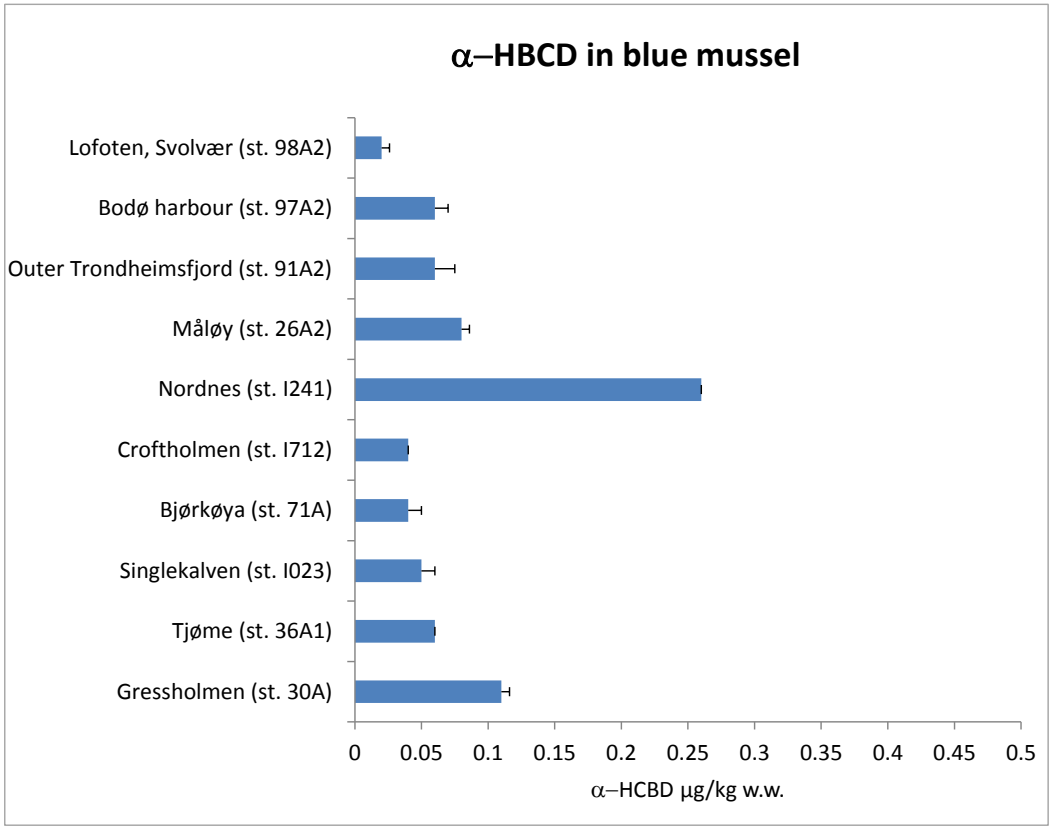

***Figure 36**. Mean concentration (µg/kg w.w.) of α–HBCD in blue mussel in 2014. The error bar indicates one standard deviation above the mean.*

General, large scale

The discharges of HBCD to water from land-based industries showed a decrease from 2004 (12.90 kg HBCD/year) to 2005 (1.50 kg HBCD/year) (***Figure 37***).

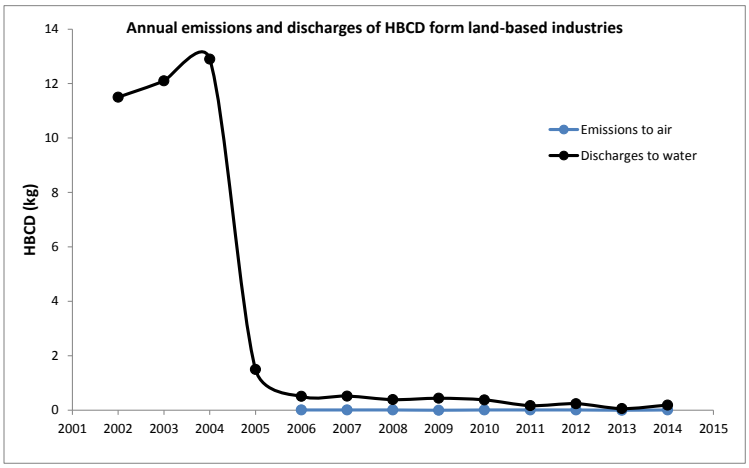

***Figure 37**. Annual emissions of HBCD to air and discharges to water from land-based industries in the period 2002-2014 (data from www.norskeutslipp.no). No data for emissions to air are reported for 2002-2005. Note that this category excludes emissions and discharges from municipal treatment plants, land runoff, transportation and offshore industry.*

3.3.2 Chlorinated paraffins (SCCP and MCCP)

Chlorinated paraffins are complex mixtures of polychlorinated organic coumpounds. They are mainly used in metal working fluids, sealants, as flame-retardants in rubbers and textiles, in leather processing and in paints and coatings. Their persistence, bioaccumulation, potential for long-ranged environmental transport and toxicity mean that they may have harmful environmental effects at a global level. Chlorinated paraffins were analysed in cod liver at 10 stations and in blue mussel at 10 stations.

Chlorinated paraffins are subdivided according to their carbon chain length into short chain chlorinated paraffins (SCCPs, C_{10-13}) and medium chain chlorinated paraffins (MCCPs, C_{14-17}). There is an EQS for SCCP in water but not for biota (2013/39/EU). SCCPs and MCCPs are classified as persistent with a high potential for bioaccumulation, and are toxic to aquatic organisms. Use and production of SCCPs are prohibited in Norway. However emission from old- or imported products cannot be excluded. MCCPs are largely used as a flame retardant and as an additive to plastics, such as PVC, to increase flexibility. To a lesser degree MCCPs are used as a lubricant in machinery for manufacturing metal products. MCCPs are mainly released to water in effluent from industry using them as metal working fluids. MCCP is used to a limited extent in Norwegian production, but may be found in imported products. There is, however, considerable uncertainty about the quantities in products used in Norway. There is an indication that the discharges from the use of imported products have been reduced by 39 % from 1995 to 2010[1].

The concentration of SCCP in cod liver ranged from 19.9 to 98.9 µg/kg w.w., with highest concentration in cod from Bømlo (st. 23B, ***Figure 38***, ***Table 15***). Reth *et al.* (2005) found similar levels of SCCP in cod from the North Sea and the Baltic Sea (19 to 143 ng/g w.w.). Concentrations observed in samples from urban areas are frequently higher than from other more sparsely populated areas.

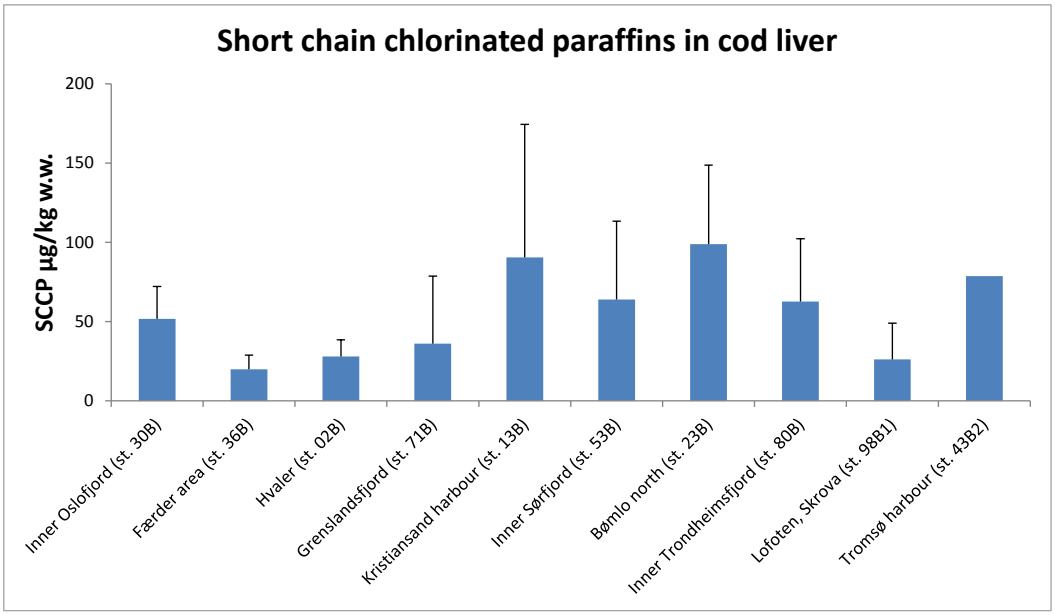

Figure 38. *Median concentration (µg/kg w.w.) of short chain chlorinated paraffins (SCCP) in cod liver in 2014. The error bar indicates one standard deviation above the median.*

[1] http://www.miljostatus.no/Tema/Kjemikalier/Noen-farlige-kjemikalier/Klorerte-parafiner/

In 2014 the concentration of SCCP in liver of flounder from Bestumkilen in the Inner Oslofjord ranged from 4.9 to 48.1 µg/kg w.w. (Ruus *et al*. 2015). Polychaetes from the Inner Oslofjord had SCCP-concentrations of 1.5 to 22.0 µg/kg w.w.

The concentration of SCCP in blue mussel ranged from 0.55 to 8.39 µg/kg w.w. in this study and the highest concentration was found in the samples from Bergen harbour (Nordnes) (st. I241, ***Figure 39***).

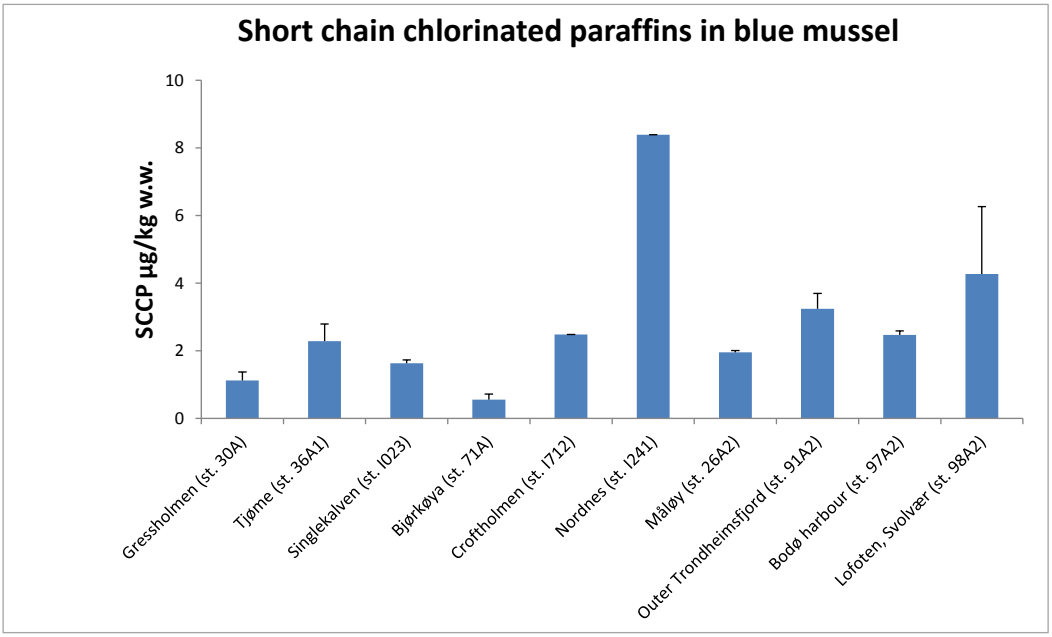

Figure 39. *Median concentration (µg/kg w.w.) of short chain chlorinated paraffins (SCCP) in blue mussel in 2014. The error bar indicates one standard deviation above the median.*

Cod from Bømlo (st. 23B) and the Inner Trondheimsfjord (st. 80B) had highest concentration of MCCPs with 262.5 and 262 µg/kg w.w. There was high individual variation (***Figure 40***, ***Table 15***). Flounder from Bestumkilen in the Inner Oslofjord had concentrations of MCCP in liver in the range 0.6 to 19.9 µg/kg w.w., and prawns had concentrations of MCCP in the range 0.1 to 1.6 µg/kg w.w. (Ruus *et al*. 2015).

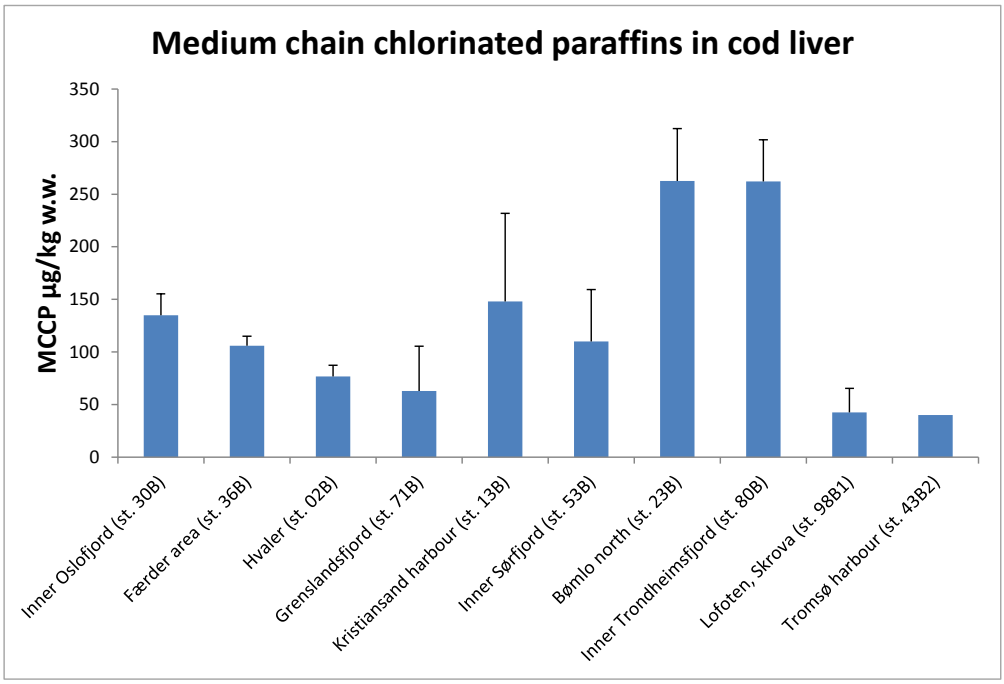

***Figure 40**. Median concentration (µg/kg w.w.) of medium chain chlorinated paraffins (MCCPs) in cod liver in 2014. The error bar indicates one standard deviation above the median.*

The concentrations of MCCPs in blue mussel were lower than in cod, and ranged from 3.3 to 54.8 µg/kg w.w. Blue mussel from Croftholmen (st. I712) in the Grenlandsfjord had the highest concentration of MCCPs (***Figure 41***).

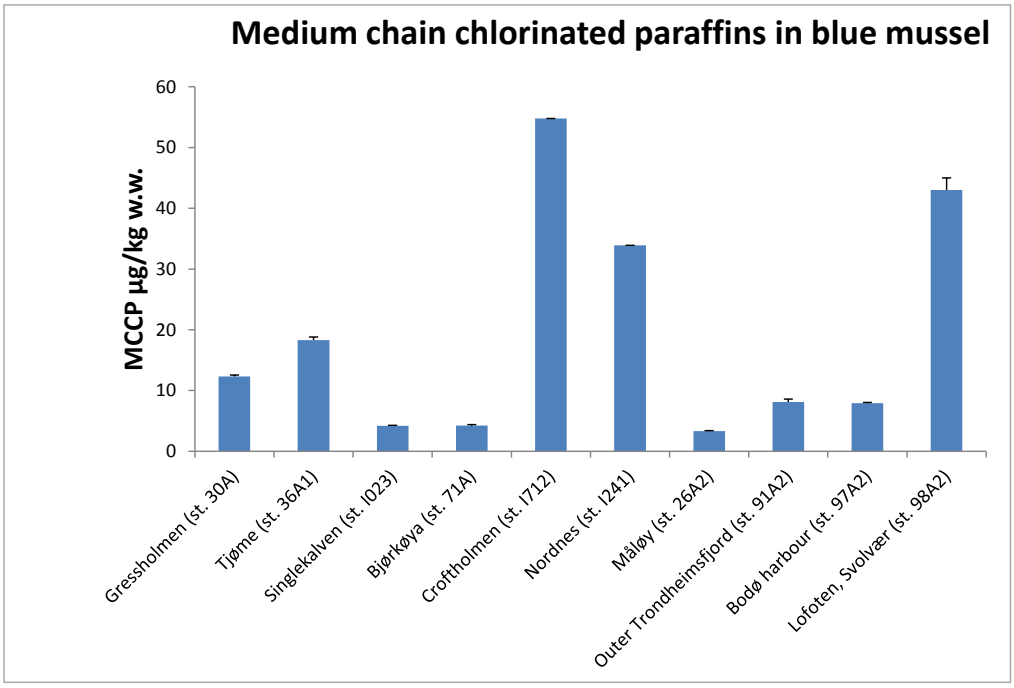

***Figure 41**. Median concentration (µg/kg w.w.) of medium chain chlorinated paraffins (MCCPs) in blue mussel in 2014. The error bar indicates one standard deviation above the median.*

***Table 15**. Median concentrations (µg/kg w.w.) with standard deviation of* short chain chlorinated paraffins (*SCCPs) and* medium chain chlorinated paraffins (*MCCPs) in blue mussel and cod in 2014. Count indicates number of samples analysed. The first number within the parentheses indicates the number of pooled samples included. The second number within the parentheses indicates for mussels the total number of individuals used in all pooled samples and for cod the number individuals in each pooled sample. Shaded values are below one half the detection limit, and indicate that over half of the values are below this limit. The standard deviation is based on all values. Caution should be used when comparing such values because the limit of detection can vary both within and among samples. Detectable data information (D.d.i.) indicates the number of data above the limit of detection (if any) and the numbers within the square brackets indicate the minimum and maximum values in this category.*

Component	Count	SCCP			MCCP		
Species and sampling locality	2014	Med.	S.d.	D.d.i	Med.	S.d.	D.d.i
Blue mussel							
Gressholmen (st. 30A)	3(3-50)	1.120	0.251	3[0.83 - 1.33]	12.300	3.681	3[6.08 - 12.6]
Tjøme (st. 36A1)	3(3-50)	2.28	0.51	3[1.76 - 2.78]	18.3	2.787	3[13.8 - 18.9]
Singlekalven (st. I023)	3(3-50)	1.63	0.098	3[1.46 - 1.63]	4.17	0.221	3[3.87 - 4.3]
Bjørkøya (st. 71A)	3(2-50)	0.55	0.167	3[0.33 - 0.657]	4.21	0.605	3[3.48 - 4.68]
Croftholmen (st. I712)	1(1-50)	2.48	0	1[2.48]	54.8	0	1[54.8]
Nordnes (st. I241)	1(1-50)	8.39	0	1[8.39]	33.9	0	1[33.9]
Måløy (st. 26A2)	3(3-50)	1.95	0.055	3[1.94 - 2.04]	3.33	0.188	3[3.3 - 3.64]
Outer Trondheimsfjord (st. 91A2)	3(3-50)	3.24	0.459	3[3.14 - 3.98]	8.13	0.857	3[6.66 - 8.16]
Bodø harbour (st. 97A2)	3(3-50)	2.47	0.115	3[2.42 - 2.64]	7.92	1.066	3[6.39 - 8.44]
Lofoten, Svolvær (st. 98A2)	3(3-50)	4.27	1.993	3[3.78 - 7.45]	43	27.38	3[30.2 - 82.7]
Cod, liver							
Inner Oslofjord (st. 30B)	13(8-3)	51.8	20.37	13[37.7 - 109]	135	68.22	13[65 - 309]
Færder area (st. 36B)	15(12-3)	19.9	9.054	15[14.8 - 47.7]	106	93.97	15[65.5 - 330]
Hvaler (st. 02B)	3(3-3)	28	10.55	3[12 - 31.9]	76.8	55.02	3[8.19 - 117]
Grenslandsfjord (st. 71B)	13(10-3)	36.1	42.6	13[10.7 - 181]	63	33.04	13[25.1 - 127]
Kristiansand harbour (st. 13B)	14(7-2)	90.55	83.82	14[37.8 - 339]	148	378.2	14[16.6 - 1100]
Inner Sørfjord (st. 53B)	9(8-4)	64	49.37	9[32.6 - 176]	110	89.27	9[65.3 - 360]
Bømlo north (st. 23B)	14(4-2)	98.85	49.87	14[17 - 194]	262.5	177.1	14[69.1 - 643]
Inner Trondheimsfjord (st. 80B)	15	62.7	39.62	15[41.9 - 180]	262	99.05	15[116 - 446]
Lofoten, Skrova (st. 98B1)	8(7-2)	26.2	22.91	8[19.6 - 90]	42.5	15.32	8[19 - 71]
Tromsø harbour (st. 43B2)	15	78.7	23.37	15[61.1 - 159]	40.2	19.77	15[15.8 - 88.6]

Supplementary analyses of cod liver samples from 1990-2014

Supplementary analyses of SCCPs and MCCPs in cod liver were carried out for two stations, Bømlo (st. 23B) and Inner Sørfjord (st. 53B), for selected years during the period (1990-2013) (**Figure 42**). The concentrations of SCCPs and MCCPs in cod from the Inner Sørfjord were much higher in 1990 and in 2012 than in the last two year. A significant downward trend was registered for SCCP in cod liver from the Inner Sørfjord.

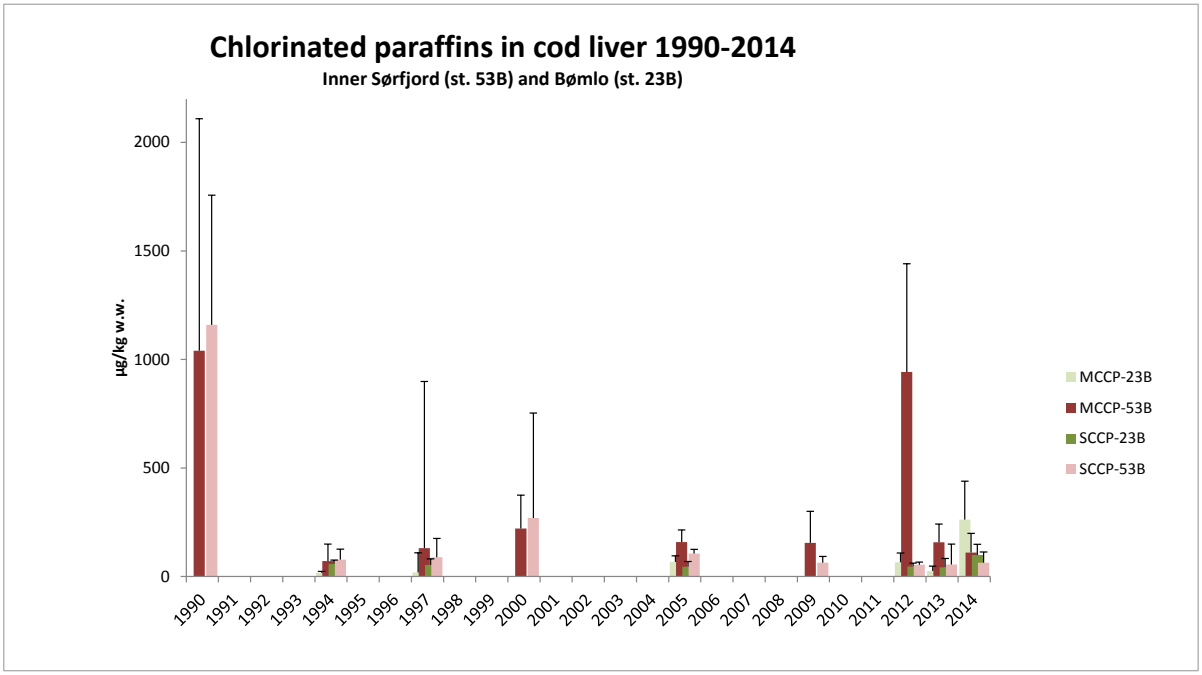

***Figure 42**. Mean concentration (µg/kg w.w.) of SCCPs and MCCPs in cod liver from Bømlo (st.23B) and Inner Sørfjord (st. 53B) during 1990-2014. The error bars indicates one standard deviation above the median.*

3.3.3 Organophosphorus flame retardants (PFRs)

Organophosphorus flame retardants (PFRs) were analysed in cod liver at 10 stations and in blue mussel at 10 stations.

Many of the PFRs are persistent and bioaccumulate. Some of the PFRs are classified as hazardous to the environment. These include: tri(2-chloroethyl)phosphate (TCEP), 2-ethylhexyl-di-phenylphosphate (EHDPP), tri(1,3-dichloro-2-propyl)phosphate (TDCP), tricresyl phosphate (TCrP) and triphenylphosphate (TPhP). TCEP is classified as harmful to reproduction. Some of the PFRs are suspected to be carcinogenic (TBP, TCEP and TDCP). TCEP is on the priority list of Norwegian Environment Agency[1]. These substances are used *inter alia* as a softener in vinyl plastics, as a flame retardant and as an additive in hydraulic fluids (van der Veen & de Boer 2012). However there is no registered used of these substances and there is considerable uncertainty as to the quantities in products in Norway.

The concentrations of PFRs were low; most of the results were below the detection limits (***Table 16***). The detection limits were lower than for the analysis for the 2013 investigation (Green *et al.* 2014). It should be noted that PFRs are generally difficult to separate from the lipid portion of a sample before chemical analysis even following extra clean-up, as was the case in this study. The difficulty to separate PFRs can lead to analytical interference and often result in a higher detection limit. This problem can vary from sample to sample. Hence more variable and higher detection limits can be found when compared to other contaminant groups such as PCBs, PBDEs (***Table 12***) or PFAS (***Table 13***).

[1] *http://www.miljostatus.no/Tema/Kjemikalier/Kjemikalielister/Prioritetslisten/*

***Table 16**. Median concentrations (µg/kg w.w.) with standard deviation of* organophosphorus flame retardants (*PFRs) in blue mussel and cod liver in 2014. Count indicates number of samples analysed. The first number within the parentheses indicates the number of pooled samples included. The second number within the parentheses indicates for mussels the total number of individuals used in all pooled samples and for cod the number individuals in each pooled sample. Shaded values are below one half the detection limit, and indicate that over half of the values are below this limit. The standard deviation is based on all values. Caution should be used when comparing such values because the limit of detection can vary both within and among samples. Detectable data information (D.d.i.) indicates the number of data above the limit of detection (if any) and the numbers within the square brackets indicate the minimum and maximum values in this category. (See Appendix B for description of chemical codes.)*

Component	Count	EHDPP			TBEP			TBP			TCEP			TCPP			TCRP		
Species and sampling locality	2014	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.
Blue mussel																			
Gressholmen (st. 30A)	3(3-50)	0.2	0.0		0.2	0.0		0.2	0.0		0.1	0.0		0.8	0.2	3[0.536 - 0.886]	0.5	0.0	
Tjøme (st. 36A1)	3(3-50)	0.3	0.1		0.3	0.1		0.3	0.1		0.2	0.0		0.3	0.0	1[0.577]	0.7	0.1	
Singlekalven (st. I023)	3(3-50)	0.2	0.5	1[1.22]	0.2	0.0		0.2	0.0		0.1	0.0		0.5	0.8	3[0.494 - 1.85]	0.4	0.1	
Bjørkøya (st. 71A)	3(2-50)	0.2	0.0		0.2	0.0		0.2	0.0		0.1	0.0		0.2	0.0		0.4	0.1	
Croftholmen (st. I712)	1(1-50)	0.3	0.0		0.1	0.0		0.2	0.0		0.1	0.0		1.1	0.0	1[1.07]	0.4	0.0	
Nordnes (st. I241)	1(1-50)	0.4	0.0		0.4	0.0		0.4	0.0		0.2	0.0		3.9	0.0	1[3.92]	0.9	0.0	
Måløy (st. 26A2)	3(3-50)	0.3	0.0		0.3	0.0		0.3	0.0		0.2	0.0		0.9	0.0	3[0.83 - 0.922]	0.6	0.0	
Outer Trondheimsfjord (st. 91A2)	3(3-50)	0.4	0.0		0.3	0.0		0.3	0.0		0.2	0.0		0.9	0.0	2[1.4 - 1.73]	0.6	0.0	
Bodø harbour (st. 97A2)	3(3-50)	0.2	0.1		0.4	0.0		0.4	0.0		0.2	0.0		1.4	0.5	3[0.842 - 1.56]	0.7	0.7	
Lofoten, Svolvær (st. 98A2)	3(3-50)	0.3	0.0		0.2	0.1		0.2	0.1		0.1	0.0		1.0	0.4	3[0.634 - 0.867]	0.5	0.1	
Cod, liver																			
Inner Oslofjord (st. 30B)	13(8-3)	4.8	17.3	2[48.5 - 62.1]	4.8	0.6		4.8	0.6		2.4	0.3		4.8	0.6		9.6	1.2	
Færder area (st. 36B)	15(12-3)	3.7	1.3		3.7	1.3		3.7	1.3		1.9	0.7		3.7	1.3		7.5	2.7	
Hvaler (st. 02B)	3(3-3)	5.7	0.4		5.7	0.4		5.7	0.4		2.8	0.2		5.7	0.4		11.3	0.8	
Grenslandsfjord (st. 71B)	13(10-3)	4.6	1.6		4.6	1.6		4.6	1.6		2.3	0.8		4.6	1.6		9.2	3.3	
Kristiansand harbour (st. 13B)	14(7-2)*	6.5	9.9	1[27]	6.4	0.8		6.4	0.8		3.2	0.4		6.4	0.8		12.8	4.4	
Inner Sørfjord (st. 53B)	7(6-3)	3.3	3.9		3.3	3.4		3.3	3.4		1.6	1.7		3.3	3.4		6.5	6.7	
Bømlo north (st. 23B)	14(4-2)	4.9	2.1		4.9	1.3		4.9	1.3		2.4	0.6		4.9	1.3		9.7	2.5	
Inner Trondheimsfjord (st. 80B)	15	6.3	8.1		6.3	8.1		6.3	8.1		3.2	4.1		6.6	8.6		12.7	16.2	
Lofoten, Skrova (st. 98B1)	8(7-2)	6.2	1.7		6.2	1.7		6.2	1.7		3.1	0.8		6.2	1.7		12.5	3.3	
Tromsø harbour (st. 43B2)	15	6.4	1.9		6.4	1.9		6.4	1.9		3.2	0.9		6.4	1.9		12.9	3.7	

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*) Count 13(6-2) for TDCP and TEHP

Table 16. *(cont.)*

Component	Count	TDCP			TEHP			TIBP			TOCRP			TPHP		
Species and sampling locality	2014	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.
Blue mussel																
Gressholmen (st. 30A)	3(3-50)	0.1	0.0	3[0.131 - 0.143]	0.1	0.0		0.5	0.1		0.1	0.1		0.1	0.0	
Tjøme (st. 36A1)	3(3-50)	0.1	0.0		0.2	0.0		0.7	0.2		0.2	0.0		0.2	0.0	
Singlekalven (st. I023)	3(3-50)	0.0	0.0		0.1	0.0		0.4	0.1		0.1	0.0		0.1	0.0	
Bjørkøya (st. 71A)	3(2-50)	0.0	0.0		0.1	0.0		0.4	0.0		0.1	0.0		0.1	0.0	
Croftholmen (st. I712)	1(1-50)	0.0	0.0		0.1	0.0		0.2	0.0		0.1	0.0		0.1	0.0	
Nordnes (st. I241)	1(1-50)	0.1	0.0		0.2	0.0		0.9	0.0		0.2	0.0		0.7	0.0	1[0.658]
Måløy (st. 26A2)	3(3-50)	0.1	0.0		0.2	0.0		0.6	0.1		0.2	0.0		0.2	0.0	
Outer Trondheimsfjord (st. 91A2)	3(3-50)	0.1	0.0		0.2	0.0		0.6	0.1		0.2	0.0		0.2	0.0	
Bodø harbour (st. 97A2)	3(3-50)	0.1	0.0		0.2	0.0		0.7	0.0		0.2	0.0		0.2	0.0	
Lofoten, Svolvær (st. 98A2)	3(3-50)	0.0	0.0		0.1	0.0		0.5	0.1		0.1	0.0		0.1	0.0	
Cod, liver																
Inner Oslofjord (st. 30B)	13(8-3)	1.0	0.1		2.4	0.3		9.6	1.2		2.4	0.3		2.4	0.3	
Færder area (st. 36B)	15(12-3)	0.7	0.3		1.9	0.7		7.5	2.7		1.9	0.7		1.9	0.7	
Hvaler (st. 02B)	3(3-3)	1.1	0.1		2.8	0.2		11.3	0.8		2.8	0.2		2.8	0.2	
Grenslandsfjord (st. 71B)	13(10-3)	1.0	2.0		2.3	0.8		9.2	3.3		2.3	0.8		2.3	0.8	
Kristiansand harbour (st. 13B)	14(7-2)*	1.3	0.2		3.2	0.4		12.8	1.5		3.2	1.1		3.2	1.1	
Inner Sørfjord (st. 53B)	7(6-3)	1.0	0.8		3.4	6.2		7.7	6.6		1.6	1.7		1.6	1.7	
Bømlo north (st. 23B)	14(4-2)	1.0	0.4		2.4	0.6		9.7	1.9		2.4	0.6		2.4	0.6	
Inner Trondheimsfjord (st. 80B)	15	1.3	1.6		3.2	4.1		12.7	16.2		3.2	4.1		3.2	4.1	
Lofoten, Skrova (st. 98B1)	8(7-2)	1.2	0.3		3.1	0.8		12.5	3.3		3.1	0.8		3.1	0.8	
Tromsø harbour (st. 43B2)	15	1.3	0.5		3.2	0.9		12.9	3.7		3.2	0.9		3.2	0.9	

*) Count 13(6-2) for TDCP and TEHP

3.3.4 Bisphenol A (BPA)

Bisphenol A (BPA) was analysed in cod liver from nine locations and in blue mussel from 10 stations.

Bisphenol A is derived from epoxy resins and polycarbonate plastics (Belfroid *et al*. 2002). Bisphenol A has been produced in large quantities world-wide and therefore can be considered ubiquitous (Flint *et al*. 2012). It is an endocrine disruptor which can mimic oestrogen, and is also carcinogenic. Studies have shown that BPA can affect growth, reproduction and development in aquatic organisms. Bisphenol A is on the priority list of Norwegian Environment Agency[1].

Most of the median concentrations of bisphenol A found in blue mussel were below the detection limit, and only three samples had concentrations above the detection limit (***Table 17***). Hence, no conclusion can be drawn regarding possible differences between stations. Liver of flounder from Bestumkilen in the Inner Oslofjord had concentrations of BPA that were higher, with a median concentration of 15.7 µg/kg w.w. (Ruus *et al*. 2015).

The concentrations of bisphenol A in cod liver were low, and mostly below the detection limit. In cod from the Inner Oslofjord (st. 30B) and Kristiansand harbour (st. 13B), the median concentrations of bisphenol A were above the detection limits (***Table 17***). Detectable concentrations found in liver samples varied between 1.0 to 1.8 µg/kg w.w.

In a recent study bisphenol A was detected in 75 % of the cod liver samples from Byfjord, Bergen, in the concentration range <4 – 46.3 ng/g w.w. (Langford *et al*. 2012).

[1] *http://www.miljostatus.no/Tema/Kjemikalier/Kjemikalielister/Prioritetslisten/*

***Table 17**. Median concentrations (µg/kg w.w.) with standard deviation of bisphenol A (BPA) in blue mussel and cod liver in 2013. The first number within the parentheses indicates the number of pooled samples included. The second number within the parentheses indicates for mussels the total number of individuals used in all pooled samples and for cod the number individuals in each pooled sample. Shaded values are below one half the detection limits, and indicate that over half of the values are below this limit. The standard deviation is based on all values. Caution should be used when comparing such values because the limit of detection can vary both within and among samples. Detectable data information (D.d.i.) indicates the number of data above the limit of detection (if any) and the numbers within the square brackets indicate the minimum and maximum values in this category.*

Component	Count	BPA		
Species and sampling locality	2014	Med.	S.d.	D.d.i.
Blue mussel				
Gressholmen (st. 30A)	3(3-50)	0.5	0.0	
Tjøme (st. 36A1)	3(3-50)	0.5	0.0	
Singlekalven (st. I023)	3(3-50)	0.5	0.0	
Bjørkøya (st. 71A)	3(2-50)	0.5	0.0	
Croftholmen (st. I712)	1(1-50)	1.0	0.0	1[1]
Nordnes (st. I241)	1(1-50)	0.5	0.0	
Måløy (st. 26A2)	3(3-50)	0.5	0.0	
Outer Trondheimsfjord (st. 91A2)	3(3-50)	0.5	0.2	1[1.3]
Bodø harbour (st. 97A2)	3(3-50)	0.5	0.7	1[2.2]
Lofoten, Svolvær (st. 98A2)	3(3-50)	0.5	0.0	
Cod, liver				
Inner Oslofjord (st. 30B)	13(8-3)	1.0	0.0	8[1 - 1]
Færder area (st. 36B)	15(12-3)	0.5	0.1	1[1.3]
Hvaler (st. 02B)	3(3-3)	0.5	0.0	
Grenslandsfjord (st. 71B)	13(10-3)	0.5	0.0	
Kristiansand harbour (st. 13B)	14(7-2)	1.0	0.0	13[1 - 1]
Inner Sørfjord (st. 53B)	9(8-4)	0.5	0.3	2[1.4 - 1.8]
Bømlo north (st. 23B)	14(4-2)	0.5	0.1	1[1.3]
Inner Trondheimsfjord (st. 80B)	15	0.5	0.1	1[1.3]
Tromsø harbour (st. 43B2)	15	0.5	0.0	

3.3.5 Tetrabrombisphenol A (TBBPA)

Tetrabrombisphenol A (TBBPA) was analysed in cod liver at nine stations and in blue mussel at 10 stations.

TBBPA is a polybrominated flame retardant and is an endocrine disruptor and immunotoxicant.

Concentrations of TBBPA found in cod liver and blue mussel were below the limit of detection for all samples except for one (***Table 18***). The exception was for liver in one cod from the Inner Oslofjord (st. 30B) that had a concentration of 48.6 µg/kg w.w.

Table 18. *Concentration (µg/kg w.w.) of TBBPA in blue mussel and cod liver. The shaded areas indicate value below the detection limit and that the values shown are one half of the detection limit.*

Component	Count	TBBPA		
Species and sampling locality	2014	Med.	S.d.	D.d.i.
Blue mussel				
Gressholmen (st. 30A)	3(3-50)	0.0	0.1	
Tjøme (st. 36A1)	3(3-50)	0.0	0.1	
Singlekalven (st. I023)	3(3-50)	0.0	0.0	
Bjørkøya (st. 71A)	3(2-50)	0.0	0.0	
Croftholmen (st. I712)	1(1-50)	0.0	0.0	
Nordnes (st. I241)	1(1-50)	0.0	0.0	
Måløy (st. 26A2)	3(3-50)	0.0	0.0	
Outer Trondheimsfjord (st. 91A2)	3(3-50)	0.0	0.0	
Bodø harbour (st. 97A2)	3(3-50)	0.0	0.0	
Lofoten, Svolvær (st. 98A2)	3(3-50)	0.0	0.0	
Cod, liver				
Inner Oslofjord (st. 30B)	13(8-3)	0.2	13.4	1[48.6]
Færder area (st. 36B)	15(12-3)	0.2	0.0	
Hvaler (st. 02B)	3(3-3)	0.2	0.0	
Grenslandsfjord (st. 71B)	13(10-3)	0.2	0.0	
Kristiansand harbour (st. 13B)	14(7-2)	0.2	0.1	
Inner Sørfjord (st. 53B)	9(8-4)	0.1	0.1	
Bømlo north (st. 23B)	14(4-2)	0.2	0.1	
Inner Trondheimsfjord (st. 80B)	15	0.2	0.0	
Tromsø harbour (st. 43B2)	15	0.2	0.0	

3.3.6 Alkylphenols

These substances are used in manufacturing antioxidants, lubricating oil additives, household detergents. They are also precursors for commercially important surfactants. Nonylphenol and octylphenol are two alklyphenols and are on the EQSD list of priority hazardous substances but have no EQS for biota. They were analysed in 2012 samples and for the first time as part of the MILKYS programme. In Norway it has since 2005 been prohibited to produce, import, export, sell or use nonylphenols, octylphenols or their etoxsilates with the exception of paints, varnish, lubricants and finished products.

Alkylphenols were analysed in cod liver from nine locations and in blue mussel from 10 stations. The concentrations in both cod liver and blue mussel were very low. All concentrations were below the detection limits (***Table 19***). Hence, no conclusion can be drawn regarding possible differences between stations.

Analyses of blue mussel and cod liver samples from 2013

Analyses of alkylphenols were also carried out on 2013-samples (***Table 20***). Detectable concentrations of alkylphenols were found in cod liver and blue mussel. Cod from the Inner Oslofjord had highest concentrations of alkylphenols, with 4-t-nonylphenol in the range 16.1 to 49.8 µg/kg w.w. In blue mussel, only 4-t-nonylphenol and 4-n-octylphenol were found with concentrations above the detection limits. Blue mussel from Bergen harbour (Nordnes, st. I241) and Tjøme (st. 36A1) in the Outer Oslofjord had the highest concentrations of alkylphenols. High concentrations were not consistant with proximity to urban areas.

General, large scale

The discharges from land-based industries to water varied between 4730 kg phenols in 2008 to 1550 kg phenols in 2014 in the period 2002-2014 (***Figure 43***).

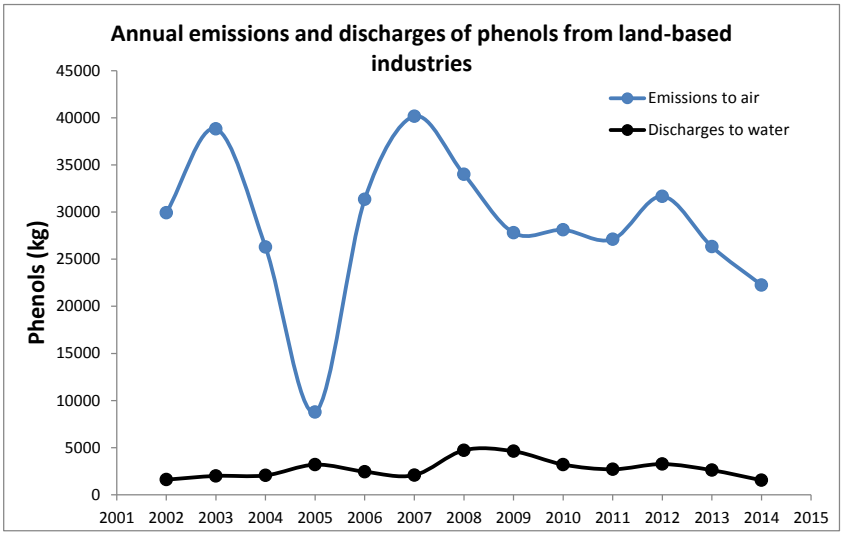


Figure 43. *Annual emissions of phenols to air and discharges to water from land-based industries in the period 2002-2014 (data from* www.norskeutslipp.no*). Note that this category excludes emissions and discharges from municipal treatment plants, land runoff, transportation and offshore industry.*

***Table 19**. Median concentrations (µg/kg w.w.) with standard deviation of alkylphenols in blue mussel and cod liver in 2014. Count indicates number of samples analysed. The first number within the parentheses indicates the number of pooled samples included. The second number within the parentheses indicates for mussels the total number of individuals used in all pooled samples and for cod the number individuals in each pooled sample. Shaded values are below one half the detection limits, and indicate that over half of the values are below this limit. The standard deviation is based on all values. Caution should be used when comparing such values because the limit of detection can vary both within and among samples. Detectable data information (D.d.i.) indicates the number of data above the limit of detection (if any) and the numbers within the square brackets indicate the minimum and maximum values in this category. (See Appendix B for description of chemical codes.)*

Component	Count	4-n-NP			4-n-OP			4-t-NP			4-t-OP		
Species and sampling locality	2014	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.
Blue mussel													
Gressholmen (st. 30A)	3(3-50)	0.5	0.00		0.5	0.00		10.0	0.00		1.0	0.00	
Tjøme (st. 36A1)	3(3-50)	0.5	0.00		0.5	0.58		10.0	0.00		1.0	0.00	
Singlekalven (st. I023)	3(3-50)	0.5	0.00		0.5	0.58		10.0	0.00		0.5	0.58	
Bjørkøya (st. 71A)	3(2-50)	0.5	0.00		1.0	0.00		10.0	0.00		0.5	0.00	
Croftholmen (st. I712)	1(1-50)	1.0	0.00		1.0	0.00		25.0	0.00		1.0	0.00	
Nordnes (st. I241)	1(1-50)	0.5	0.00		0.5	0.00		10.0	0.00		0.5	0.00	
Måløy (st. 26A2)	3(3-50)	0.5	0.00		0.5	0.58		10.0	0.00		0.5	0.00	
Outer Trondheimsfjord (st. 91A2)	3(3-50)	0.5	0.00		0.5	0.58		10.0	0.00		0.5	0.58	
Bodø harbour (st. 97A2)	3(3-50)	0.5	0.00		0.5	0.58		10.0	10.39		0.5	0.00	
Lofoten, Svolvær (st. 98A2)	3(3-50)	0.5	0.00		1.5	1.16		10.0	0.00		0.5	0.00	
Cod liver													
Inner Oslofjord (st. 30B)	13(8-3)	5.0	4.80		5.0	7.68		50.0	0.00		5.0	2.77	
Færder area (st. 36B)	15(12-3)	10.0	16.82		15.0	17.81		25.0	25.82		10.0	8.84	
Hvaler (st. 02B)	3(3-3)	5.0	0.00		5.0	0.00		25.0	0.00		5.0	0.00	
Grenslandsfjord (st. 71B)	13(10-3)	5.0	0.00		10.0	0.00		50.0	0.00		5.0	0.00	
Kristiansand harbour (st. 13B)	14(7-2)*	10.0	5.61		5.0	6.02		50.0	0.00		5.0	4.99	
Inner Sørfjord (st. 53B)	7(6-3)	5.0	0.00		10.0	0.00		50.0	0.00		5.0	0.00	
Bømlo north (st. 23B)	14(4-2)	5.0	0.00		10.0	0.00		50.0	0.00		5.0	0.00	
Inner Trondheimsfjord (st. 80B)	15	5.0	0.00		10.0	0.00		50.0	0.00		5.0	0.00	
Tromsø harbour (st. 43B2)	15	10.0	3.52		5.0	4.14		50.0	0.00		5.0	3.52	

*) Count 12(5-2) for 4-t-NP

***Table 20**. Median concentrations (µg/kg w.w.) with standard deviation of alkylphenols in blue mussel and cod liver in supplementary analsyses of 2013 samples (indicated with an asterisk *). Count indicates number of samples analysed. The first number within the parentheses indicates the number of pooled samples included. The second number within the parentheses indicates for mussels the total number of individuals used in all pooled samples and for cod the number individuals in each pooled sample. Shaded values are below one half the detection limits, and indicate that over half of the values are below this limit. The standard deviation is based on all values. Caution should be used when comparing such values because the limit of detection can vary both within and among samples. Detectable data information (D.d.i.) indicates the number of data above the limit of detection (if any) and the numbers within the square brackets indicate the minimum and maximum values in this category. (See Appendix B for description of chemical codes.)*

Component	Count	4-n-NP			4-n-OP			4-t-NP			4-t-OP		
Species and sampling locality	2013	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.
Blue mussel													
Akershuskaia (st. I301)*	2(2-50)	0.5	0.00		0.8	0.71		25.0	0.00		0.8	0.71	
Gressholmen (st. 30A)	3(3-204)	1.0	0.00		1.0	0.00		13.0	2.55	3[12.1-16.9]	1.0	0.00	
Gåsøya (st. I304)*	2(2-15)	0.5	0.00		0.8	0.71		25.0	0.00		0.8	0.71	
Solbergstrand (st. 31A)*	1(1-48)	0.5	0.00		0.5	0.00		25.0	0.00		1.0	0.00	
Mølen (st. 35A)*	3(3-88)	0.5	0.00		0.5	0.58		25.0	0.00		0.5	0.58	
Tjøme (st. 36A1)	3(3-50)	1.0	0.00		2.3	0.20	3[2.23-2.61]	19.3	8.87	3[17.2-33.5]	1.0	0.00	
Singlekalven (st. I023)	3(3-33)	1.0	0.00		1.0	0.00		11.8	2.22	3[9.76-14.2]	1.0	0.00	
Bjørkøya (st. 71A)	3(3-20)	1.0	0.00		1.0	0.00		5.0	0.00		1.0	0.00	
Nordnes (st. I241)	2(2-20)	1.0	0.00		3.1	1.58	1[4.23]	26.3	11.10	2[18.4-34.1]	1.0	0.00	
Måløy (st. 26A2)	3(3-53)	1.0	0.00		1.0	0.00		12.5	3.71	2[12.5-17.3]	1.0	0.00	
Outer Trondheimsfjord (st. 91A2)	3(3-72)	1.0	0.00		1.0	0.58	1[3]	19.7	6.72	2[19.7-22.9]	1.0	0.00	
Bodø harbour (st. 97A2)	3(3-230)	1.0	0.00		1.0	0.00		19.1	3.87	3[14.6-22.3]	1.0	0.00	
Lofoten, Svolvær (st. 98A2)	3(3-99)	1.0	0.00		1.0	0.00		17.1	4.48	3[16.6-24.6]	1.0	0.00	
Cod liver													
Inner Oslofjord (st. 30B)*	12(4-3)	1.0	3.11		4.2	1.32	10[2.96-7.18]	22.8	54.27	9[16.1-49.8]	1.0	3.11	
Færder area (st. 36B)	3(3-7)	1.0	0.00		3.6	1.12	3[2.28-4.5]	20.6	8.36	3[13.1-29.8]	1.0	0.00	
Hvaler (st. 02B)	2(2-8)	2.5	0.00		2.5	0.00		25.0	0.00		2.5	0.00	
Grenlandsfjord (st. 71B)	5(3-3)**	2.5	2.64		2.5	3.10		25.0	16.43		10.0	8.97	
Grenlandsfjord (st. 71B)	9(7-3)	2.5	1.23		2.5	1.23		25.0	0.00		3.8	2.74	
Kristiansand harbour (st. 13B)	6(6-2)	0.5	0.52		0.5	0.43	1[2.04]	5.5	3.09	3[5.92-12.1]	0.5	1.52	1[3.89]
Inner Sørfjord (st. 53B)	6(4-6)	0.5	0.10	2[1-1.3]	0.5	1.33	1[1.09]	5.0	29.66		0.5	1.32	1[1.52]
Bømlo north (st. 23B)	9(5-2)	2.5	7.50		5.0	6.29		0.0	0.00		5.0	6.29	
Ålesund area (st. 28B)	4	1.0	1.06		2.5	1.04	3[6-6.76]	25.0	14.08		1.0	4.79	
Inner Trondheimsfjord (st. 80B)	15	2.5	0.00		2.5	0.00		25.0	0.00		2.5	0.00	

**) Count 5(3-3) for 4-t-NP

3.3.7 Phthalates

Supplementary analyses of phthalates in 2013

Phthalates are mainly used as plasticizers and have large variety of usages such as in paints, building products, lubricants, dispersants, emulsifiers, electronics as well as personal-care products, pharmaceuticals, medical devices and food products. Phthalate comprise a number of substances one of which (di(2-thylhexyl)-phthalate or DEHP) is on the EQSD list a priority hazardous substances but has no EQS for biota. Eleven phthalates, including DEHP, were analysed in 2012 samples and for the first time as part of the MILKYS programme. In Norway since 1999 phthalates have been prohibited in toys and products for children less than three years of age. From January 1 2007 it has been prohibited for all toys for children to the age of 14.

Concentrations in samples from 2013 were assessed for 34 cases in five fjord areas: the Inner Oslo fjord, Grenlandsfjord area, Inner Sørfjord, Ålesund and Tromsø harbours (***Table 21***). With two exceptions all values were below the limit of detection. The limit of detection varied from 0.2 mg/kg w.w. for BBP, DIBP and DIPA to 2.5 mg/kg w.w. for DIDP, DIHP, and SDD. The exceptions were to cod liver samples from the Kristiansand harbour where SDD (dinonylphthalte+diisononylphthalate) was 7 and 7.3 µg/kg w.w. Bakke *et al.* (2007) found concentrations of DEHP in cod liver from the Oslofjord, Ålesund, Tromsø and Varanger to vary from 0.3 (Tromsø) to 55.7 (Varangerfjord) mg/kg w.w. Phthalates were also analysed in liver of cod caught in 2012 (Green *et al.* 2014). The concentrations were low. No concentrations were found to be above the detection limits.

***Table 21**. Median concentrations (µg/kg w.w.) with standard deviation of phthalates in blue mussel and cod liver in supplementary analyses of 2013-samples. Count indicates number of samples analysed. The first number within the parentheses indicates the number of pooled samples included. The second number within the parentheses indicates for mussels the total number of individuals used in all pooled samples and for cod the number individuals in each pooled sample. Shaded values are below one half the detection limits, and indicate that over half of the values are below this limit. The standard deviation was based on all values. However caution should be used when comparing such values because the limit of detection can vary both within and among samples. Detectable data information (D.d.i.) indicates the number of data above the limit of detection (if any) and the numbers within the square brackets indicate the minimum and maximum values in this category. (See Appendix B for description of chemical codes.)*

Component	Count	BBP			DBP			DBPA			DEHA			DEHP			DEP			DEPA			DIBP			DIDP		
Species and sampling locality	2013	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.
Blue mussel																												
Akershuskaia (st. I301)	1(1-50)	0.2	0.00		0.2	0.00		0.3	0.00		0.3	0.00		0.5	0.00		0.3	0.00		0.3	0.00		0.2	0.00		2.5	0.00	
Gåsøya (st. I304)	2(2-15)	0.2	0.00		0.2	0.00		0.3	0.00		0.3	0.00		0.5	0.00		0.3	0.00		0.3	0.00		0.2	0.00		2.5	0.00	
Byrkjenes (st. 51A)	3(3-100)	0.2	0.00		0.2	0.00		0.3	0.00		0.3	0.00		0.5	0.00		0.3	0.00		0.3	0.00		0.2	0.00		2.5	0.00	
Kvalnes (st. 56A)	3(3-100)	0.2	0.00		0.2	0.00		0.3	0.00		0.3	0.00		0.5	0.00		0.3	0.00		0.3	0.00		0.2	0.00		2.5	0.00	
Cod liver																												
Inner Oslofjord (st. 30B)	6	0.2	0.00		0.2	0.00		0.3	0.00		0.3	0.00		0.5	0.00		0.3	0.00		0.3	0.00		0.2	0.00		2.5	0.00	
Kristiansand harbour (st. 13B)	5(5-2)	0.2	0.00		0.2	0.00		0.3	0.00		0.3	0.00		0.5	0.00		0.3	0.00		0.3	0.00		0.2	0.00		2.5	0.00	
Ålesund area (st. 28B)	5	0.2	0.09		0.2	0.09		0.3	0.22		0.3	0.67		0.5	0.22		0.3	0.22		0.3	0.00		0.2	0.09		2.5	2.24	
Tromsø harbour (st. 43B2)	9	0.2	0.00		0.2	0.07		0.3	0.17		0.3	1.50		0.5	0.17		0.3	0.17		0.3	0.00		0.2	0.07		2.5	1.67	

Component	Count	DIHP			DINCH			DIPA			DMP			DNOP			DPF			SDD			TOA		
Species and sampling locality	2013	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.
Blue mussel																									
Akershuskaia (st. I301)	1(1-50)	2.5	0.00		0.3	0.00		0.2	0.00		0.3	0.00		0.3	0.00		0.3	0.00		2.5	0.00		0.3	0.00	
Gåsøya (st. I304)	2(2-15)	2.5	0.00		0.3	0.00		0.2	0.00		0.3	0.00		0.3	0.00		0.3	0.00		2.5	0.00		0.3	0.00	
Byrkjenes (st. 51A)	3(3-100)	2.5	0.00		0.3	0.00		0.2	0.00		0.3	0.00		0.3	0.00		0.3	0.00		2.5	0.00		0.3	0.00	
Kvalnes (st. 56A)	3(3-100)	2.5	0.00		0.3	0.00		0.2	0.00		0.3	0.00		0.3	0.00		0.3	0.00		2.5	0.00		0.3	0.00	
Cod liver																									
Inner Oslofjord (st. 30B)	6	2.5	0.00		0.3	0.00		0.2	0.00		0.3	0.00		0.3	0.00		0.3	0.00		2.5	0.00		0.3	0.00	
Kristiansand harbour (st. 13B)	5(5-2)	2.5	0.00		0.3	0.00		0.2	0.00		0.3	0.00		0.3	0.00		0.3	0.00		2.5	1.18	2[7-7.3]	0.3	0.00	
Ålesund area (st. 28B)	5	2.5	0.00		0.3	2.01		0.2	0.31		0.3	0.22		0.3	0.22		0.3	0.67		2.5	2.24		2.5	2.01	
Tromsø harbour (st. 43B2)	9	2.5	0.00		0.3	1.50		0.2	0.23		0.3	0.17		0.3	0.17		0.3	0.50		2.5	1.67		2.5	0.00	

3.3.8 Triclosan in 2013

Triclosan is a chlorinated aromatic compound (5-chloro-2-2,4-dichlorophenoxy)phenol). It is an antibacterial and antifungal agent which is common in shampoos, deodorants, toothpastes and mouth washes and cleaning supplies. It is highly toxic to different algae and has been found in algae, aquatic blackworms (Annelida), fish and dolphins.

Triclosan was analysed in cod liver from four locations and in blue mussel from four stations. The concentrations in both cod liver and blue mussel were very low. All concentrations were below the detection limits (***Table 22***). Hence, no conclusion can be drawn regarding possible differences between stations.

Table 22*. Concentration (µg/kg w.w.) of triclosan in blue mussel and cod liver in 2013. The shaded areas indicate value below the detection limit and that the values shown are one half of the detection limit.*

Component	Count	Triclosan		
Species and sampling locality	2013	Med.	S.d.	D.d.i.
Blue mussel				
Akershuskaia (st. I301)	2(2-50)	0.005	0.000	
Byrkjenes (st. 51A)	3(3-100)	0.005	0.000	
Kvalnes (st. 56A)	3(3-100)	0.005	0.000	
Cod liver				
Inner Oslofjord (st. 30B)	6	0.025	0.000	
Kristiansand harbour (st. 13B)	7(6-2)	0.025	0.000	
Ålesund area (st. 28B)	5	0.013	0.000	
Tromsø harbour (st. 43B2)	9	0.013	0.000	

3.3.9 Diuron and Irgarol in 2013

Diuron (3-(3,4-dichlorophenyl)-1,1-dimethylurea) is an herbicide that inhibits photosynthesis. Diruon has a relatively low K_{OC}, which indicates a relatively low tendency to sorb to soils and seidments, while its hydrolysis and aqueous half-lives are relatively long. Consequently Diuron is both mobile and relatively persistent. Diuron is moderately toxic to fish and aquatic invertebrates. The commonly used antifouling agent tirbutyltin (TBT) has been shown to cause serious problems to aquatic organisms. As a result, alternative biocides with a reduced toxicity were developed. Diuron and Irgarol are used as a supplement to copper oxides in anifauling paintand are therefore called booster biocides. Irgarol is an algaecide, and may affect non-target photosynthetic organisms such as phytoplankton, periphyton and aquatic macrophytes. Toxicity tests show that small aquatic plant systems seem to be the most affected by Irgarol. Toxicological data seems to indicate that the effects of Irgarol decrease in higher forms of marine life.

Diuron and Irgarol were analysed in cod liver from four locations and in blue mussel from seven stations. The concentrations in both cod liver and blue mussel were very low. All concentrations were below the detection limits (***Table 23***). Hence, no conclusion can be drawn regarding possible differences between stations.

Table 23*. Concentration (µg/kg w.w.) of Diuron and Irgarol in blue mussel and cod liver 2013. The shaded areas indicate value below the detection limit and that the values shown are one half of the detection limit.*

Component	Count	Diuron			Irgarol		
Species and sampling locality	2013	Med.	S.d.	D.d.i.	Med.	S.d.	D.d.i.
Blue mussel							
Akershuskaia (st. I301)	2(2-50)	0.0025	0.0000		0.0050	0.0000	
Gåsøya (st. I304)	2(2-15)	0.0025	0.0000		0.0050	0.0000	
Solbergstrand (st. 31A)	1(1-48)	0.0025	0.0000		0.0050	0.0000	
Mølen (st. 35A)	3(3-88)	0.0025	0.0000		0.0050	0.0000	
Byrkjenes (st. 51A)	3(3-100)	0.0025	0.0000		0.0050	0.0000	
Kvalnes (st. 56A)	3(3-100)	0.0025	0.0000		0.0050	0.0000	
Cod liver							
Inner Oslofjord (st. 30B)	6	0.0025	0.0000		0.0050	0.0000	
Kristiansand harbour (st. 13B)	5(5-2)	0.0025	0.0000		0.0050	0.0000	
Ålesund area (st. 28B)	5	0.0025	0.0000		0.0050	0.0000	
Tromsø harbour (st. 43B2)	9	0.0025	0.0000		0.0050	0.0000	

3.4 Biological effects methods for cod in the Inner Oslofjord

Biological effect parameters (BEM) are included in the monitoring program to assess the potential pollution effects on organisms. This cannot be done solely on the basis of tissue concentrations of chemicals. There are five BEM methods used (including analyses of degradation products of PAH in bile). Each method is in theory specific for individual or groups of chemicals. One of the advantages of these methods used at the individual level is the ability to integrate biological and chemical endpoints, since both approaches are performed on the same individuals. The results can be seen in relation to newly established reference values (e.g. OSPAR 2013).

3.4.1 OH-pyrene metabolites in bile

Analysis of OH-pyrene in bile is not a measurement of biological effects, per se. It is included here, however, since it is a result of biological transformation (biotransformation) of PAHs, and is thus a marker of exposure. Detection methods for OH-pyrene have been improved two times since the initiation of these analyses in the CEMP/MILKYS programme. In 1998, the wavelength for measurement of light absorbance of the support/normalisation parameter biliverdine was changed to 380 nm. In 2000, the use of single-wavelength fluorescence for quantification of OH-pyrene was replaced with HPLC separation proceeding fluorescence detection. The single wavelength fluorescence method is much less specific than the HPLC method. Although there is a good correlation between results from the two methods, they cannot be compared directly.

PAH compounds are effectively metabolized in vertebrates. As such, when fish are exposed to and take up PAHs, the compounds is biotransformed into polar metabolites which enhances the efficiency of excretion. It is therefore not suitable to analyse fish tissues for PAH parent compounds as a measure of exposure. However, since the bile is a dominant excretion route of PAH metabolites, and since the metabolites are stored for some time in the gall bladder, the bile is regarded as a suitable matrix for analyses of PAH metabolites as a measure of PAH exposure.

In 2014 the median concentration of OH-pyrene metabolites in bile from cod in the Inner Oslofjord (st. 30B) was about 10 % lower than the 2013-concentration and 30 % lower than the 2012-concentration. Al though there has been apparent annual decrease in the median concentration of OH-Pyrene over the last five years, no significant temporal trend could be observed over the last 10 years (Appendix F). Median OH-pyrene bile concentration in 2014 was above the ICES/OSPAR assessment criterion (background assessment criteria, BAC) in this area as well as in fish from the Inner Sørfjord (st. 53B), Lista area (st. 15B) and Bømlo on the West coast (st. 23B, reference station). Note that the unit of the assessment criterion is ng/ml, without normalization to absorbance at 380nm.

3.4.2 ALA-D in blood cells

Inhibited activity of ALA-D indicates the influence of lead contamination. Although ALA-D inhibition is lead-specific, it is not possible to rule out interference by other metals or organic contaminants.

In 2014, ALA-D activities in the blood of cod from the Inner Oslofjord (st. 30B) had apparently increased from the activities observed in 2012 and 2013, to approximately the same levels as observed in 2010 and 2011. No significant temporal trends could be observed over the last 10 years (Appendix F). The ALA-D results are not in agreement with the apparent increase in the median concentration of lead in cod liver from the last five years (see section 3.2.3). No significant

temporal trend in lead concentrations, however, could be observed over the last 10 years (Appendix F).

Most years up to 2011 the activity of ALA-D in cod was somewhat inhibited in the Inner Oslofjord (st. 30B), compared to reference stations, i.e. Outer Oslofjord (st. 36B; only data to 2001), Bømlo in the Bømlo-Sotra area (st. 23B), and Varangerfjord (st. 10B; only data to 2001, not shown) (Green *et al*. 2012a). The ALA-D activity at Bømlo in 2014 was apparently higher than both the Inner Oslofjord and the Inner Sørfjord (st. 53B). The lower activities of ALA-D in cod from the Inner Oslofjord and Inner Sørfjord compared to the reference station (basis for comparison prior to 2007, 2009-2011 and 2013-2014) indicate the contamination of lead. The higher concentrations of lead in cod liver are generally observed in the Inner Oslofjord and Inner Sørfjord compared to Bømlo, though with a relatively large individual variation.

3.4.3 EROD-activity and amount of CYP1A protein in liver

High activity of hepatic cytochrome P4501A activity (EROD-activity) normally occurs as a response to the contaminants indicated in ***Table 5***. It was expected that higher activity would be found at the stations that were presumed to be most impacted by planar PCBs, PCNs, PAHs or dioxins such as the Inner Oslofjord (st. 30B). In 2014, median EROD-activity in liver of cod from the Inner Oslofjord (30B) was very similar to that observed in 2012 (i.e. half of that in 2013). Since 2000, the median EROD-activity has generally been higher in the Inner Oslofjord compared to the reference station on the west coast (Bømlo, st. 23B); however, in 2014 this was not the case. No significant temporal trends could be observed for EROD in cod liver, and median EROD-activities were below the ICES/OSPAR assessment criterion (background assessment criteria, BAC).

No adjustment for water temperature has been made. Fish are sampled at the same time of year (September-November) when differences between the sexes should be at a minimum. Statistical analyses indicate no clear difference in activity between the sexes (Ruus *et al*. 2003). It has been shown that generally higher activity occurs at more contaminated stations (Ruus *et al*. 2003). However, the response is inconsistent (cf. Appendix F), perhaps due to sampling of populations with variable exposure history. Besides, there is evidence from other fish species that continuous exposure to e.g. PCBs may cause adaptation, i.e. decreased EROD-activity response.

As for EROD, Median CYP1A protein level in 2014 in the Inner Oslofjord was approximately half of that in 2013, and thus more resembled that in 2012. No significant long-term or short-term (last ten years) temporal trends in CYP1A protein content or EROD activities could be observed. CYP1A protein levels were apparently higher in the Inner Oslofjord, compared to the Sørfjord and Bømlo, with the possible explanation that the exposure to PCBs was higher in the Inner Oslofjord than at least at Bømlo. It was earlier also observed, however, that EROD activities apparently were not significantly influenced by a substantial increase in cod liver PCB content (Ruus *et al*. 2006). Berge *et al*. (2012) also found higher values in the Inner Oslofjord compared to the Outer Oslofjord. An explanation (besides the adaptation hypothesis) may be that the inducing effect of specific contaminants may be inhibited by other contaminants present (e.g. dioxins or PAHs).

3.5 Monitoring of contaminants with passive samplers

Sampling rates for samplers deployed for a year until July-August 2015 were low, particularly considering the surface area of the samplers (1000 cm^2). However these sampling rates were extremely similar to those obtained in the two previous deployments. The standard errors on the estimation of sampling rates were ~ 10 % (***Table 24***). Sampling rates were lowest for samplers deployed in Hvaler and highest in Ålesund. Sampling rates ranged from 2.0 L d^{-1} for the least hydrophobic substances (e.g. 4-t-octylphenol) to 0.24 L d^{-1} for the most hydrophobic substances (e.g BDE-209). These sampling rates are lower than those obtained with the same type of silicone rubber samplers as part of the *Tilførselprogrammet* (Allan *et al*. 2011; Allan *et al*. 2012).

The extraction and analysis of one QA spiked samplers together with this batch of exposed passive samplers resulted in amount per samplers close to those determined in the initial batch of six QA spiked samplers (Appendix G).

***Table 24**. Estimated sampling rates, R_s for AlteSil silicone rubber samplers (1000 cm^2, 30 g) deployed at three sites for > 300 days.*

	Site					
	Hvaler		Oslofjord		Ålesund harbour	
	Replicate 1	Replicate 2	Replicate 1	Replicate 2	Replicate 1	Replicate 2
R_s* for 2013	0.45	0.58	0.30	0.43	1.41	1.36
+/-	0.04	0.04	0.01	0.01	0.07	0.03
R_s* for 2014	0.53	0.50	0.75	0.68	1.26	1.26
+/-	0.05	0.05	0.07	0.07	0.10	0.15
R_s[a] for 2015	0.42	0.62	0.63	0.62	1.30	1.43
+/-	0.03	0.18	0.05	0.06	0.08	0.08

* Rs (L d^{-1}) at $logK_{sw}$ = 5

As shown in ***Table 25***, most compounds were below limits of detection. In the case of 4-t-OP, 4-t-NP, and BDE-209, non-negligible amounts of these substances were measured in field control samplers (and/or in solvent blanks). This affected limits of detection for these compounds. Overall limits of detection depend on the quality of sampler preparation, contamination during sampler extraction and analysis, and instrumental limits of detection.

Masses of alkylphenols (4-t-OP and 4-t-NP) in exposed samplers were not significantly higher than those present in control samplers. Masses of 4-t-OP measured in duplicate samplers were very similar, but levels found in control samplers prevent us from calculating water concentrations. Masses for p-t-NP in duplicate exposed samplers were more variable. The limit of detection was calculated based on 3x the levels found in the control samplers. These LODs remain below the WFD EQS level (New EQS values, Appendix G of 0.3 μg L^{-1} for nonylphenol. Limits of detection range from 0.01 to 23 ng L^{-1} for para-t-octylphenol and para-t-nonylphenol and 0.01-0.13 ng L^{-1} for para-n-octylphenol and para-n-nonylphenol, respectively. No other alkylphenol measurements have been undertaken using silicone rubber samplers until now. Sack & Lohmann (2011) used LDPE to sample these substances and were able to measure freely dissolved concentrations of t-octylphenol in the low ng L^{-1} range (3-11 ng L^{-1}) in Narragansett Bay, a small and heavily urbanized bay (US) with a surrounding population of two million inhabitants.

The technical mixture of HBCD is mainly composed of the γ-isomer (80-85 %), while α-HBCD and β-HBCD account for 8 and 6 % of the mixture, respectively. The concentration of β- and γ-HBCD were below limits of detection (with these in the range 9-20 pg L^{-1}). A freely dissolved concentrations of the α-isomer of HBCD of 14 pg L^{-1} was estimated for the Oslofjord. This is in a very similar range to the data from 2013 (Green *et al.* 2014). Freely dissolved concentrations appear to be well below WFD EQS values for HBCDD published in 2014.

GC-MS analysis of extracts (sum of all isomers) from silicone samplers exposed at Jan Mayen (Allan *et al.* 2012) as part of the *Tilførselsprogrammet* showed that concentrations of HBCD in these samplers were below limits of detection. While passive air sampling of HBCD has been undertaken, passive sampling in water has not been reported (to the author's knowledge).

The exposure of samplers for almost a year (2014-2015) resulted in the accumulation of significant amounts of a multitude of polybrominated substances rendering the quantification of specific PBDEs challenging. Most PBDEs were found below limits of detection and for reasons we do not fully comprehend, chromatograms from exposed samplers from two stations (Hvaler and Oslofjord) did not allow proper quantification of PBDEs. The matrix of these extracts resulted in significant amounts of interferences, despite repeated clean-up of the extracts with concentrated sulphuric acid combined with acetonitrile partitioning steps in an attempt to lower the amount of interfering substances in the samples prior to GC analysis. These substances are likely to be polybrominated and stable compounds (e.g. naturally produced polybrominated compounds) that resist sulphuric acid clean-up. We need to consider an alternative strategy for the sample clean-up and/or analysis to avoid the quantification problems experienced with these samples.

For the Ålesund harbour, freely dissolved concentrations of 3.0 and 4.9 pg L^{-1} for BDE-47 and BDE-99, respectively were estimated for this latest deployment period (data not corrected for temperature or salinity). This values for BDE-47 is lower than that obtained in the Oslofjord in previous years but clearly higher than data from further north on the coast obtained with silicone rubber samplers exposed at Andøya during the *Tilførselsprogrammet* (Allan *et al.* 2011; Allan *et al.* 2012). Freely dissolved concentrations of PBDE congeners measured during the RiverPOP programme (2008-2011) were generally in the low pg L^{-1} range or below for rivers such as the Drammenselva and Glomma (Allan *et al.* 2009; Allan *et al.* 2010; Allan *et al.* 2011).

***Table 25**. Freely dissolved concentrations measured with silicone rubber samplers exposed at three sites.*

Substances		**Freely dissolved contaminant concentrations**		
Sites	**Unit**	**Hvaler**	**Oslofjord**	**Ålesund harbour**
Alkylphenols				
4-t-OP	ng L^{-1}	< 1.2[a]	< 1.2[a]	< 1.3[a]
4-t-NP	ng L^{-1}	< 23[a]	< 15[a]	< 11[a]
4-n-OP	ng L^{-1}	< 0.06[a]	< 0.13[a]	< 0.07[a]
4-n-NP	ng L^{-1}	< 0.03[a]	< 0.05[a]	< 0.04[a]
HBCD				
α-HBCD	pg L^{-1}	< 24	< 15	11.4 (11)
β-HBCD	pg L^{-1}	< 24	< 15	< 9
γ-HBCD	pg L^{-1}	< 25	< 15	16.7 (28)
PBDEs				
BDE47	pg L^{-1}	nd	nd	3.0 (27)
BDE99	pg L^{-1}	nd	nd	4.9 (10)
BDE100	pg L^{-1}	nd	nd	< 1.2
BDE126	pg L^{-1}	nd	nd	
BDE153	pg L^{-1}	nd	nd	< 1.3
BDE154	pg L^{-1}	nd	nd	< 1.3
BDE183	pg L^{-1}	nd	nd	< 1.5
BDE196	pg L^{-1}	nd	nd	
BDE209	pg L^{-1}	< 200	< 120	< 73

[a]) Limit of detection calculated from 3 times the average of amounts found in the field controls (n = 3) and sampler-specific sampling rates.

[b]) Relative percent difference of replicate measurements (%) given in brackets

[c]) Amounts found in exposed samplers higher than 3 times the amounts found in field controls

3.6 Analysis of stable isotopes

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 δ13C) 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 (15N). The relative increase of 15N over 14N (δ15N) is 3-5‰ per trophic level (Layman *et al*. 2012; Post 2002). It thus offers a continuous descriptor of trophic position. As such, it is also the basis for Trophic Magnification Factors (TMFs). TMFs give the factor of increase in concentrations of contaminants per trophic level. If the concentration increase per trophic level can be expressed as:

Log Concentration = a + bTrophic Level

Then:

TMF = 10^b

The trophic magnification factor has recently 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 merely been reviewed to indicate any possibilities that spatial differences in contaminant concentrations may partially be attributed to different energy sources between locations, or that the same species may inhabit different trophic levels on different locations (***Table 26***). It is anticipated that statistical temporal analyses may be applied to perform more "refined" assessments, when the "MILKYS" stable isotope database is further expanded. The δ15N data (Atlantic cod) is also assessed in relation to concentrations of selected contaminants. As fish grow, they feed on larger prey organisms, thus a small increase in trophic level is likely to occur. It is of interest to assess whether concentrations of specific contaminants correlate with δ15N, since this will warrant further scrutiny of the contaminant's potential to biomagnify.

For selected contaminants (BPA, TBBPA, MCCP and TCEP), relationships between concentrations and $\delta^{15}N$ have been investigated to examine potential increase in concentration of the specific contaminants with increasing $\delta^{15}N$. Such correlation will give reason for future examination of the potential of the contaminant to increase in concentration with higher level in the food chain (biomagnification). It is previously shown that e.g. the concentration of mercury increase with $\delta^{15}N$ among individuals of the same species (more specifically tusk; *Brosme brosme*) in the Sørfjord (Ruus *et al*. 2013b). For that reason, also concentrations of mercury, as well as CB153 (another compound with known biomagnifying properties), is plotted against δ15N in cod. The data material for Hg is larger (more individuals analysed per station), than for the other contaminants. For BPA, most concentrations fell below the limit of detection.

There were no great differences in $\delta^{13}C$ between mussels or fish from the different areas, with some exceptions. Furthermore, there were no major differences in $\delta^{15}N$ between cod from different locations, with some exceptions, indicating that the different populations surveyed can be placed on approximately the same trophic level. As mentioned, an increase in $\delta^{15}N$ of 3 to 5 ‰ represent a step of one full trophic level, while the differences observed were generally lower. It

is therefore reasonable to assume that any differences in the concentrations of pollutants between areas are due to differences in exposure (either from local sources or through long-range transport). It can be noted, however, that differences in e.g. mercury content in tusk from Sørfjord area could be partly attributed to small differences in trophic position/$\delta^{15}N$ (less than one full trophic level) (Ruus *et al*. 2013b).

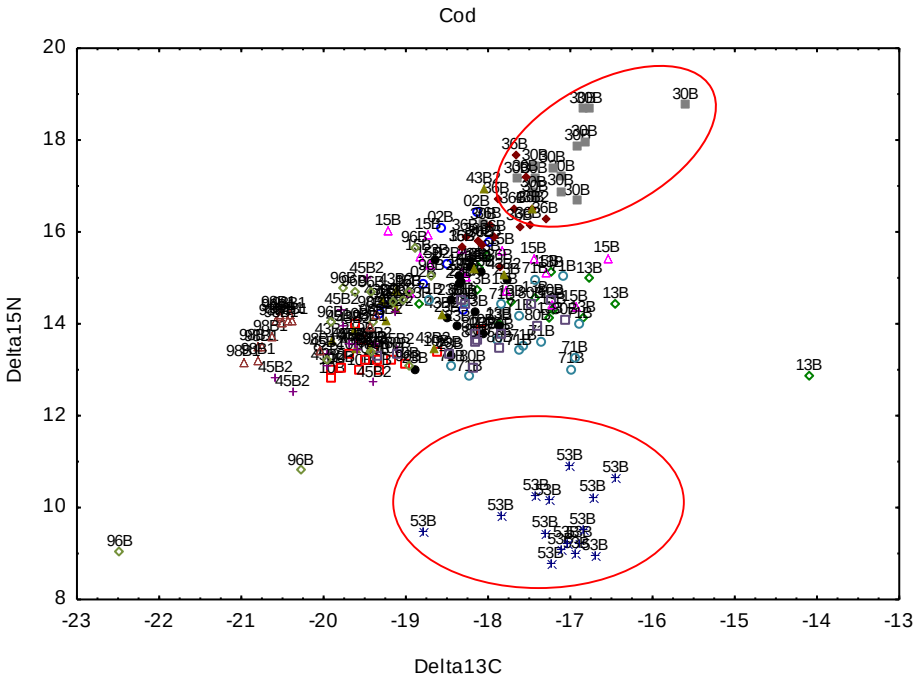
Table 26*.Summary of analyses of stable isotopes: $\delta^{13}C$ and $\delta^{15}N$ in blue mussel and cod, 2014. Statistics shown are count (n), mean and standard deviation.*

	Blue mussel						Atlantic Cod					
	$d^{13}C_{VPDB}$			$d^{15}N_{AIR}$			$\delta^{13}C_{VPDB}$			$\delta^{15}N_{AIR}$		
Station ID	n	mean	st.dev.	n	mean	st.dev.	n	mean	st.dev.	n	mean	st.dev.
presumed less impacted, summary >>	**-21**	**-20.81**	**-20.81**	**7**	**7.37**	**7.37**	**-19**	**-18.67**	**-18.67**	**15**	**14.90**	**14.90**
Tjøme (st. 36A1)	3	-19.98	0.10	3	8.27	0.21						
Gåsøy (st. 15A)	3	-20.73	0.35	3	7.41	0.13						
Espevær (st. 22A)	3	-21.53	0.12	3	7.15	0.12						
Outer Trondheimsfjord (st. 91A2)	3	-21.01	0.12	3	6.65	0.09						
Inner Oslofjord (st. 30B)							15	-17.12	0.55	15	17.45	0.79
Færder area (st. 36B)							15	-17.83	0.31	15	16.26	0.62
Farsund area (st. 15B)							15	-18.04	0.81	15	15.24	0.50
Bømlo north (st. 23B)							15	-18.30	0.29	15	14.44	0.73
Helgeland (st. 96B)							15	-19.60	0.92	15	13.75	1.73
Lofoten, Skrova (st. 98B1)							15	-20.42	0.47	15	13.85	0.38
Varangerfjord (st. 10B)							15	-19.36	0.49	15	13.29	0.34
presumed more impacted, summary >>	**-20**	**-20.31**	**-20.31**	**6**	**6.10**	**6.10**	**-18**	**-18.16**	**-18.16**	**13**	**13.48**	**13.48**
Gressholmen (st. 30A)	3	-19.43	0.22	3	8.11	0.11						
Gåsøya (st. I304)	3	-19.31	0.34	3	8.18	0.05						
Håøya (st. I306)	3	-18.44	0.12	3	8.30	0.18						
Ramtonholmen (st. I307)	3	-19.07	0.05	3	8.38	0.08						
Mølen (st. 35A)	3	-19.67	0.10	3	7.39	0.12						
Singlekalven (st. I023)	3	-20.29	0.06	3	7.80	0.04						
Kirkøy (st. I024)	3	-20.57	0.20	3	7.88	0.12						
Bjørkøya (st. 71A)	3	-19.49	0.02	3	6.61	0.08						
Croftholmen (st. I712)	1	-20.30		1	6.02							
Odderøy (st. I133)	3	-21.13	0.02	3	6.68	0.04						
Byrkjenes (st. 51A)	3	-20.03	0.18	3	3.26	0.18						
Eitrheimsneset (st. 52A)	3	-20.23	0.28	3	2.88	0.59						
Kvalnes (st. 56A)	3	-20.07	0.24	3	2.93	0.08						
Krossanes (st. 57A)	3	-19.92	0.12	3	3.20	0.16						
Ranaskjær (st. 63A)	3	-19.96	0.27	3	4.20	0.45						
Lille Terøy (st. 69A)	3	-21.34	0.06	3	4.85	0.05						
Måløy (st. 26A2)	3	-20.46	0.12	3	5.38	0.06						
Moholmen (st. I965)	3	-22.66	0.09	3	6.41	0.14						
Bjørnebærviken (st. I969)	3	-22.52	0.21	3	6.31	0.09						
Bodø harbour (st. 97A2)	3	-21.40	0.09	3	7.26	0.23						
Hvaler (st. 02B)							8	-18.61	0.49	8	15.19	0.86
Grenslandsfjord (st. 71B)							15	-17.66	0.57	15	13.87	0.71
Kristiansand harbour (st. 13B)							15	-17.34	1.10	15	14.55	0.65
Inner Sørfjord (st. 53B)							15	-17.17	0.56	15	9.65	0.65
Inner Trondheimsfjord (st. 80B)							15	-18.07	0.63	15	13.82	0.48
Tromsø harbour (st. 43B2)							15	-18.89	0.73	15	14.54	1.05
Hammerfest (havn) (st. 45B2)							14	-19.70	0.42	14	13.54	0.67
Grand Total	**70**	**-20.40**	**1.03**	**70**	**6.32**	**1.84**	**202**	**-18.43**	**1.18**	**202**	**14.22**	**1.88**

Although there were generally no major differences in $\delta^{15}N$ between cod from different locations, cod from the Sørfjord (station 53B) stand out with particularly low $\delta^{15}N$ signature. The same is shown for mussels from the same area (stations 51A and 56 A, as well as 63A in the Hardangerfjord area), indicating that the $\delta^{15}N$ -baseline of the food web in the Sørfjord is lower. The reason for this is unknown, but a higher influence of allochthonous nitrogen is possible. Likewise, isotope signatures of both fish and mussels from the Oslofjord are among the highest observed (***Figure 44***) indicating a high baseline (and not a higher trophic position of the Oslofjord cod). Furthermore, this was also shown in 2012 and 2013. In fact the stations show very similar patterns from 2012,

through 2013, to 2014 in terms of isotopic signatures, suggesting that this is a spatial trend more than a temporal trend.

a



b

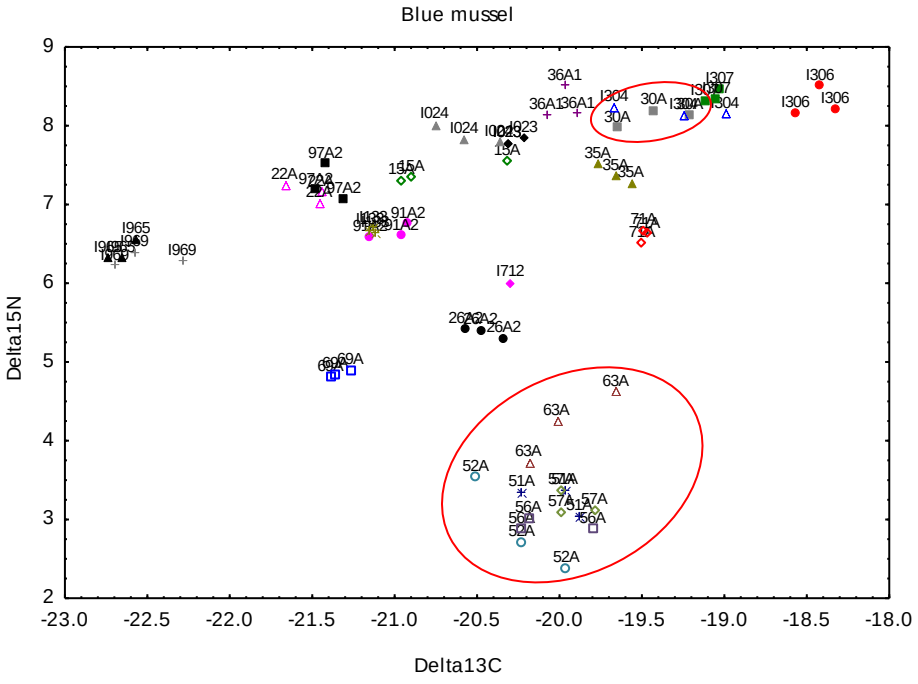


Figure 44. *$\delta^{13}C$ plotted against $\delta^{15}N$ in for cod (a) and blue mussel (b). Station codes are superimposed. Red ellipses indicate cod and blue mussel from the Inner Oslofjord and the Sørfjord, respectively.*

Plotting $\delta^{15}N$ against the concentration of Hg in cod could suggest higher concentrations in individuals with higher $\delta^{15}N$ (significant linear regression between $\delta^{15}N$ and Log[Hg], with very poor goodness-of-fit; R^2=0.093; P=0.00001; ***Figure 45***), However, this is likely partly a result of different exposure, as well as difference in isotopic signature (baseline) among stations (high Hg-exposure as well as high $\delta^{15}N$ in cod from 30B, and low $\delta^{15}N$ baseline at 53B). But a linear regression excluding stations 53B and 30B also produced significant result (R^2=0.249; P<0.0001). However, from ***Figure 45***, there are some indications of increasing Hg-concentrations with increasing $\delta^{15}N$ within stations. Linear regressions isolated for each station produced significant positive linear relationships between $\delta^{15}N$ and Log[Hg] for stations 10B, 15B, 23B, 30B, 36B, 43B2, 45B2, 53B, 71B, 80B and 96B.

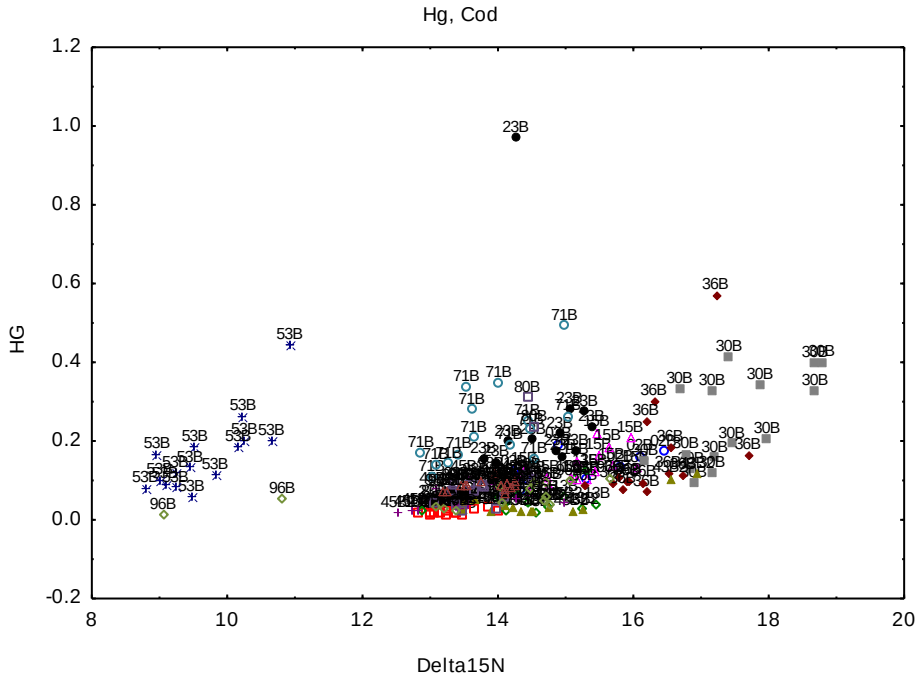


Figure 45. *$\delta^{15}N$ plotted against the concentration of Hg in cod. Station codes are superimposed.*

Plotting $\delta^{15}N$ against the concentration of CB153 in cod could suggest higher concentrations in individuals with higher $\delta^{15}N$ (significant linear regression between $\delta^{15}N$ and Log[CB153]; R^2=0.327; P<0.0001; ***Figure 46***). However, this is most likely partly a result of different exposure, as well as difference in isotopic signature (baseline) among stations (high CB153-exposure as well as high $\delta^{15}N$ in cod from 30B). A linear regression excluding stations 30B and 53B (only one individual cod with high concentration of CB153) also produced significant result (R^2=0.321; P<0.00001). Linear regressions isolated for each station produced significant positive linear relationships between $\delta^{15}N$ and Log[CB153] for stations 43B2, 80B and 96B.

Plotting $\delta^{15}N$ against the concentration of MCCP in cod gives no indication of higher concentrations in individuals with higher $\delta^{15}N$, but merely indicates stations with the highest exposure (especially 80B and 23B), as well as the above mentioned difference in isotopic signature among stations (***Figure 47***). In 2012 and 2013, the highest MCCP concentrations were found at station 80BHB, in addition to station 53B.

Plotting $\delta^{15}N$ against the concentration of TBBPA or TCEP in cod gave no indication of higher concentrations in individuals with higher $\delta^{15}N$.

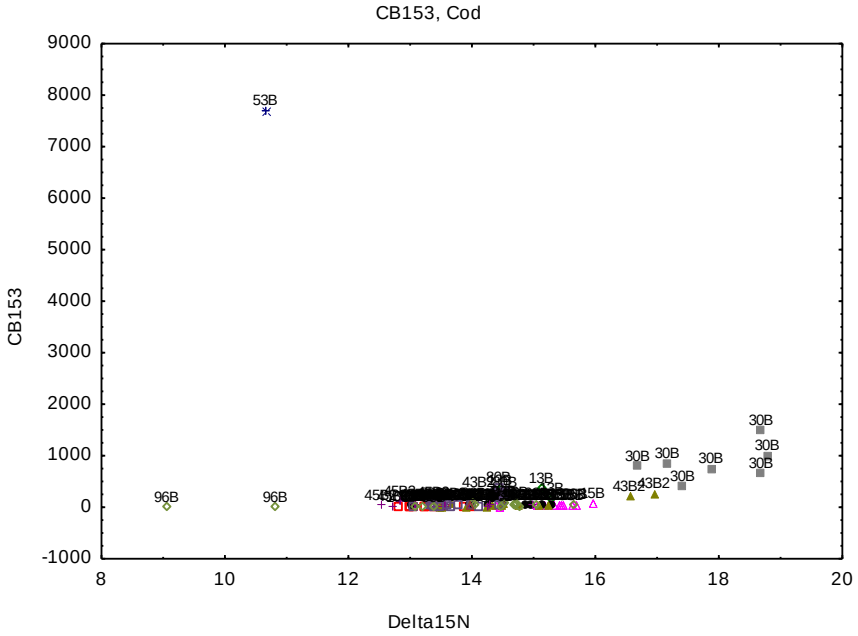


Figure 46. *δ^{15}N plotted against the concentration of CB153 in cod. Station codes are superimposed.*

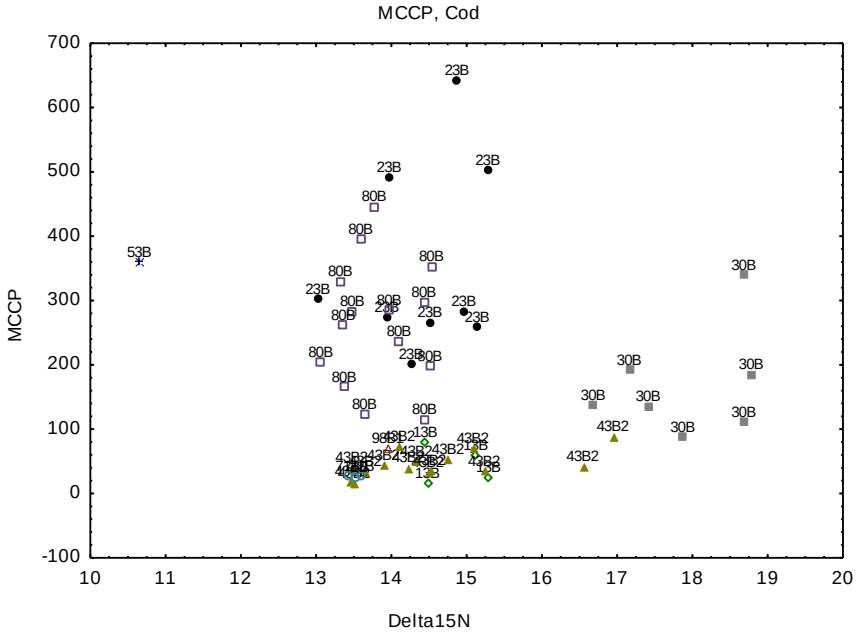


Figure 47. *δ^{15}N plotted against the concentration of MCCP in cod. Station codes are superimposed.*

3.7 Note on methods: Cod length and Hg concentrations in the Inner Oslofjord

The influence of cod length on Hg-concentrations is well established (e.g. Green & Knutzen 2002, Juhlshamn *et al*. 2013, Jones *et al*. 2013, Ruus *et al*. 2012; 2013a, Sacket *et al*. 2013; Eikenberry *et al*. 2015).This factor is examined more carefully for cod from the Inner Oslofjord.

Upward trends of Hg have also been registered in freshwater fish species of Norway (see Fjeld *et al*. 2010). Fjeld *et al*. (2010) point to observations that the atmospheric deposition of Hg in the south eastern part of Norway has decreased significantly over the last years (Wängberg *et al*. 2010), and thus they expected to find a decrease or unchanged levels of Hg in fish (inland waters). They suggested that increased wash-out of humus substances in inland water can lead to increased microbial activity in the sediment and increased methylation of Hg. This would make Hg more bioavailable. The amount of particles in the surface water in the Inner Oslofjord has however been reduced over several decades (Berge *et al*. 2013a) and the input of organic carbon to the sediments in the Inner Oslofjord have more likely been reduced. The factors controlling methylation processes in sediments are not well understood and it should not be ruled out that change in organic carbon input and deep water renewals may have altered redox conditions towards increased methylation at the sediment water boundary. Other possible mechanisms might be weakened photodemethylation in surface waters or altered trophic links, e.g. a shift in cod diet to prey items with higher Hg-content. It should be noted that detecting the impact of changes in discharges/inputs of Hg will also depend on how well fish biotmetrics (length, age and growth rates) are taken into account (Jones *et al*. 2013).

Annual median Hg-concentrations in cod from the Inner Oslofjord showed both significant upward long-term and short-term trends (***Table 10***, ***Figure 8***). The median length of the cod sampled has also shown increasing trends (***Figure 48***). This is consistent with results of the beach seine surveys (dating back to 1919) conducted in the Inner Oslofjord (Espeland & Knutsen, 2014), showing that cod recruitment in the area has been low since the start of the 2000s, and in particular recruitment since 2008 has been low even compared to the historical low trend the most recent years. No recruits of cod were found in the Inner Oslofjord in 2014, so there is no improvement in sight with respect to the lack of 40-50 cm cod (2-4 year old cod).

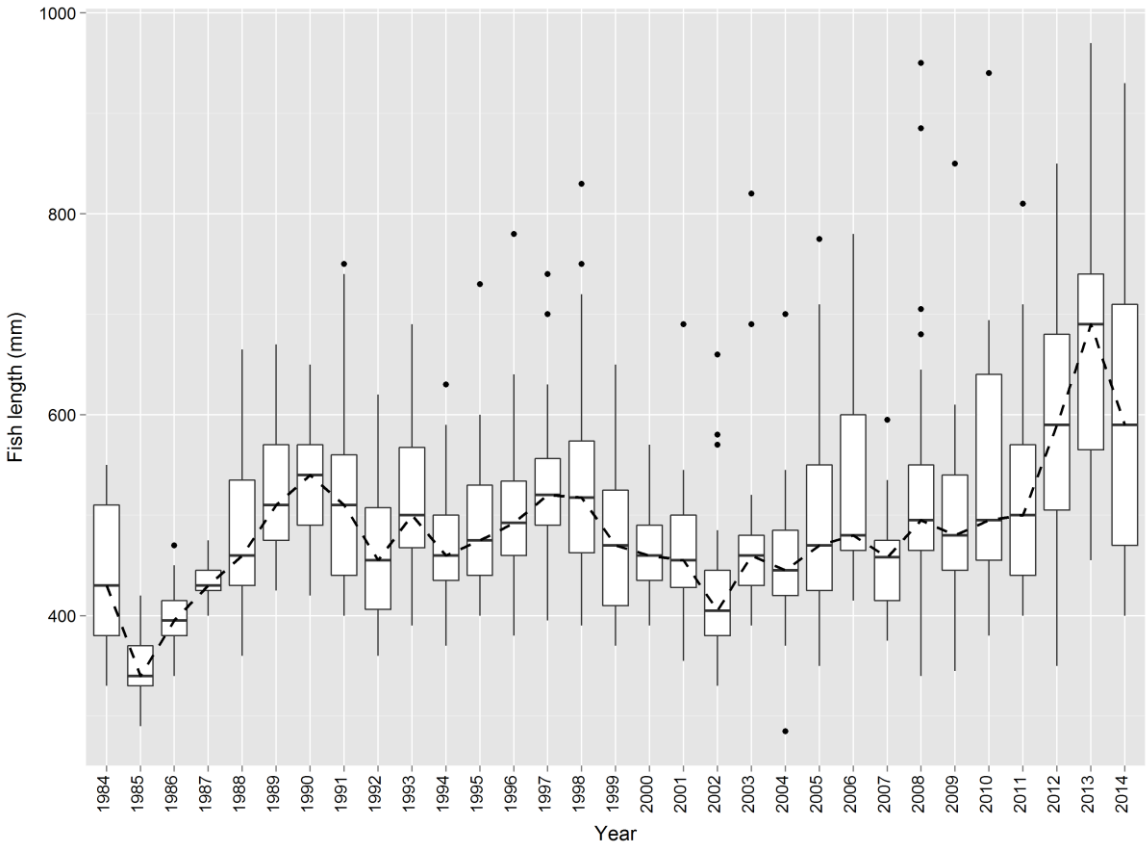


***Figure 48**. Annual median cod length of catch in the Inner Oslofjord for the environmental monitoring (1984-2014).*

As mentioned above it is well known that Hg in fish accumulates with age and thus length. A General Linear Model ([Hg] = a + bLength + c_iYear$_i$ + d_iLength×Year$_i$ + ε; where a to d are constants and ε is the error term; i pertains to each year), yielded the interaction term Length×Year significant, indicating different increase in Hg concentration with length among years ($p < 0.0001$, F-test; ***Figure 49***). The result is the same if we analyse only the last 20 or 10 years ($F > 4.48$, $p < 0.0001$). ***Figure 50*** depicts this relationship for the last six years with median Hg concentration (as used in the time trend analyses) and body length superimposed.

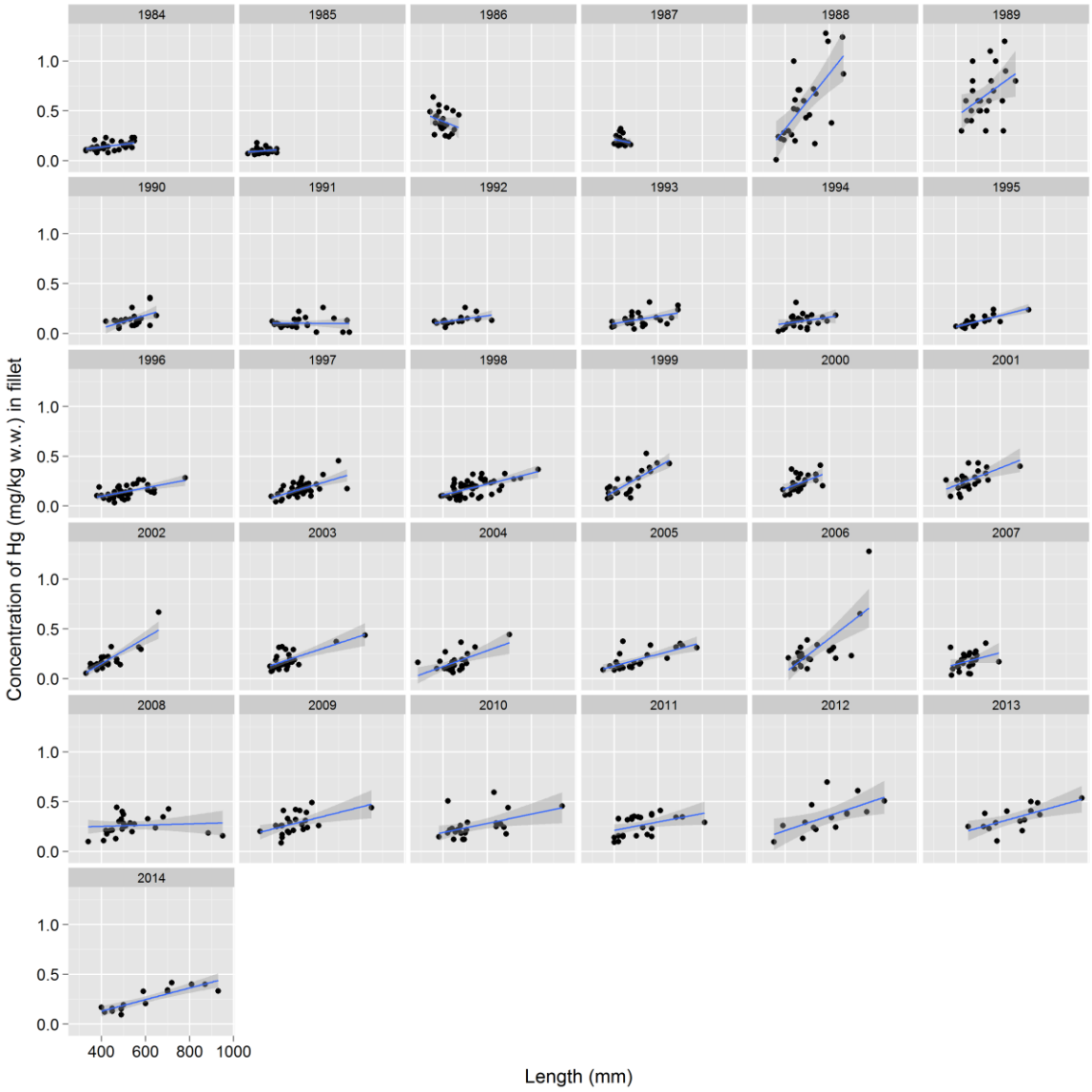


Figure 49. *Relationships between Hg-concentrations (mg/kg w.w. in fillet) and body length (mm) of cod from the Inner Oslofjord (st. 30B) each year (1984-2014) of the environmental monitoring. The dots show the actual data, omitting a single very high value (1.41 mg/kg w.w.) in 1986. The line represents a linear regression of the relationship between Hg-concentrations and length, whereas the shaded area is the 95 % confidence interval of this regression.*

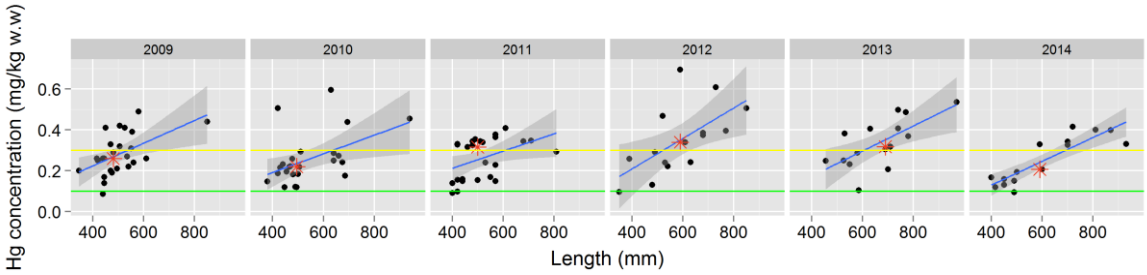
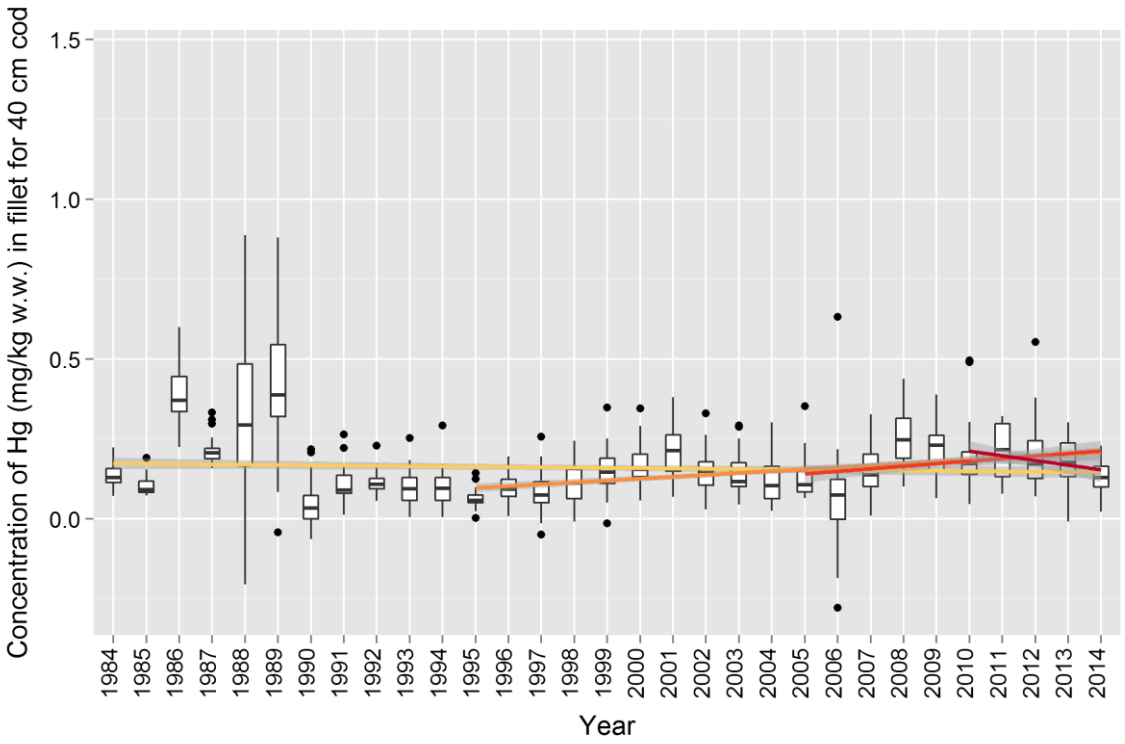


Figure 50. *Relationships between Hg-concentrations (mg/kg w.w. in fillet) and body length (mm) of cod from the Inner Oslofjord (st. 30B) each of the years 2009-2014. Median fish length and Hg-concentration are superimposed (*). Also shown are the lower limits for environmental condition Class II (moderately polluted marked by green line) and Class III (markedly polluted marked by a yellow line).*

Because of the significant interaction between length and year, the time trend will depend on which length we use to normalize concentrations. Therefore, the Hg-concentrations in the cod were normalized to two arbitrary lengths: 400 mm and 600 mm to illustrate how this interaction would impact the trend analyses. If one succeeds in sampling cod following the length groups given in ***Table 1***, the median size will be close to 500 mm. The lengths 400 and 600 mm were chosen to represent "moderately small" and "moderately large" cod respectively; in all years 1988-2009 the median as well as the 25th and 75th percentiles of the data were between 400 and 600 mm. The normalization was done by projectingthe Hg-concentration in each fish parallel to the regression line for the respective year (***Figure 49***), based on the difference in body length between the reference lenth (e.g. 500 mm) and the slope of the regression for the respective year. Results for reference lengths 400 and 600 mm are presented in ***Figure 51***. We used the OSPAR procedure (using only the median value for each year) to test whether there were time trends for the whole time series as well as for the last 20, 10 and 5 years. For concentrations normalized to 500 mm or 600 mm long cod, no time trends were significant at a level of $P < 0.05$, although the 20 year time trend was close to be significantly positive ($P = 0.06$) for concentrations adjusted to 500 mm length. For concentrations normalized to 400 mm long cod, concentration was significantly increasing by 0.0050 (standard error of 0.0018) on a 20 year time scale ($P = 0.012$). Thus, Hg concentrations of small cod have changed more over time (upwards and downwards) than Hg concentrations of larger cod.

From these data it was concluded that most of the upward trend in Hg-concentrations in cod fillet from the Inner Oslofjord during the last 10-20 years could be attributed to the catching of larger fish. If we analyse the variation in all cod over the last 20 years (excluding 2006, when the effect of length was particularly important), 30% of the total variation in Hg concentration can be explained solely by variation in length. Adding a linear time trend explains further 8%, and year-to-year variation ("ups-and-downs") in addition to the linear time trend explains further 12%. Finally, the interaction between year and length explains 5%, bringing the total explained variation to 55%, while 45% of the variation cannot be explained by these two variables. More research is needed to investigate the influence of other explanatory variables than length. Furthermore, there is a need to investigate the influence of length and other explanatory variables, also in time series from other localities and different contaminants, for example PCBs which has been in Class III or Class IV in the Inner Oslofjord since 2000.

a



b

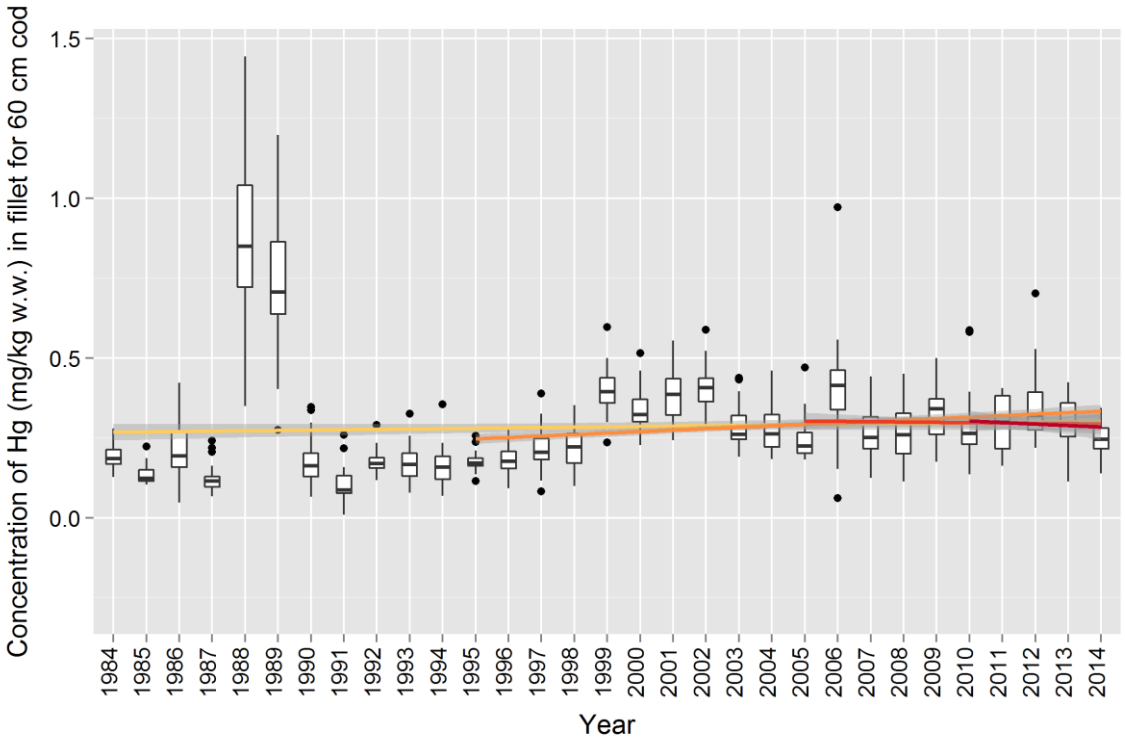


***Figure 51**. Concentrations of Hg in cod from the Inner Oslofjord (st. 30B), each year for the period 1984-2014. Concentrations are normalized to fish length of 400 mm (**a**) and 600 mm (**b**), respectively. Trend lines (whole period, 20 years, 10 years and 5 years) are superimposed.*

3.8 Note on methods: Pooled samples

3.8.1 Background

Depending on economy and practical circumstances, laboratory analyses of contaminants may be done for samples from individual fish, pooled samples where one sample contain the tissue (typically liver) from several fish, or a combination. For instance, from a sample of 25 cod, one may analyse each fish individually and obtain 25 concentrations for each contaminant, or samples may be divided in 5 equal pools (with 5 fish in each), resulting in 5 concentrations, or one may analyse 5 fish individually and divide the 20 remaining fish in 4 pools of 5 fish, resulting in 5 concentrations. We here analyse historical data from cod to explore how pooling of affect results, in particular pooling with unequal number of fish in pools. This is a follow-up from a similar study in a previous report (Green *et al*. 2014), where only pooling of 5 individuals per pool were considered.

As in the previous study, we use the existing historical data on contaminants in cod, taken from the MILKYS database, to simulate mathematically how the pooling of samples affects the estimated contaminant levels and trends. In other words, we can mimick the results of the physical pooling by pooling the concentrations mathematically. We assume that physically pooling and homogenising the tissue samples from several individuals is equivalent to taking the arithmetic mean of the contaminant concentrations of the individual samples. One important assumption is therefore that the amount of tissue taken from each animal is the same.

3.8.2 Methods

The data that is the basis of the present analysis is concentrations of environmental contaminants in cod fillet (for mercury) and liver (for all other compounds). We used only data from 1990 onwards, and only stations with at least 18 years of measurements since 1990. For these stations, we picked only the parameters that have been measured at least 20 years for at least one station. Finally, we picked only those data series (one data series = one contaminant measured in one tissue in one station) which had a sample size of 25 individual samples in at least 7 years. This left us with 87865 concentrations of 22 different contaminants measured in 8 stations, and a total of 168 time series (which is slightly less than 22*8 = 176, as not all compounds were measured at all stations).

We simulated pooling by dividing the total sample (one contaminant, station and year) into groups of individuals, and then took the arithmetic mean of the measured concentrations for each of these groups.

We based the study on four data sets (strategies), the original data set, and three data sets derived from the original data set:

1) Unpooled: The original data set of samples from individual fish
2) Equal pools: For each station and year the individuals were divided into equally large pooled groups of 5 individuals, and we used the mean concentration (for each compound) of each group as the new value. If the total sample size could not be divided by 5, the last pooled sample was smaller than 5 (e.g. 23 individuals was divided in 5 pooled tissue samples, each consisting of 5, 5, 5, 5 and 3 individuals). Since most station-years had 21-25 individuals, we typically ended up with five concentration for each compound.
3) Singles + pooled samples: From each site/concentration, we used the individual concentrations of 5 random individuals, and divided the rest of the concentrations into groups of 5 as described above. So if 23 fish were collected, this mimicks the situation that 5 fish were analysed individually, while the rest were divided into groups of 5, 5, 5, and 3 individuals and pooled, giving a total of 5+4 = 9 concentrations.

4) Pools following weight: Divide the catch of fish into samples with approximately equal weight. One example: If one catches twelve 1 kg-fish, six 2 kg-fish and three 4 kg-fish, on could make three samples of the fish where each sample would weight 12 kg. This mimicks the situation where the researchers have to combine livers from several fish in order to get pooled samples that are large enough for laboratory analyses. Weight of fish were used as a proxy for liver weight, for which there are not always good measurements. In the analyses below, we used pooled the individual samples (typically 21-25 individuals) into 5 pools of unequal size (typically with 2-7 individuals per pool). Thus, this data set had approximately the same number of values (5) per station/year/compound as data set 2 ("equal pools"), and thereby the cost of chemical analysis was similar in data set 2 and 4.

Thus, we ended up with four sets of data – the original data and the three resampled pooled data sets – each containing 168 time series. We compared the suitability of the pooling methods by subjecting each of these 4 data sets to two analyses:

1) Detection of time trends (increasing / decreasing concentrations) in time series of varying length (4 – 20 years)
2) Estimation of concentrations in a single year (for a specific compound at a specific station), in our case 2011.

3.8.3 Detecting time trends

Here, we used the 168 time series as the basis for resampling shorter time series from the 20 year period 1994-2013. The subsets were of length 4, 8, 12, and 16 years. The start year of each series were from 1992 with 4 year intervals (1992, 1996, … to 2010). Thus we got 5 4-year-long series (1994-1997, 1998-2001 etc. until 2010-2013), 4 8-year-long series (1994-2001, 1998-2005 etc.), 3 12 year long series and 2 16 year long series. It should be noted that the shorter time series overlap with the longer, and the longer time series (8-16 years) were mutually overlapping, Therefore, our time series are not independent (which does not invalidate the data for our purpose, which focuses on measuring how pooling affect time trend analysis). For each contaminant and station, we got 14 time series (5+4+3+2) of length 4-16 years, in addition to the full time series of 20 years.

For all time series (of each compound and station) we analysed the presence of time trends using ordinary regression analysis. For the raw (unpooled) data, the result is shown in ***Table 27***. For the 4-year time series, 42% (353/840) time series showed no significant trends due to low sample size, while most of the long time series showed statistically significant ($P < 0.05$) time trends one way or the other.

When we pooled the data, sample size went down from typically 20-25 per year in the original data to a sample size of 5 (data set 2 and 4) or 9 (data set 3). Therefore, it was expected that the power of the trend test (i.e. how likely you are to detect a trend that actually exists) went down. However, the decrease in power was modest, even in the case of short time series of 4-8 years (***Table 28***). The best strategy was strategy (data set) 2 and 3, i.e. equally-sized pools and singles + pools, where the power relative to the full data set was around 70-85 % compared to the full data for 4-8 year long time series. In other words, if the full data set picked up a time trend, there was a 15-30% chance that the time trend would be missed if we pooled our data. Taking into account that the strategy 3 "singles + pools" is about twice as costly (with regard to number of chemical analyses) as the strategy 2 "equally-sized pools", the latter strategy is clearly the best of the two for picking up time trends. Strategy 4 "Pools following weight" misses many more time trends and is clearly not an effective strategy in this regard.

We also calculated how often pooling leads to detecting a trend that probably doesn't exist, i.e. what is known as Type I error in statistical terms. The results indicate that pooling doesn't lead to

a substantial increase in this type of error (***Table 29***). In no cases did we find that there were opposite trends in the raw data and the pooled data.

Table 27*. Trends in the raw data, shown for a set of time series of different lengths (4 – 20 years). The time series were subsets of real time series of the of 22 different contaminants measured in 8 stations (see text for detailed explanation).The significance level was set to P < 0.05.*

Length of time series	Significant trend		No trend	Total
	Increasing	Decreasing		
4 years	169	318	353	**840**
8 years	95	361	216	**672**
12 years	33	355	116	**504**
16 years	19	280	37	**336**
20 years	11	141	16	**168**

Table 28*. Power of the pooling strategies. In those cases where a significant trend was found in the raw (unpooled) data (see* ***Table 27*** *), we counted how many percent of the cases we found the same trend in the pooled data.*

		4 years	8 years	12 years	16 years	20 years
Data set (strategy)	**Positive trend detected when unpooled analyses showed a positive trend (%)**					
1	Unpooled	100	100	100	100	100
2	Equal pools	74.6	68.4	84.8	84.2	63.6
3	Single + pooled	71	71.6	75.8	73.7	63.6
4	Pools following weight	63.9	64.2	66.7	78.9	63.6
	Negative trend detected when unpooled analyses showed a negative trend (%)					
1	Unpooled	100	100	100	100	100
2	Equal pools	78.9	85.3	89.6	91.8	95
3	Single + pooled	76.7	84.8	89.9	94.3	97.2
4	Pools following weight	73.6	80.1	84.5	87.5	88.7

***Table 29**. Type I errors (detecting a trend that probably doesn't exist) of the pooling strategies.In those cases where there was no trend in the raw (unpooled) data (P > 0.2), we counted how many percent of the cases we found a trend in the pooled data.*

		4 years	8 years	12 years	16 years	20 years
Data set (strategy)	**Positive trend detected when unpooled analyses showed no trend**					
1	Unpooled	0	0	0	0	0
2	Equal pools	1.2	0	0	0	0
3	Single + pooled	1.2	0.6	1.3	0	0
4	Pools following weight	0.8	0.6	4	0	0
	Negative trend detected when unpooled analyses showed no trend					
1	Unpooled	0	0	0	0	0
2	Equal pools	1.2	0.6	1.3	0	0
3	Single + pooled	0.4	0	2.7	4.8	0
4	Pools following weight	3.6	1.3	1.3	0	0

3.8.4 Estimating mean concentration for one year

When the data are pooled, one result of pooling is that if the concentrations of individuals have a skewed distribution, the mean concentration based on pooled samples will be biased. In this data set, it is typically the case that many individuals have relatively low concentrations, while a few have much higher concentrations. In this case, the mean concentration based on pooled samples will be positively biased (i.e., higher than the true mean). For this data set, concentrations of individuals seems to be relatively close to a log-normal distribution (one of several types of skewed distributions; however, other distributions may fit even better).

If we call the individual concentrations ***x***, what we generally want to know is the concentration mean(***x***) with a confidence interval. A log-normal distribution means that the log-transformed values *y* (*y* = log(***x***)) are normally distributed. If we analyse the individual concentrations of each fish, the mean of the log-concentration, hereafter denoted μ, can be estimated in the usual way. The standard error is also calculated as usual, so one can easily calculate the confidence interval of mean(*y*), which can be back-transformed to give a confidence interval for the mean concentration mean(***x***).

If the mean concentration is based on *N* pooled samples, where each pooled sample consists of *I* individuals, the ordinary mean is no longer an unbiased estimator of the real mean(y). Instead, there is a bias of

$$Bias = 0.5 * \tau^2(1 - I^{-1}) \qquad \text{(eq. 1)}$$

so the expected mean based on pooled samples is

$$\mu + 0.5 * \tau^2(1 - I^{-1}) \qquad \text{(eq. 2)}$$

where $\tau^2 = I*N*\sigma^2$, where σ^2 is the variance of the pooled samples (Aitchison and Brown 1957, cited in Nicholson & Fryer 1996). Thus, as long as the pooled samples consist of an equal number of fish (*I*), we can estimate the bias and remove it in order to obtain an unbiased estimate of the mean concentration μ.

We tested this approach for estimating the true mean with the data set 2 (equal pools), as we can compare it with the true means estimated from data set 1 (unpooled data). ***Figure 52***

demonstrates that the bias of using the mean of the pooled samples without correcting for the effect of pooling; the mean concentrations are typically 10-30% too high, up to 40% in some cases. After removing the estimated bias (***Figure 53***), the situation is much improved; the bias is strongly reduced and is between -15% and 15% in the majority of the time series. There is room for improvement, though; there is a systematic overcorrection of the bias (i.e., an overestimation of the magnitude of the bias) for some compounds (many of the PCBs) and an undercorrection for others(QCB, HCB). These systematic differences among compounds are probably due to departures from the lognormal distribution. As we have a "data bank" of individual-based data, we can use this to further improve on the bias formula (eq. 1) and thereby reduce the bias even further. However, this assumes that the distribution of the concentrations remain roughly the same over time, if no new individual-based data is collected.

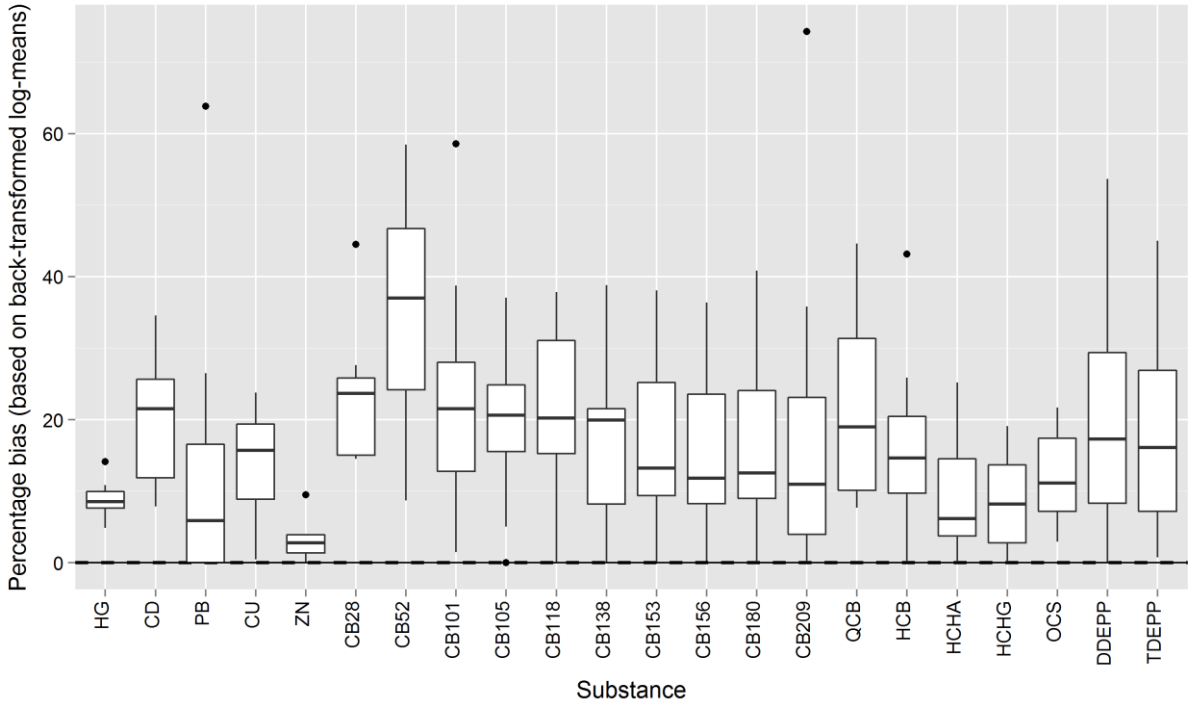


Figure 52*. The bias (the difference between the pooled mean and the individual-based mean, as percentage of the individual-based mean) of estimating mean concentrations using pooled data (equally large pools, data set 2) when we did not attempt to correct the bias. The means of the log-concentrations were back-transformed to actual concentrations, and thereafter we calculated the difference between the means based on data set 2 (the pooled data) and the means based on data set 1 (the original individual-based data).*

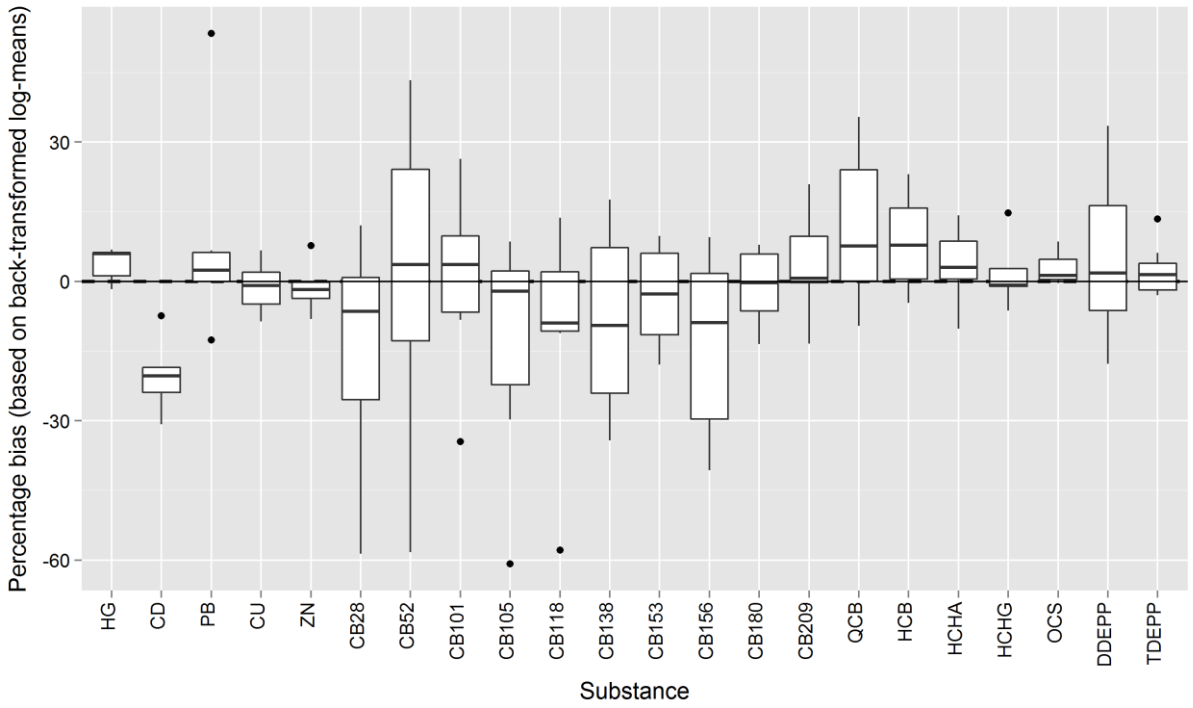


***Figure 53**. As **Figure 52** (data set 2), but after attempting to remove the bias due to pooling using the formulae given in the text.*

Pooled samples of unequal size result in a bias similar to when pools are equally large (***Figure 54***). However, when one tries to use eq. 1 to correct for the bias, the problem is that there is no single value of *I* (number of fish per pool). Using *I* = arithmetic mean of number of fish per pool clearly results in an overcorrection. However, there may still be ways to achieve reliable estimates of mean concentration. Using other values of *I*, for instance the geometric or harmonic mean of number of fish per pool (example in ***Figure 56***) results in a "mean bias" that is close to zero, but the bias of individual compounds is clearly above or below zero. As mention above, we may use historic individual-based data to improve our calculation of the bias, assuming that the distribution of the concentrations remain roughly the same over time. The advantage of this strategy is that some individual-based data is collected, which over time may be enough to check whether concentration distributions stay the same or change. Thus, this strategy may be advantegeous if statistical analysis is performed with care.

If pooled samples are of unequal size, and the division of fish into pools are done systematically with regard to size (some pools with only big fish, some with only small fish), this results in a very large bias for the organic compounds – 20-40% for most organic compounds, and over 50% for some compounds/stations (***Figure 57***). However, it is possible to get rid of most of this bias using the bias correction with the geometric mean of *I*, although the mean cadmium (Cd) is much better left uncorrected (***Figure 58***).

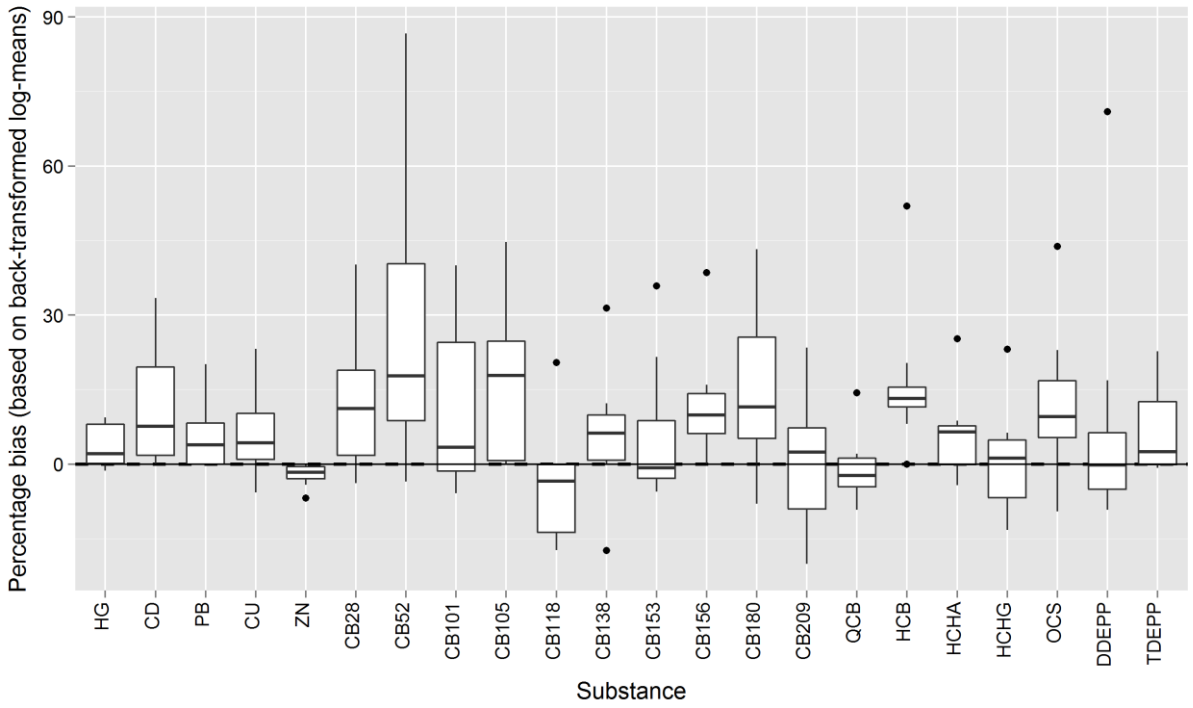


***Figure 54**. Uncorrected bias when the mean concentration is based on unequally pooled samples, in this case the "singles + pools" data (data set 3; 5 single fish and pooled samples of 5 fish each).*

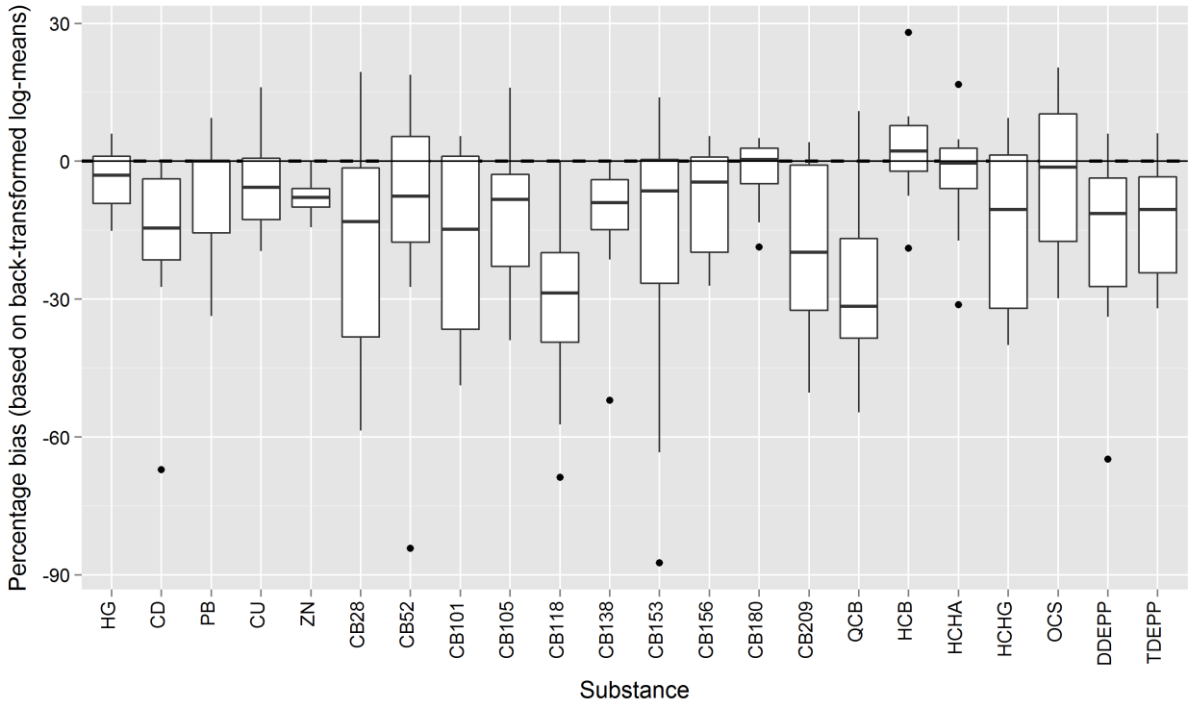


***Figure 55**. As **Figure 54** (data set 3), but after correction of bias using the formulae used in the text. As there was no single value of I (number of fish per pool), we set I to the arithmetic mean of number of fish per pool.*

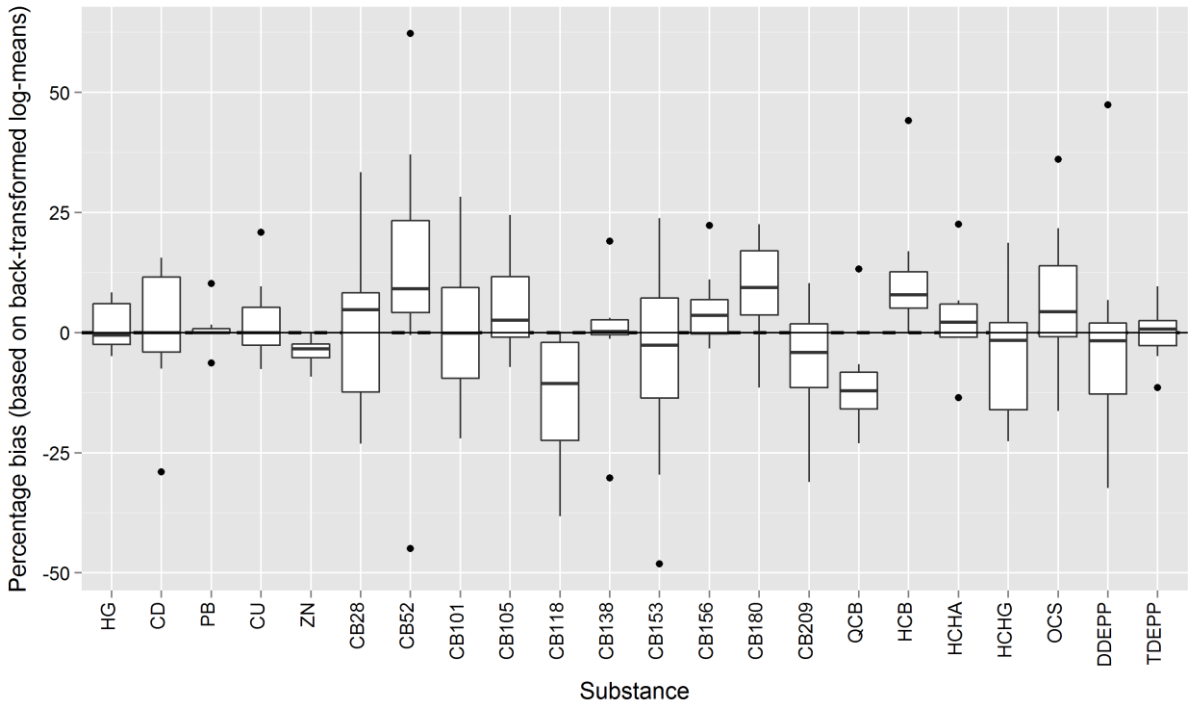


***Figure 56**. As **Figure 55** (data set 3), but we have attempted to remove the bias in a different way, namely by setting I equal to the harmonic mean of number of fish per pool.*

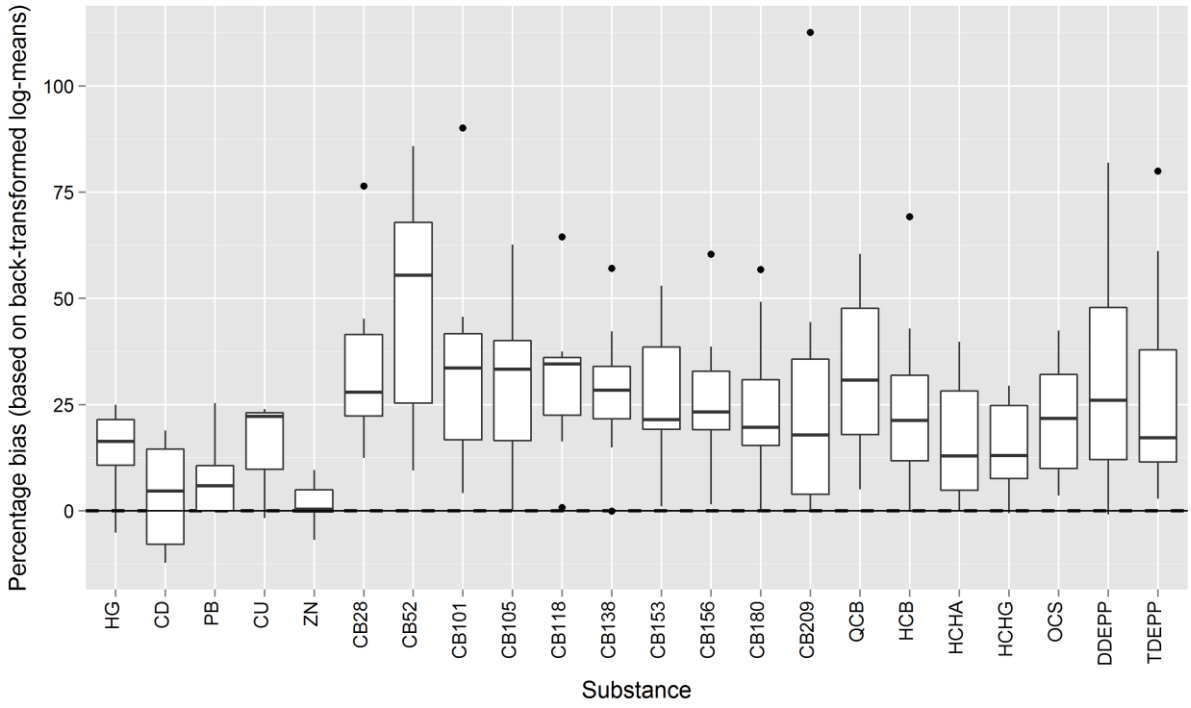


***Figure 57**. Uncorrected bias when the mean concentration is based on unequally pooled samples, in this case the "pools following weight" data (data set 4).*

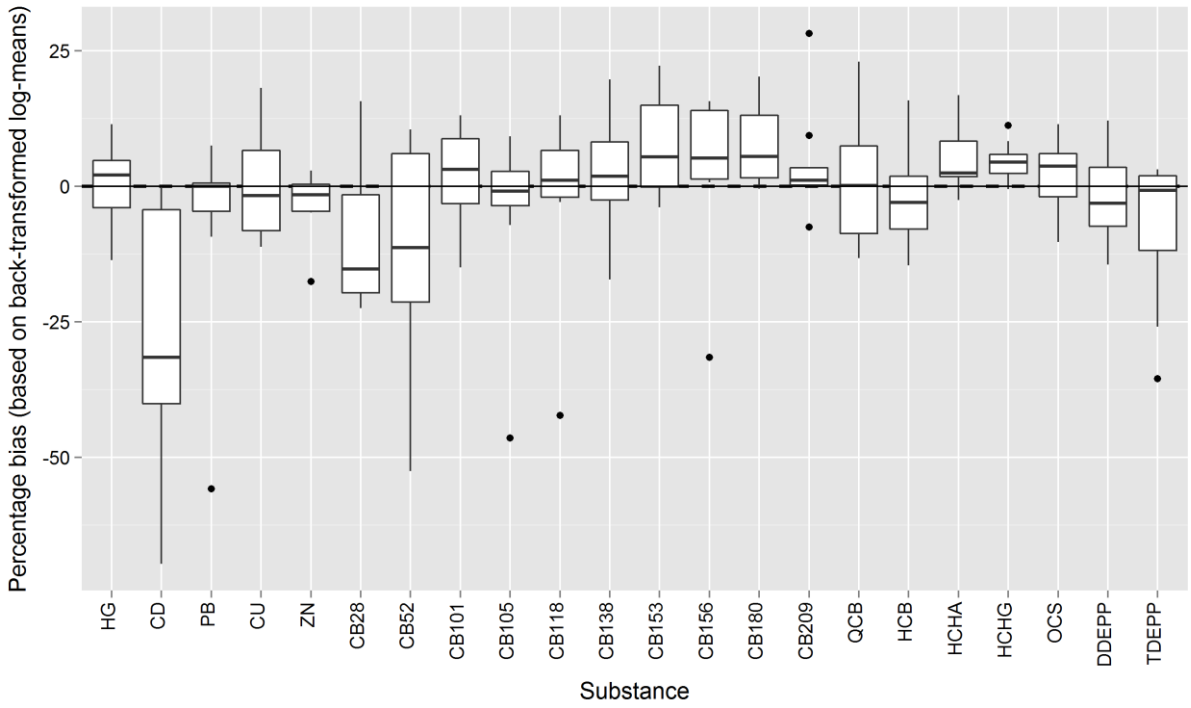


***Figure 58**. As **Figure 57** (data set 4), but after correction of bias, setting I to the geometric mean of number of fish per pool.*

3.8.5 Discussion and conclusion

Pooling samples leads to a lower number of samples to analyze and a corresponding reduction in cost. The disadvantage is that this leads to lower power (i.e., detecting time trends is less likely or takes a longer time series) and a bias (leading to too high estimates of mean concentrations). Comparing pools of the same size and pools of unequal size, the former strategy more effectively picks up time trends (compared to the cost of the strategy) and the bias of concentration estimates can more easily be corrected. From this perspective, one would clearly say that samples of equal size should be favoured. This is in line with Nicholson & Fryer (1996), who stated that "if the numbers of individuals per pool or the weights of the individuals in a pool vary, then the bias will vary from pool to pool and the estimation of the mean log-concentration becomes even more complicated. Inconsistent pooling should be avoided, if possible." There is one advantage of using unequal pool size of a specific kind, namely "singles + pools". In this case, the samples from individual fish can be used to state whether the assumed distribution of the statistical analysis (still) holds true. The inclusion of some single samples my make it easier to detect whether the actual distribution changes over time and starts to depart substantially from the assumed distribution (e.g., log-normal), although massive departures probably also can be inferred from pooled samples.

3.9 Note on methods: Small cod samples

3.9.1 Background

Cod (and cod liver) may not in the long-term represent a suitable monitoring matrix for contaminant monitoring along the Norwegian coastline. Basic requirements to be considered when selecting species for monitoring (MacGregor *et al*. 2010, OSPAR 2012, European Commission 2014):

- Widespread and abundant throughout the study area;
- Eurytopic (i.e. be able to adapt and thrive in a wide range of environments) and have a wide distribution throughout the country in which the monitoring is being undertaken; though use of multiple species is likely to be necessary, attempts should still be made to use common species where possible to minimise complexity.
- Relatively sedentary, and thus reflecting the local concentration of contaminants;
- Sufficiently long-lived for bioaccumulation of contaminants to occur;
- Of sufficient size to yield enough tissue for analysis;
- Of no significant conservation or socio-economic interest, or otherwise protected by legislation;
- Of a size and trophic level that is relevant to the protection goal, where possible.

It is becoming increasingly challenging to achieve required analyses from the amount of cod liver tissue that can be practically collected at a specific monitoring station or area. Early in the programme, during the 1980s and 1990s, analyses of metals, PCBs and pesticides was the focus of monitoring of cod. MILKYS monitoring on cod caught in 2014 included additional analyses of PBDEs, HBCDs, PFCs, chlorinated paraffins, PFRs, alkylphenols, TBBpa and BPA (***Table 2***). This increase in required parameters has necessitated the need to use different laboratories which has impacted the need for more material despite improvements in analytical methods since 1980s.

This section considers alternative approaches for gaining the required amount of tissue for performing the assessment of the environment. For example, we can extend the catch period or the catch area to get a large sample to choose from? We might also consider if there are other tissues or species that are suitable? Lowering the analytical limit of detection might also be an option? Passive sampling may also be a supplement?

Biomonitoring is generally used for:

(i) water/sediment chemical quality assessment (i.e. comparisons with established EQS or to distinguish sites of contrasting water quality statuses),
(ii) the monitoring of temporal trends in contaminant concentrations,
(iii) (iii) consumer safety (predators and humans).

With regards to water quality assessment, we have to assume that concentrations of contaminants in cod (liver) reflect contaminant levels in the environment the fish lives in if we want to infer water/sediment chemical quality status from these measurements. For WFD priority substances with newly established EQS_{biota}, protection goals are set in terms of predator and secondary poisoning as well as of human exposure through food consumption.

When the monitoring goal is to assess the chemicals quality of a water body, it is often performed through the comparison of environmental data with environmental assessment criteria such as the Norwegian Environmental Agency classification system or EU's EQS values. A direct link is therefore needed between the contaminant level in the environment and that in the matrix being monitored. Often it is the most contaminated medium which is monitored.

No matter which monitoring method is selected it is important to maintain adequate quality assurance and control. For example in regards to sampling, there is a complex of influencial factors such as the age of the fish, its size and trophic level or the period during which the sampling is undertaken that can affect contaminant body burden significantly. In addition, the collection of specific organs or tissues from an organism can also affect the results and standardization is also needed.

3.9.2 Extending sampling time

Whether sampling is extended in either time, space (section 3.9.3) or both, the effects on the estimated concentrations and assessment of trends depends on the variance in concentrations on several levels:

(i) variance among fish within one site/date,
(ii) variation among dates and among sites,
(iii) whether year-to-year trends are similar/consistent among sites.

If the variance is increased by extending sampling in time and/or space, standard errors of the estimated concentrations and trends may increase. On the other hand, such an increase in standard errors may actually give a more realistic picture of the actual uncertainty. This is the case if on samples fish from a single site/date, but the fish tends to be similar, for instance, if the fish in the sample have a similar size/age, or have tended to feed in the same locations in the past. In this case, the estimated standard error will be too small and any estimated trends may be specific for the site sampled, and not real for the area the sample is supposed to represent.

VIC data revisited

We have some information on the effect of extending sampling time and areas from a special sampling program, the voluntary international contaminant-monitoring programme (VIC), which was carried out for three years in the late 1990s. In this project (Bjerkeng & Green 1999; Green *et al*. 2000), JAMP station 30B (Oslo City Area) was sampled in 3 different sites at around the same date. In addition, for one of the sites (Slemmestad-Måsane), samples were taken at 2 additional dates (***Figure 59***). This scheme was followed for 3 years, 1997-1999 (in January each year, so the samples belong to JAMP years 1996-1998). Similar but less extensive sampling schemes were followed for station 53B Sørfjordn (2 sites, and 2 dates in one of the three years) and 67B Hardangefjord (one site, 2 dates in one of the years). (Similar schemes were followed by Netherlands and Sweden, but for flounder and herring, respectively.)

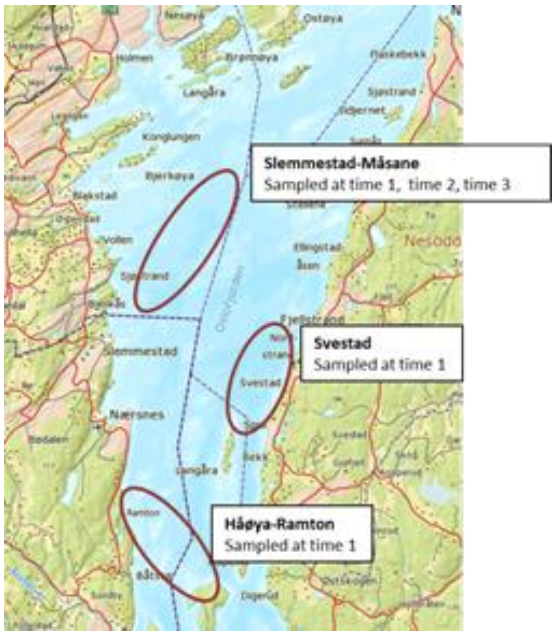


***Figure 59**. Sampling scheme of the VIC programme at station st. 30B in the Inner Oslofjord for each of the years 1997-1999 (JAMP years 1996-1998). In each year, Slemmestad-Måsane was sampled at three occasions. At one of these sampling occasions (denoted "time-1"); samples were also collected at the two other sites. (Note that "time 1" was not always the first date). The span of the dates was from 14 January to 3 February.*

Bjerkeng (2000) analysed these data with respect to two organic contaminants (CB153 and pp'DDE) and three metals (cadmium, mercury and zinc). He concluded that for organic contaminants, the between-site and time variance component is at least 20 % of the between-specimen variance for samples taken at the same site and time, and that "by distributing the 25 fish on 3 samples, the variance of the annual mean estimate could be reduced by 65 %". In contrast, for metals there was no statistically significant small-scale variation, and the conclusion was that the total number of fish sampled was most important (however, inside the report it says that "the indication of systematic location differences could be taken as a warning against using samples from different locations, even within an area of a few kilometres").

For the purpose of the present report, we have revisited the 1997-1999 data from station 30B used in the VIC report of 2000. Whereas Bjerkeng analysed 5 contaminants, we analysed the concentrations of all contaminants based on single-fish measurements, i.e. 17 organic contaminants (including 10 PCBs) and 5 metals. We first checked whether including variation among dates (as a so-called random effect) in the statistical models improved model fit. For 15 of the 22 contaminants (including 13 of the 17 of the organic contaminants), the optimal model includes variation among dates within each year. This indicates that for organic contaminants, fish sampled on the same date were more similar than fish sampled on different dates, although there was not more than 19 days from the first to the last sampling date in any year.

Using this data set, we draw subsets of the data using different strategies in ordert to simulate how sampling strategy affects estimates and standard errors. In all cases, we took into account the length of the fish as a covariate. First, in order to establish a baseline to compare with, we analysed all the avaliable data from the VIC samples, i.e. 50 samples per year. This is an unrealistically big sample, but is used as the "truth" to compare smaller samples with. In ***Figure 60***, we use the model using all the data to predict tissue concentrations at Slemmestad-Måsane each year, and compare these predictions to predictions using the data only from the Slemmestad-Måsane site at only one date per year. As the figure shows, the results are comparable in some cases (CB105, DDEPP); in other cases, the estimates based on the smaller sample have larger confidence intervals (e.g., Pb); in still other cases, the small-sample estimates actually have

smaller confidence intervals than the large-sample estimates (e.g. CB52). The latter examplifies that the width of the confidence interval may be misleading when data from a single date/site is used, probably due to lack of independence among individual fish in the sample. The figure also shows that in some cases, the small-sample estimates are quite biased compared to the "real" estimates, e.g. Cd, Cu, Zn and HCHA in 1996.

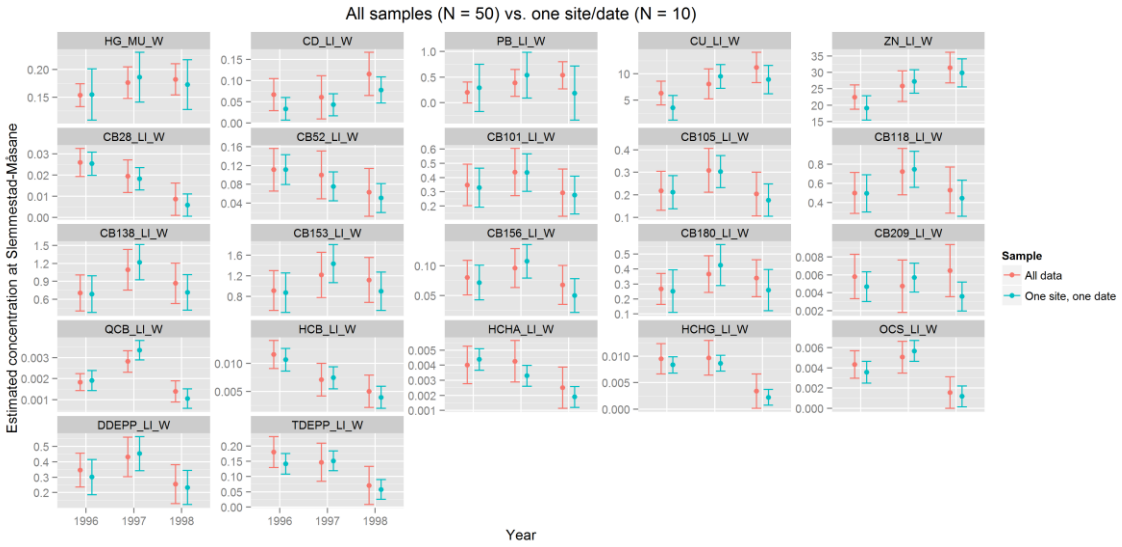


***Figure 60**. Comparing the estimates of concentrations using all data (N = 50 per year per compound) to estimates using only data from a single site (Slemmestad-Måsane) and date per year (1997.01.15, 1998.01.15, and 1999.01.21). The predicted "all data"-estimates are for the Slemmestad-Måsane site (meaning that data from all sites were used ini the estimation, but corrected for general differences among sites).*

Moving over to the simulation of extending sampling in time, we used data from the Slemmestad-Måsane site to compare sampling 10 fish/year at a single date (here, the same date that was used for the other sites) with sampling 10 fish/year at a several dates (by picking 10 random fish from all dates). Thus the total sample size stayed the same (10 fish). The results (***Figure 61***) show that sampling at several dates in some cases results in very similar resurs, while in other cases, the confidence intervals increase considerably. However, the cases where extended sampling in time increases the confidence interval – such as Cd, Cu, Zn and HCHA – are the cases where the results from the single-date sampling were clearly biased compared to the "truth" (***Figure 60***). In some cases, the point estimates from several dates are also less biased (Cd, Cu, Zn in 1998). Thus, the increase in the confidence intervals **from** the extended sampling reveals that the smaller confidence intervals at a single date was misleading, resulting from the fact that the fish sampled at a single date was more similar to each other (with respect to concentrations) than fish sampled at different dates.

The conclusions from this analysis indicates that extending the sampling time in some cases does not increase the uncertainty of the estimates. In other cases the uncertainty of the estimates increases, but this is only because single-date fish appeared to not be truly independent, causing the uncertainty of the single-date datasets to be underestimated. Thus, these data indicates that a moderate extension of sampling time has either neutral or positive effects on analysis, and makes the analysis more robust by decreasing the risk that statistical analysis erronously indicates time trends. These results adds to the previous analysis of these data (Bjerkeng 2000) by using a

different (and complementary) approach and by increasing the number of contaminants analysed from 5 to 22.

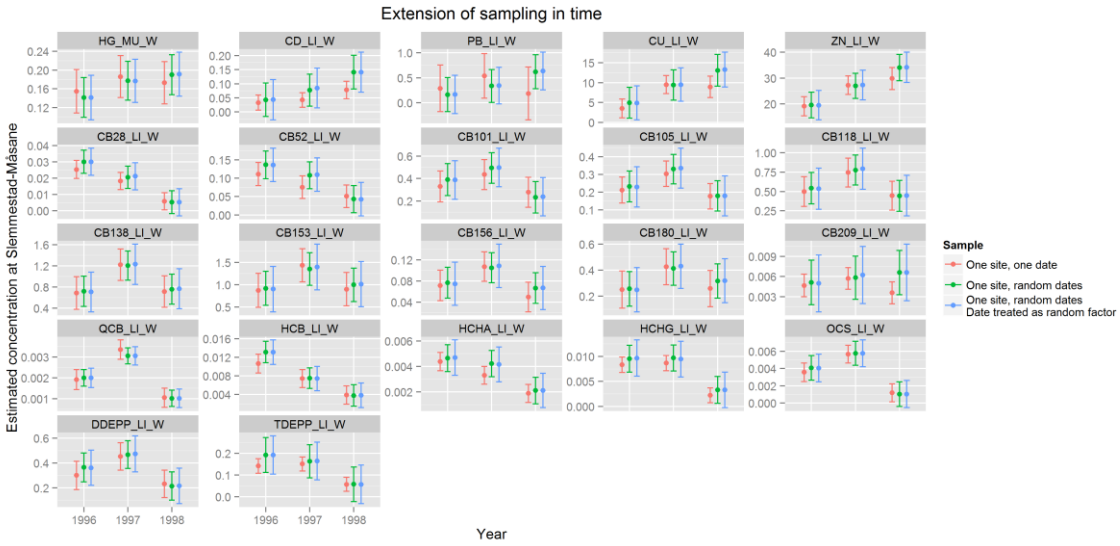


***Figure 61**. Increasing sampling time from sampling on a single date to sampling over 14-19 days per year. In all cases the samples were taken from the site Slemmestad-Måsane and the total sample size was N=10 per year. In the "random date" cases, the random draws were performed 10 times and we used the mean of the point estimates and the standard errors. For the "random date" cases, we show two sets of estimates (using exactly the same data): in one case, we analysed the data ignoring the information about the samples' date, in the other case, date is treated as a random factor in the statistical analysis.*

3.9.3 Extending sampling area

VIC data revisited

Also for this issue, we used the data set from the VIC sampling of station 30B, described in the previous section. The analysis of the full data set showed that when year was taken as a continuous variable (i.e., we were testing for time trends over the three years of the study), we found a significant ($P < 0.05$) interaction between site and year for a majority of the organic compounds (13 of 17) and for 2 of the 5 metals. Thus, sampling from a single site increases the probability that the analysis indicates a time trend which is not representative for the area at large.

As in the previous section, we used randomly drawn subsets of the full data set to analyse the effect of extending sample area, in a similar way as we did for times in the previous section. Again we kept the total sample size constant (N = 10), but used either the samples from one site (Slemmestad-Måsane) at one time, or 10 fish were picked at random from the three sites (but still keeping to the same date or as close to that date as possible). In a similar way as when sampling time was extended, the confidence interval became broader for some compounds, but mostly in the cases where the single site/time estimates were biased and with underestimated uncertainty (***Figure 62***). Thus, increasing the sampling area tends to make the analysis more robust.

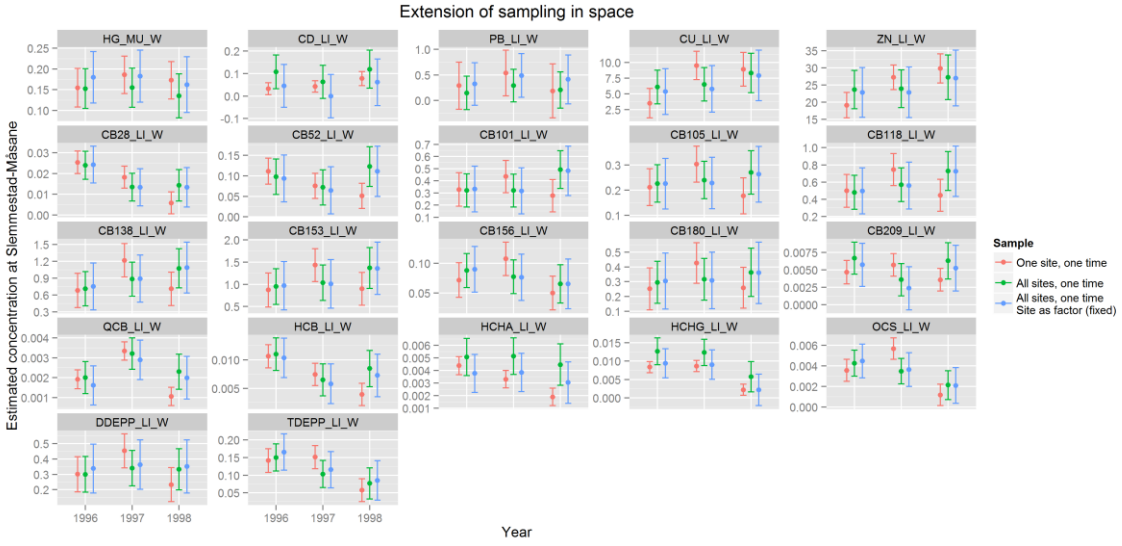


Figure 62*. Increasing sampling area from a single site (Slemmestad-Måsane) to all three sites. In all cases the samples were taken at the same date as at Slemmestad-Måsane or as close to this date as possible, and the total sample size was N=10 per year. In the "all sites" cases, the random draws were performed 10 times and we used the mean of the point estimates and the standard errors. For the "all sites" cases, we used the same data to perform two kinds of analyses: ignoring the information about the samples' site, and treating site as a fixed factor in the statistical analysis.*

Finally, one can extend the analysis to include both a larger time span and sevaral sampling sites. The results (***Figure 63***) show in some cases a strong increase in estimate uncertainty, especially when attempting to include both date and site in the statistical analysis (which is not surprising, taking into account that there were 5 combinations of site and time, while sample size was N=10 per year, leaving on average only 2 samples per site/time combination). However, in general, the conclusions from the previous analyses holds: the increase in the confidence intervals reflects mostly that the single site/time sample for some compounds showed lack of independence among fish and lack of representativity for the total sampling area.

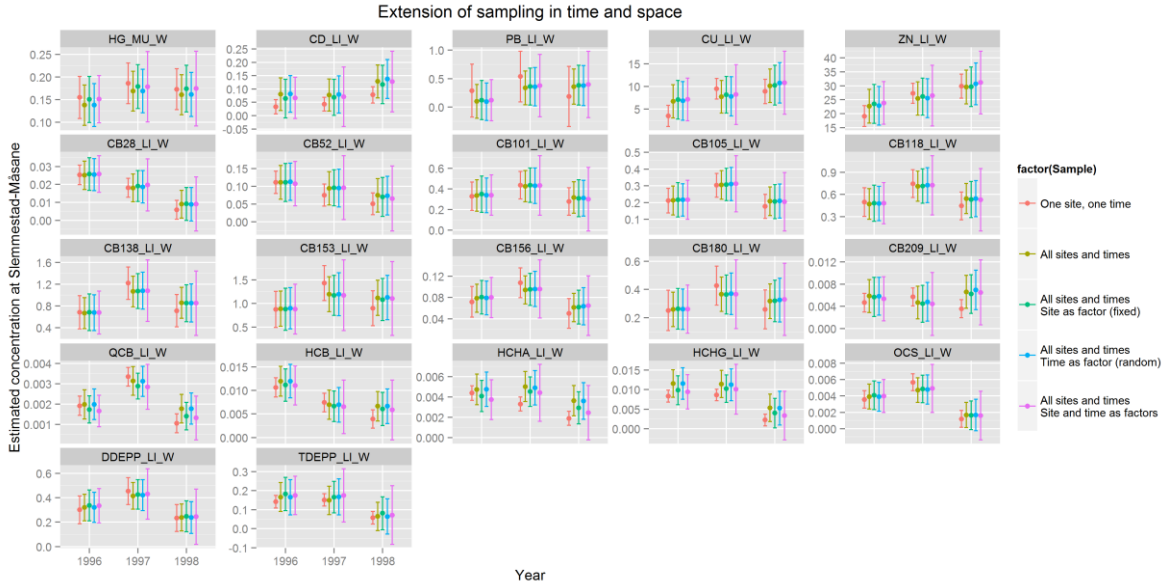


Figure 63. *Increasing both sampling area (from a single site to three sites) and sampling time (from a single day to 14-19 days). In all cases the total sample size was N=10 per year. In the "all sites and times" cases, the random draws were performed 10 times and we used the mean of the point estimates and the standard errors. For the "all sites and times" data, estimates are shown for four types of analyses: (1) ignoring the information on site and time, (2) ignoring time info but including the samples' site, (3) ignoring site info but including the samples' time, and (4) including both time (as a random factor) and site (as a fixed factor).*

Analysis of stations not affected by pollution

The focus of the VIC programme was on contaminated areas. A question remains as whether or not the conclusions from revisiting VIC data would apply to stations not as impacted by contamination. To do this we analysed data from 5 MILKYS stations in two areas: (1) the exposed waters of southern Norway (Færder -st. 36B and the Lista area - st. 15B) and (2) in Western/Northern Norway (Bømlo – st. 23B, Lofoten st. 98B1, Varangerfjord st. 10B). Sample pollutants were Hg, Cd, Pb, CB153, BDE-47 and PFOS.

While it could be expected that the stations had relatively similar trends within each area, plots of the raw data appeared to show that trends were quite different in some cases. For instance, in station 15B, Hg showed a distinct pattern of decrease during the early 2000s and an increase after 2005, while 23B showed a steady increase (***Figure 64***). However, trends in e.g. fish length may affect these patterns in the raw data

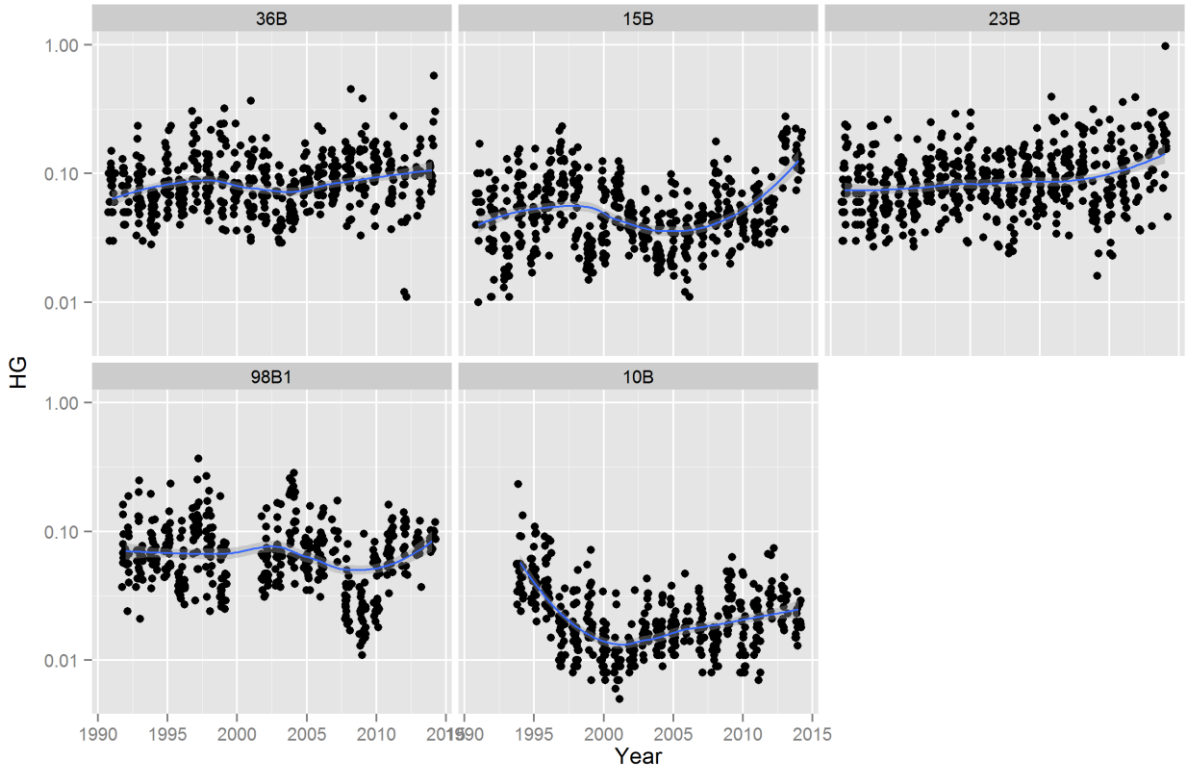


***Figure 64**. Trends of Hg in 5 stations assumed to be not affected by pollution, the first two (36B, 15B) in the Skagerrak area, the others three in Western/Northern Norway.*

We performed a somewhat more rigourous test by attempting to remove the effects of length by fitting four linear models for each compound. For each model, the response variable was log(concentration + 0.1). We assumed independence among individuals and years. The four models were

1) all 2-way interactions between fish length, station and year (year treated as a categorical variable)
2) effects of station, year and their interaction (allowing for time patterns to differ between each station), and an additional (not interacting) effect of length
3) effects of station and the interaction area*year, where area was either Skagerrak or West/North Norway (allowing for time patterns to differ between these two areas); an additional (not interacting) effect of length
4) independent effects of station, year and length

For each compound, the models were compared using Akaikes Information Criterion (AIC), which measures the fit of the models to the data, but gives more complex models a "penalty". A lower value indicates a more "optimal" model (better fit while not being too complex). We found that for 2 of the 6 pollutants (Cd and Hg), model 1 was the best model, meaning that the effect of length on concentrations differed among stations (***Figure 65***). Thus, we cannot easily "remove" the effect of length. If we – for now – disregard this interaction and assume length to have the same effect in all stations (i.e., we ignore model 1 and focus on models 2-4), we found that model 2 better than model 3 and 4 for all pollutants with the exception of BDE-47. That is, the time pattern of concentrations differed among stations, and they also differed among stations with in the same area (as model 2 was generally better than model 3). In the case of BDE-47, model 3 was the best model, i.e., the development of concentrations over time was different among areas but similar within each area. The predicted log-concentration for a 50 cm fish (close to the median size of the target length groups given in ***Table 1***), PCB-153 and BDE-47 in ***Figure 66***. This figure clearly shows

the parallel development of the BDE-47 (bottom); however, it should be noted that although the trends within each area are similar, the general *level* of contamination varies quite a lot between stations within the same area; e.g. 15B is about 1 unit higher on the log-scale than 36B, meaning that the concentrations are approximately 2.5-3 times higher. In the case of Hg, it also shows that although stations still have different trends in the development (***Figure 66***), much of the apparent differences in time trends shown by the first figure (***Figure 64***) have been removed by adjusting for length: all stations show a long-term decline since 1995.

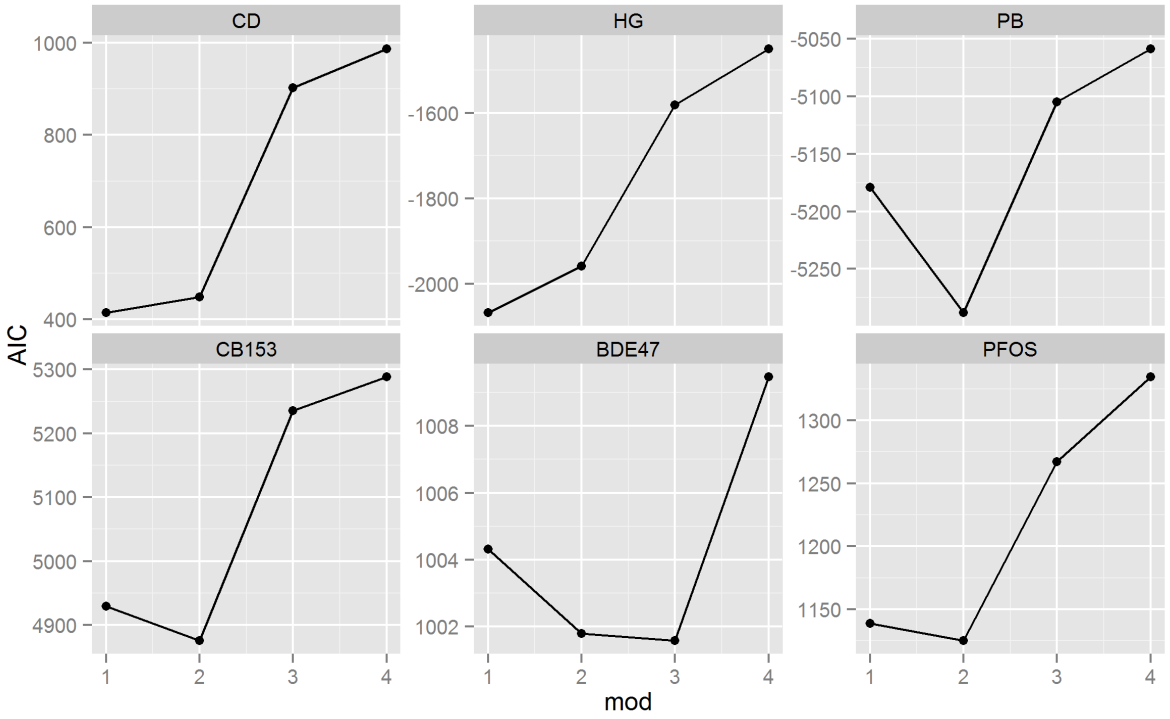
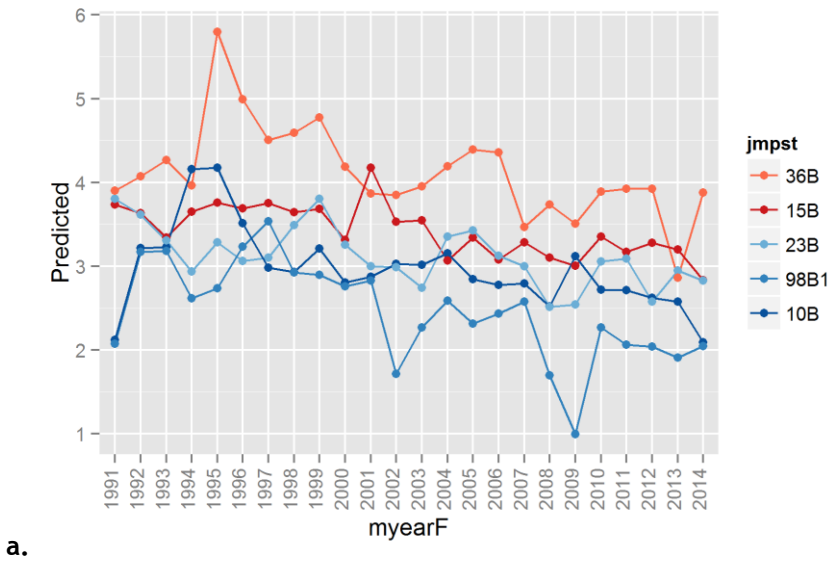
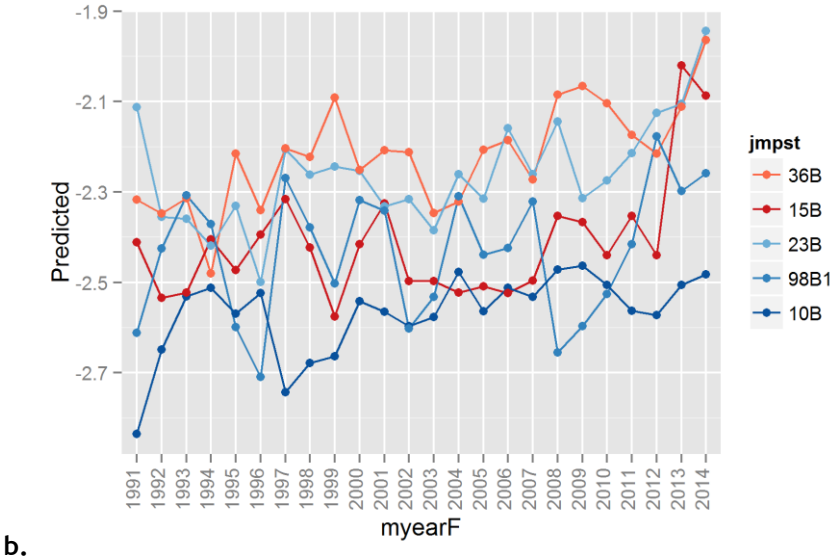


Figure 65. *AIC values for models (mod) 1-4 for 5 stations assumed to be not affected by pollution. For models 1-4, see text. Year is a categorical variable.*



a.



b.

c.

***Figure 66**. Predicted annual mean log(concentration + 0.1) for mercury (a.), PCB153 (b.) and BDE-47 (c.). The effect of length has been removed by using the predicted concentrations for 50 cm cod. The red lines are for stations in the Skagerrak area, the blue ones for West/North Norway.*

If we analyse 20- and 10-year trends in a similar manner (disregarding model 1), we got a somewhat different result: For 20-year trends, for 2 of the six parameters, Hg and BDE.47, the time trend does not differ significantly among stations within areas (for Hg, time trends doesn't even differ between areas) (***Figure* 67**). For 10-year trends, for 3 of the six parameters, Pb, BDE.47 and CB-153, the time trend does not differ significantly among stations within areas (for CB-153, time trends doesn't even differ between areas) (***Figure 68***). This may indicate that the station*year interaction to some degree is caused by single years.

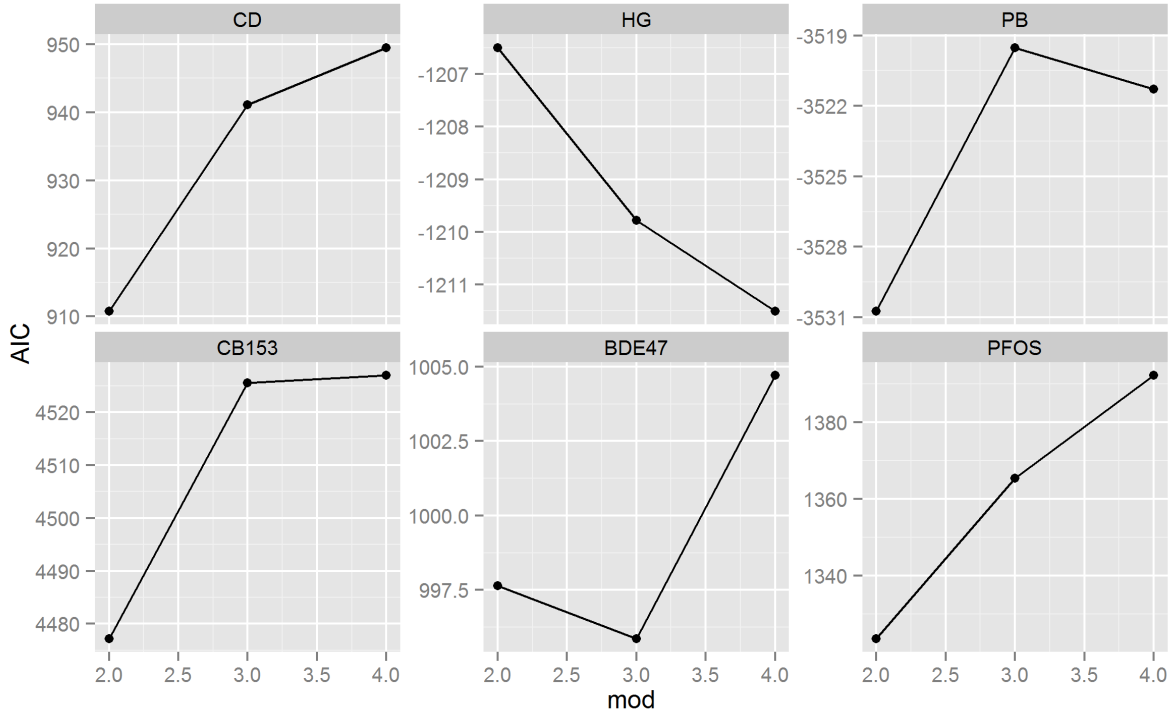


***Figure 67**. AIC values for models (mod) 2-4 for the 5 stations assumed to be not affected by pollution, in analysis of 20-year time trends. Year is a continuous variable.*

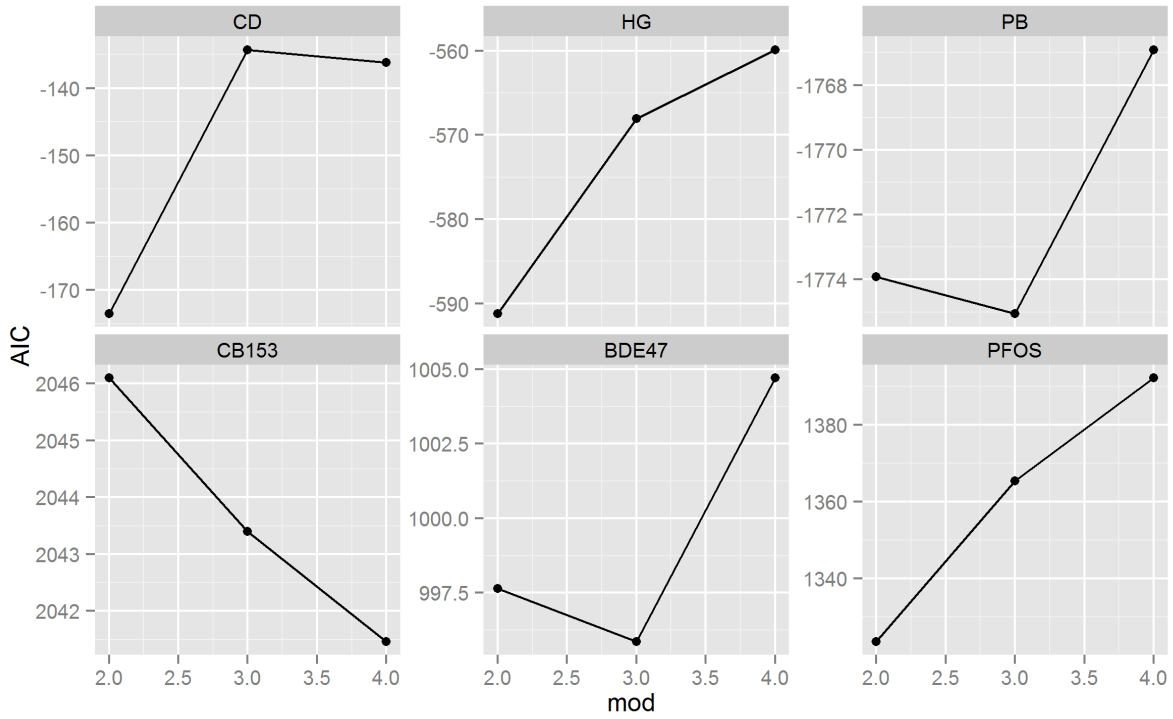


***Figure 68**. AIC values for models (mod) 2-4 for the 5 stations assumed to be not affected by pollution, in analysis of 10-year time trends. Year is a continuous variable.*

Analysis of stations affected by pollution – Oslo fjord area

The stations analysed here were all in the Oslo fjord area (stations 30B, 36B, 02B and 71B). Here we only checked whether time trends were similar or dissimilar among stations when we analyse time trends for the last 20, 10 or 5 years. In all cases, a linear effect of fish length was taken into account in the model (i.e., removing the fish length effect). For the 20-year time trends, the trends were significantly different among stations in all cases except for lead (***Table 30***). The 10-year trends, however, were similar among stations for 4 out of 6 compounds. For the 5-year trends, there were little or no evidence for 5 of the 6 compounds that trends differed among stations (only in the case of BDE-47, dAIC was >2 for the "Site and year independent" model).

***Table 30**. Test of independence (AIC values) among pollution-affected stations in the Oslofjord area (30B, 36B, 02B and 71B), and for different lengths of the time period (the whole period as well as the last 20, 10 and 5 years). For each parameter and period, the table shows dAIC (AIC minus the lowest AIC for the two models). Zero means that this model is the optimal; however, models with dAIC values <2 are considered to be more or less equally good as the optimal model.*

		dAIC		
Parameter	**Model**	**20 years**	**10 years**	**5 years**
CD	Station*year interaction	0	4.955	5.734
CD	Site and year independent	8.088	0	0
HG	Station*year interaction	0	2.393	0
HG	Site and year independent	6.701	0	1.684
PB	Station*year interaction	5.836	2.505	5.033
PB	Site and year independent	0	0	0
CB153	Station*year interaction	0	0.65	0
CB153	Site and year independent	37.971	0	0.433
BDE47	Station*year interaction	0	0	0
BDE47	Site and year independent	3.448	3.448	6.107
PFOS	Station*year interaction	0	0	1.783
PFOS	Site and year independent	2.376	2.376	0

Extending sampling area – conclusions

We looked at differences in concentrations among sample locations on two scales: extensions from one to several locations within one station (e.g., the VIC data: several locations within the Oslo City area, station 30B), and extensions to several locations within a larger area (e.g., the open Skagerrak area). In both cases, we found interactions between sites and trends in some cases. In the first case, we conclude that the presence of local-scale variations in trends may be misleading if sampling is very localized. In the second case, we found that time patterns and trends may be quite variable among stations in the same general area, even when the stations are classified as "not affected by pollution". Even in the single case where trends where similar (BDE-47), the levels of contamination were very different among stations (***Figure 66***).

3.9.4 Alternative tissue to cod liver

Good correlation may allow calculation of concentrations in one tissue from concentrations measured in other tissue, e.g. if the size of the liver is insufficient. Comparison of concentrations in cod liver and fillet were done for PCBs, PBDEs, HBCDs, SCCPs, MCCPs, PFRs, BPA, TBBPA and alkylphenols from the same individuals collected in 2012 (Green *et al.* 2014).

Concentrations of PCB-congeners in liver from the Inner Oslofjord (n=12), Outer Oslofjord (n=4) and Kristiansand harbour (n=9) were consistently higher than in corresponding fillet, with a factor of 207-219 higher on a wet weight basis, and a factor of 3.3 on a lipid basis. The correlation was considerably better (r^2=0.69-0.98) on lipid weight basis than wet weight basis (r^2=0.05-0.88). The ratios compared reasonably well to earlier investigations (Green & Knutzen 2003). Concentrations of PBDE-congeners in liver in the same individuals were consistently higher than fillet, on an average of 247 and 191 times higher for BDE47 and BDE100 on a wet weight basis, respectively and showing a similar distribution to PCB.

Comparison of concentrations of HBCD from the Inner Oslofjord (n=1) and Karihavet (n=1) indicated that HBCD found in liver was two to four orders of magnitude higher than fillet on a wet weight

basis. Concentrations of SCCP and MCCP were higher in liver from the Inner Trondheimsfjord (n=3) and Tromsø harbour (n=8) than fillet (by a factor of 3.9 and 4.5, respectively on a wet weight basis) and higher concentrations in liver corresponded to lower concentrations in fillet. Except for TCPP, concentrations of PFRs from Tromsø harbour (n=10) were below the limit of detection for both liver and fillet. The limit of detection for liver was higher than for fillet. Concentrations of BPA in liver from the Inner Oslofjord (n=3), the Grenlandsfjord (n=2), Karihavet (n=5) and Inner Trondheimsfjord (n=4) were 2-153 times higher than fillet on wet weight basis and all concentrations in fillet were below the limit of detection. Concentrations of TBBPA from the same individuals were below the limit of detection for both liver and fillet, with one exception. Levels of alkylphenols in fillet from the Inner Oslofjord (n=3), the Grenlandsfjord (n=3), Karihavet (n=5) and the inner Trondheimsfjord (n=8), Tromsø harbour (n=6) were all below the limit of detection.

Comparison of concentrations in cod liver, fillet, blood and bile

Levels and distributions of PFAS in cod tissues from the Inner Oslofjord in 2009, 2011 and 2012 were estimated by Rundberget *et al.* (2014). PFOSA was the most abundant PFAS compound detected in blood (65-310 ng/g), while PFOS and PFOSA were more evenly distributed in liver (3-33 ng/g). PFOS was abundant (12-66 ng/g) in bile, while PFOSA was mainly below LOD (0.5 ng/g). Both PFOS and PFOSA were detected (0.5-3.9 ng/g) in fillet. PFAS such as PFNA, PFDA, PFUnDA, PFDoDA, PFTrDA, PFTeDA, PFBS, PFHxS, PFDS, and PFDoS were also found at various levels in blood by using HRMS (ToF) with lower detection limit. Blood was the preferred matrix for analysing PFAS in cod. PFOS were roughly at the same levels in blood as in liver and bile, but levels of other PFAS were higher and therefore easier to detect. The analysis clean-up of blood is easier than e.g. liver.

Conclusions

Concentrations of PCBs, SCCP and MCCP in cod liver were well correlated with concentrations in fillet, but no strong conclusions could be drawn for correlations of PBDEs, α-HBCD, PFRs or bisphenol A (Green *et al.* 2014). Analysis of fillet cannot replace analysis of liver regarding PFAS, but blood can (Rundberget *et al.* 2014). A more thorough investigation is needed before it can be recommended to replace analyses of liver with analyses of fillet (Green *et al.* 2014).

Tissue versus whole fish

The EQS for fish are based on analyses on whole fish (2013/39/EU).Therefore, the EQS cannot be directly compared to concentrations found in certain tissues. Converting concentrations in fillet to concentrations in whole fish is uncertain. Skeletal muscle is e.g. the largest tissue in the body, 40-60 % of the total live weight (Houlihan *et al.* 1988).

3.9.5 Use of other species

Experience from the collection of cod for environmental monitoring purposes the later years has presented challenges regarding obtaining the optimal number and sizes of fish at the various locations. Results from the Institute of Marine Research also suggest future difficulties in obtaining sufficient number of cod, as recruitment is weak at several areas along the Norwegian coast (Espeland *et al.* 2014). Therefore, it may be necessary to have a backup sampling protocol in terms of alternative species as supplement, or substitute for cod.

Alternative species should resemble cod as much as possible in terms of biology and propagation, thus members of the *Gadidae* family are relevant candidates. In the following are six species suggested (not prioritized order) and important characteristics (Pethon 1989) are presented, so that advantages and disadvantages ('pros & cons') are clear. All of these species spawns during spring, which favours sampling in the autumn. There is an ongoing process to contact fishermen along the coast to learn about experiences with catch/bycatch of these species.

Norway pout (Norwegian: *øyepål*) (*Trisopterus esmarkii*): Found along the entire coast of Norway. Will not reach more than 25 cm of size (and rarely more than 13-19 cm). This suggests a limited amount of available material (such as liver) from individual fish. Common at 80-300m depth and feeds mainly on crustaceans. Benthic (soft bottom), but may also live pelagic. Spawns January to July.

Poor cod (Norwegian: *sypike*)(*Trisopterus minutus*): Not very abundant in the North Trøndelag County, but is more common along the Norweagian coast south of this. It may reach 30 cm of size (but rarely more than 25 cm). Lives at 10-300m depth, most commonly down to 100 m. Feeds mainly on crustaceans and small fish (mostly gobys, *Gobiidae*). Benthic, but also benthopelagic. Spawns spring to early summer.

Blue whiting (Norwegian: *kolmule*) (*Micromesistius poutassou*): Propagated along the entire coast of Norway. May reach 50 cm of size. Mesopelagic, common at 200-500m depth (but may live down to approximately 600-700 m), and performs diurnal vertical migration. Feeds mainly on krill (euphasiids) and other zooplankton organisms, as well as small fish. Spawns March to April.

Whiting (Norwegian: *hvitting*) (*Merlangius merlangus*): Very common along the Norwegian coast north to Stadt, but les common further north. May reach >50 cm of size. Benthic (10-200m depth), but may live pelagic. Feeds mainly on crustaceans and fish (such as sand eel, *Ammodytidae*, and herring, *Clupea Harengus*). Spawns January to July. The youngest speciments are commonly found close to shore, while older individals are more common further off the coast. Whiting migrates, however, the migration pattern is not well known.

Haddock (Norwegian: *hyse*) (*Malanogrammus aeglefinus*): Propagated along the entire coast of Norway. May reach >100 cm of size (but rarely more than 80 cm). This species is benthic (soft bottom) and most common at 40-300m depth. Feeds mainly on roe (especially of herring), fish, and benthic organisms (such as polychaetes, crustaceans, gastropods and echinoderms). Spawns March to June. Haddock conducts long migration, however, the migration pattern is not well known.

Pollock/Saithe (Norwegian: *sei*) (*Pollachius virens*): Propagated along the entire coast of Norway. May reach >120 cm of size. Both benthic (soft bottom) and pelagic. Common at 0-300m depth, both close to shore and further off the coast. The largest specimens are most ommen in deeper waters and feeds on fish (such as Herring and European sprat, *Sprattus sprattus*) and pelagic crustaceans. Spawns January to April.

Conclusions

Six candidates are suggested as possible alternative to cod. There is a need to know more about the availability of these species and how concentrations would relate to those found in cod.

3.9.6 Lowering the Limit of Quantification (LOQ)

The ongoing development of chemical analyses may result in a reduction of sample material required. The sample amount available for analysis is an important factor in determining limit of quantification (LOQ). Recent developments in analytical methods may result in lower LOQ. If a lower LOQ can be achieved with the same sample amount, then it generally stands that the original LOQ would require less sample material. For MILKYS this would potentially mean that more analyses could be carried out on the same cod liver sample.

Background

Many laboratories are faced with a demand from the food and manufacturing industry to analyze new and complex pollutants where access to sufficient quantities of material to be investigated is not an issue. Access to large amounts of material for analysis is often not the case where

monitoring of the environment is concerned. Hence, commercial laboratories have a tendency to develop as few and standardized analytical methods as possible instead of analyzing several compound groups together. This often requires more sample quantity than if the laboratory is set up to address several provisions from the same extract.

Several commercial laboratories demand more sample materiale in case of reanalyses. Hence, excess sample material is both a practical and an important economic aspect for laboratories. NIVA uses *inter alia* the commercial laboratory Eurofins. NIVA has arranged with Eurofins to reduce the requirement for sample quantity of 85 to 45 grams of cod liver, but this reduces the chance of having sufficient material for reanalysis.

Another factor is that MILKYS has to use several laboratories to address all the analyses demanded. Three different laboratories perform the determination of different organic compounds, and all three have as a standard the need for more material in case reanalyses are necessary.

When overseeing the development of the analytical methods with regards to lower sample material, much effort has been devoted to ensure that the quality of the many long time series MILKYS maintains is not impared.

Possible way forward
Some research laboratories who currently use newer methods can manage with significantly lower amount of sample. For some very standardized methods, such as fat determination and dry content, it is difficult to reduce the sample amount requirement. However, for other analyses, using state of the art technology it is possible to reduce the required sample amount by as much as 50 % without reducing the LOQ. Normally this would result in significant increase in the analytical cost.

Reducing LOQ is generally an on-going process for a laboratory taking into account market demand, available amount of sample, technical development and the most important the cost of development which usually requires more manual labor or new equipment. It should be noted that a change in sample quantity based on a change in method would likely alter status in regards to accreditation, inferring extra costs if this were to be remedied.

Considering MILKYS analyses
There are several groups of contaminants investigated by MILKYS where values below the LOQ are dominant. The methods applied should be evaluated in order to lower the LOQ.

For the higher PBDEs compounds with six bromines or more (e.g. BDE -153, -183, 196, -209) the technical requirements for reducing LOQs are available. The main problem for these compounds is their wide spread use and most laboratories have blank values that reveal several of these compounds. This means that a very important first step is that the sample preperation needs to be done in a clean laboratory to reduce PBDEs in the blank values. Unfortunately, most laboratories do not have access to such facilities. It should be noted that BDE 153 is the only one of the higher PBDEs that are included in the EQS values for biota (EU directive 2013/39/EU). Other lower PBDEs like BDE 47 are normally the most dominant congener and all biota samples have levels of BDE 47 above LOQ.

The ß and γ isomers of HBCD are often reported to be below the LOQ. This is expected becauses ß- and γ-HBCD, in contrasts to the α- isomers, are not shown to bioaccumulate due to *inter alia* biotransformation and chemical properties (Haukås *et al*. 2010). Fortunately the LOQs for alle three diasteromers are also significantly lower than the EQS for HBCD.

The chlorinated paraffins MCCP and SCCP are known to analytical chemists to be some of more difficult compounds to analyze in biota. The most important issue is that this group consist of many varieties of MCCP and SCCP. Intercalibration exercises show that the coefficient of variation between laboratories are 100-200% when biota and oil are analyzed (van der Veen *et al*. 2012, Pellizzato *et al*. 2009). This indicates that the main focus should be on reducing the analytical uncertainty before working with reducing of the LOQ.

The PFAS compound PFBS has on the whole not been detected in MILKYS investigations of cod (e.g. ***Table 13***). This is perhaps not unexpected since PFBS are known to bioaccumulate poorly. PFAS with longer chains such as PFDCA, PFDCS, PFNA, PFUdA are detected more frequently in freshwater fish in Norway, but the reason for this is not fully understood. A significant reduction of the LOQ would require that PFAS compounds are analyzed in at least two different methods, but this would increase the analytical costs significantly. It can be noted that the LOQ for PFOS is significantly lower than the EQS for this compound (it is the only PFAS compound with an EQS, cf. Eu directive 39/2013/EU).

PFRs are mostly reported as below LOQ despite the fact that the LOQ was reduced by about a factor of 10 from analyses of 2013 samples to analyses of 2014 samples. Though this is a positive development some investigations indicate that LOQ should be even a factor of 10 lower in order to be detected (Ruus *et al*. 2009).

In regards to BPA and TBBPA the LOQ reported for MILKYS are similar to many other studies (Ruus *et al*. 2014).

The levels and the LOQ of nonyl and octyl phenols varies considerably among samples witin and amoung differenct studies. There are however some investigations which have a lower LOQ than for MILKYS by a factor of two to five.

Conclusions

In general it is possible to lower the LOQ for all compounds presented in this study, however, this would likely mean a significant increase in analytical costs. A cost-benefit analysis should be considered before embarking on this route. EU legislation within environmental monitoring such as WFD could be a driving force for laboratories to lower their LOQs.

3.9.7 Use of passive samplers

The discussion below considers the replacement of cod liver-based biomonitoring by the use of absorption-based passive sampling for the monitoring of hydrophobic substances in water. Cod (and specifically cod liver) and mussels are currently used for contaminant monitoring under MILKYS and these species are also relevant for monitoring activities under the European Water Framework Directive (WFD) and the Marine Strategy Framework Directive (MSFD). Cod liver is generally used for the measurement of hydrophobic non-ionised chemicals that are not thought to be substantially metabolized in the fish (e.g. PBDEs, HBCDD, TBT, PCDD/Fs and DL-PCBs). Contaminants that can be metabolized in fish, like PAHs, are preferentially monitored in mollusks and crustaceans. ***Table 31*** lists hydrophobic non-ionised EU'spriority substances (cf. 2013/39/EU, the so-called EQSD), which are relevant to WFD and MSFD, that are generally amenable to passive sampling by absorption-based passive sampling devices (low density polyethylene and/or silicone rubber). Priority substances for which WFD environmental quality standards for biota (EQS_{biota}) have been established are also identified in the table.

The reasons for the use of cod (and biota in general) for monitoring purposes need to be reviewed before the usefulness of data from passive sampling can be considered to provide the same level of information as that obtained with cod sampling (cod liver). Hydrophobic substances are most often monitored using biota since these compounds are generally challenging to measure in water owing to trace level concentrations. Since selected organisms concentrate these chemicals, this facilitates the detection and quantification of these bioaccumulative substances. In addition, the concentration of contaminants found in biota provides information on contaminant bioavailability in the environment organisms live in. (see section 3.8.1).

It is clear that if the goal of biomonitoring is predator and secondary poisoning and human exposure safety, there is no role for passive sampling since contaminant measurements in the whole fish and in edible tissues, respectively should be prioritized. Unless robust relationships between whole fish and liver concentrations of contaminants exist or are developed, the use of cod liver as monitoring matrix may also not be suitable when the protection goal is human exposure (European Commission, 2010).

When the goal of monitoring is to infer chemical quality status of a water body or to assess trends in contaminant level in the environment, we believe passive sampling has a role to play to fulfil these objectives since the use of a biological matrix is not necessarily a priority.

Passive sampling provides time-integrated data for period of days to months or years depending on the chemical of interest and the deployment time. The use of performance reference compounds (PRCs) with absorption-based passive samplers reveal *in situ* contaminant exchange kinetics between water and the sampler (Booij *et al*. 1998). It is then possible to calculate time-integrated freely dissolved concentrations from masses of contaminants accumulated in the sampler. These concentrations are the driver of bioconcentration of hydrophobic contaminants into organisms (at lower trophic levels). When both lipid-polymer and polymer-water partition coefficients are known, passive sampling data can be converted to lipid-based concentrations for an organism considered at equilibrium with the environment the sampler was exposed to. This is a unit more closely related to biota concentrations, particularly when these are normalised to the lipid content of the matrix being monitored.

For the monitoring of trends in contaminant concentrations in the aquatic environment, the use of cod liver is not essential. Cod liver was selected simply because of the high bioconcentration factors obtained with the use of this matrix (compared with sampling water). Concentration factors for passive samplers (polymer-water partition coefficients, $K_{p\text{-}w}$) are of a similar order of magnitude and this essentially means that limits of detection will be of the same level for the two

methods. Passive sampling has an inherent low variability and this is an attractive characteristic when there is a need to establish trends (Smedes *et al*. 2007). This is due to the ability to use the same polymer with well-defined properties and standardize the uptake kinetics with PRCs.

***Table 31**. List of EU priority substances with EQS_{biota}, types of organisms these EQS values are applicable to, the protection goal and an assessment of passive sampling capacity for these chemicals according to EQS directive 2013/39/EU and supporting dossiers.*

Priority substance	EQS_{biota} (ng g^{-1})	Relevant organisms	Protection goal	Passive sampling
Anthracene				P[1], $K_{p\text{-}w}$[2]
Brominated diphenyl ethers (PBDEs)	0.0085	Fish	Human health via consumption of fishery products	P
C10-13 Chloroalkanes				P
Chlorpyrifos				P
Cyclodienes (Aldrin, Dieldrin, Endrin, Isodrin)				
DDT total (p,p'-DDT)				P, $K_{p\text{-}w}$
Di(2-ethylhexyl)phthalate (DEHP)				
Endosulfan				
Fluoranthene	30	Crustaceans and mollusks	Human health via consumption of fishery products	P, $K_{p\text{-}w}$
Hexachlorobenzene	10		Human health via consumption of fishery products	P, $K_{p\text{-}w}$
Hexachlorobutadiene	55		Secondary poisoning	P
Hexachlorocyclohexane				P, $K_{p\text{-}w}$
Naphthalene				P, $K_{p\text{-}w}$
nonylphenols (4-nonylphenol)				P, $K_{p\text{-}w}$
Octylphenols				P, $K_{p\text{-}w}$
Pentachlorobenzene				P, $K_{p\text{-}w}$
Benzo[a]pyrene	5	Crustaceans and mollusks	Human health via consumption of fishery products	P, $K_{p\text{-}w}$
Benzo[b]fluoranthene		Crustaceans and mollusks	Human health via consumption of fishery products	P, $K_{p\text{-}w}$
Benzo[k]fluoranthene		Crustaceans and mollusks	Human health via consumption of fishery products	P, $K_{p\text{-}w}$
Benzo[g,h,i]perylene		Crustaceans and mollusks	Human health via consumption of fishery products	P, $K_{p\text{-}w}$
Indeno[1,2,3-cd)pyrene		Crustaceans and mollusks	Human health via consumption of fishery products	P, $K_{p\text{-}w}$
Tributyltin and its compounds				P
Dicofol	33		Secondary poisoning	
Dioxins and dioxin-like compounds (PCDD/Fs and DL-PCBs)	0.0065 ng g^{-1} TEQ		Human health via consumption of fishery products	P
Cypermethrin				
Hexabromocyclododecane (HBCDD)	167		Secondary poisoning	P

[1]P: Passive sampling for these substances using absorption-based samplers has been performed already and freely dissolved contaminant concentrations in water can be estimated

[2]$K_{p\text{-}w}$: Contaminant polymer-water partition coefficients for low density polyethylene or silicone rubber are needed and available in the literature

Since passive sampling is based on the sampling of the freely dissolved concentration, which is thedriver of contaminant bioconcentration in organisms of low trophic levels, it provides relevant information for contaminant bioavailability in the aquatic environment and therefore also chemical quality status for the water body being sampled.

Table 32 is a summary of some of the relevant characteristics related to passive sampling and monitoring with cod when these techniques are aimed at trend monitoring and chemical quality status assessment. Absorption-based passive sampling can be applied to any hydrophobic and non-ionised substances with a $\log K_{ow} > 3$ while monitoring with cod (or any other biota) is restricted to contaminants that are not metabolized to any significant extent in the organism. For passive sampling, polymer-water partition coefficients for contaminants with $\log K_{ow} < 6$ are needed. For the more hydrophobic chemicals, K_{p-w} values are not required so long as uptake of these chemicals remains linear (i.e. far from equilibrium) during the sampling period. For species that can be metabolized by the organisms, passive sampling can help evaluate exposure of biota to these substances which cannot be undertaken reliably with body burden measurements.

Table 32*. Relevant characteristics of biota (cod) monitoring and passive sampling when applied to trend monitoring and chemical quality status assessment of water.*

	Cod (cod liver) monitoring	**Passive sampling**
Method variability	Method with a relatively high variability (requires the analysis of a high number of samples from each monitoring site)	Method with inherent low variability (allowing reduced number of analyses)
Relevant chemicals	Monitoring restricted to contaminants that are not metabolized by the fish	Monitoring of most hydrophobic non-ionised substances (K_{p-w} values need to be known for substances with $\log K_{ow} < 6$)
QA/QC	Selection of single species of fish, of age and size of the fish, trophic level Challenge with regards biota sample integrity	Robust QA/QC with blanks/controls, in situ standardization of contaminant uptake kinetics Challenge with sampler fouling in the field
Sampling location	Data representative of a wider area as a result of migratory behaviour of the fish	Static deployment at a single location (time integration but no real space integration)
Sampling schedule	Need to take into account seasonality: fish reproduction, foraging or overwintering	Seasonality can be taken into account with a correction of K_{p-w} for differences in temperature

From sampler preparation to field deployment analysis, most step of passive sampling can be standardized and controlled. Remaining challenges include the optimization of passive sampler exposure length since (bio)fouling has the potential to reduce sampling rates over time resulting in sampling rates that are likely to decrease with increasing exposure time. A water temperature during deployment that is significantly different from the temperature at which K_{p-w} values were obtained in the laboratory may need to be taken into account by adjusting K_{p-w} values for differences in temperature. One has to bear in mind though that EQS are themselves not temperature-dependent.

The effect of differences in water salinity on K_{p-w} can also be corrected if needs be. A potential difference between the use of cod liver and passive sampling is related to what the sample is representative of. Fish move within their home range and measurements in cod liver will therefore be representative of a certain body of water while measurements with passive samplers exposed in a static way on a mooring will not necessarily represent the same body of water. For one sampling site, it is possible to deploy multiple samplers at different locations or depths to increase spatial coverage.

The life cycle and behaviour of the fish needs to be taken into account when establishing a sampling programme since reproduction, foraging and overwintering can affect contaminant concentrations. The issue of seasonality resulting in the sampling of biota at the same time of the year may be acceptable for the determination of trends but may not be suitable for chemical quality status assessment. The assessment of the performance of laboratories involved in passive sampling can be conducted in a similar way to more common monitoring matrices. In recent years, passive sampler intercomparisons have been undertaken through the French network of reference laboratories AQUAREF, the NORMAN network and QUASIMEME.

Implementation of passive samplers in regulatory monitoring

According to the WFD, it is possible to convert EQSbiota into equally protective EQSwater standards against which passive sampler data can be compared for compliance checking. The low limits of detection that can be achieved with passive samplers should enable their use for compliance against freel dissolved water concentration-based EQS values. Data from passive samplers can also be converted to lipid-based concentrations expected for an organism that would be at thermodynamic equilibrium with the water through the use of lipid-polymer partition coefficients. While lipid-silicone rubber partition coefficients have been measured (Jahnke *et al.* 2008) for PCBs, these are not available yet for other priority substances. Such measurements should be undertaken in the future to be able to convert passive sampling data into lipid-based contaminant concentrations in biota assumed to be at equilibrium with the water.

In case the objective is for monitoring with passive sampling devices to supersede the use of cod liver, the implementation of passive samplers can be done with a stepwise approach. The continuity of long-term datasets needs to be considered. There should therefore be some overlap in the use of the two monitoring techniques for a sufficient period of time. Paired datasets of passive sampler-derived contaminant concentrations and biota concentrations could be established at multiple sites and for chemicals for which both methods are sufficiently robust. As reported in a recent position paper (Miège *et al.* 2015), a major recommendation is that whenever possible those in charge of monitoring should deploy passive samplers at a number of biomonitoring sites (OPSAR and/or WFD) to enable the development of paired passive sampler-biomonitoring datasets. Ideally, biomonitoring should include multiple trophic levels. These will enable us to further our understanding of bioaccumulation factors (BAFs) and their variability (particularly with regards to those used to establish EQS_{biota} values for priority substances). Only then can passive sampling be used for compliance checking. It may be that passive sampling does not need to totally replace cod (liver) monitoring, but can help significantly lower the number of sampling locations at which biota analysis is conducted.

According to the recently published technical guidance for the implementation of EQS_{biota}, passive sampling could be used within a tiered approach. Here, passive sampler application in the first tier may be used to identify locations of potential EQS exceedance (Miège *et al.* 2015; European Commission 2010). In this approach, passive samplers are applied in the first tier and data are compared with trigger or threshold values. Exceedance of these values triggers the second tier that requires the implementation of biota monitoring. Trigger values may be based on freely dissolved concentrations estimated from EQSbiota values and BAFs or be based on EQSbiota expressed on a lipid-basis for comparison with passive sampling data also reported on a lipid concentration basis.

3.9.8 Conclusions

The need for more material for the complex chemical analyses demanded for cod has enforce the need to explore alternative approaches that would achieve suitably similar assessment of the environment. Some alternatives were examined based mainly on experience from MILKYS in the sections above, and the main conclusions follow but it should be emphasized that there is a need to investigate all altnerative more thoroughly.

- Simulated sampling strategies based on earlier investigations of the MILKYS programme in the Inner Oslofjord revealed that a moderate extension of sampling time or area has either a neutral of positive effects on analysis, and thereby reducing the risk that statistical analysis erroneously indicates a significant trend. Interactions between sites and trends were found on larger geographic scale in exposed coastal areas of Norway. Here the presence of local-scale variations may lead to misleading interpretation of possible common regional changes.
- Other tissues might be an adequate alternative for liver when investigating PCBs and chlorinated paraffins (fillet) or PFAS (blood) but probably not for PBDEs, , α-HBCD, PFRs or bisphenol A.
- There are six fish species that might be good candidates as an alternative to cod: Norway pout, Poor cod, Blue whiting, Whiting, Haddock and Pollock/Saithe. None of these can however replace cod totally and should only be used after close consideration if cod is not available.
- It is generally technically possible to lower the LOQ but usually at significantly higher analytical costs. Fewer common driving forces (e.g. like EU EQS legislation) affecting laboratories doing environmental monitoring might improve this situation.
- Passive sampling may be used to replace biomonitoring. The implementation of passive samplers will require a better understanding of the relationship between biota concentrations obtained through biomonitoring and freely dissolved concentrations obtained with passive samplers. This should include an overlap with existing methods and through this and other research develop a better understanding of contaminant bioaccumulation factors in play.

4. Conclusions

This programme examines long-term changes for legacy contaminants in biota along the coast of Norway in both polluted and in areas remote from point sources. In addition, the programme includes supplementary analyses of some emerging contaminants. As such, the programme provides a basis for assessing the state of the environment for the coastal waters with respect to contaminants and changes over time. The main conclusions were:

- Most temporal trends are downwards, predominantly for metals, including TBT and its effect, but also PCBs and PFOS.
- The decrease in TBT can be related to legislation banning the use of this substance.
- Significant increase in mercury was found in cod from the Inner Oslo fjord. The concentrations were however lower in 2014 than in 2013. The reasons for the upward trend are at least in part related to the length of fish caught. Upward trends were also found in cod from Farsund and Bømlo.
- Highest concentrations of PBDEs, predominantly BDE47, were found in the Inner Oslofjord and Trondheimsfjord for cod liver, and in Inner Oslofjord (Akershuskaia) and Bergen harbour (Nordnes) for blue mussel.
- Blue mussel from one station in the Sørfjord was extremely polluted with DDE, presumably related to the earlier use of DDT as pesticide in this orchard district.
- Cod liver from the Inner Oslofjord and the Outer Oslofjord had significantly higher levels of PFOS and PFOSA than the six other stations investigated.
- Significant downward short-term trends at seven of the eight stations were identified for PFOS in cod liver.
- The dominant hexabromcyclododecane in cod liver was α–HBCD. The concentration of α–HBCD in cod liver was highest in the Inner Oslofjord and in blue mussel it was highest in Bergen harbour, probably related to urban activities.
- Short chain chlorinated paraffins (SCCP) were higher in blue mussel from Bergen harbour compared to other mussel-stations, whereas medium chain chlorinated paraffins (MCCP) were higher in the Grenlandsfjord whereas MCCP in cod liver was highest in the Inner Trondheimsfjord and Bømlo. A significant downward trend was found for SCCP in cod liver from the Inner Sørfjord for the period 1990 to 2014.
- The median concentrations of organophosphorus flame retardants (PFRs) were low or for the most part below the detection limit.
- The median concentrations of bisphenol A were below the detection limit or low (cod from Bømlo).
- The median concentrations of TBBPA, phthalates, triclosan, Diuron, Irgarol were generally below the detection limit.
- Alkylphenol results have indicated that 4-tert-nonylphenol and 4-n-octylphenol were the most dominant. High concentrations were not consistant with proximity to urban areas.
- The ICES/OSPAR Background Assessment Criteria (BAC) for OH-pyrene in cod bile was exceeded at all four stations investigated.
- Inhibited ALA-D activity in cod liver from the Inner Oslofjord and Inner Sørfjord indicated exposure to lead.
- EROD activities and CYP1A protein levels in cod liver from the Inner Oslofjord indicated exposure to contaminants.
- The Inner Oslofjord, and to a lesser degree the harbour areas of Bergen, Kristiansand, Trondheim, seems all together to be an area where contaminants tend to appear in high concentrations. This is probably caused by a high population in watershed area, a multitude of urban activities, and former and present use of products containing contaminants. A reduced

water exchange in the Inner Oslofjord with the outer fjord will also contribute to higher contaminant levels in water and biota.

- High levels of PCBs and Hg in cod are reasons for concern, particularly in the Inner Oslofjord. There is some evidence that elevated concentrations may result from increased fish length due to poor recruitment of cod in recent years this area.
- Freely dissolved contaminant concentrations measured with passive sampling are mostly close to or below limits of detection in the low pg/L range.
- Results from stabile isotopes indicate that the stations show very similar patterns from 2012 to 2014 in terms of isotopic signatures, suggesting that this is a spatial trend more than a temporal trend.
- Pooling samples of the same size can more effectively detect trends than pools of unequal sizes
- Extending the catch time or area moderately can enhance the means to detect trends in cod, though on a larger scale of extension there can be local influences that make assessments of regional trends misleading.
- Cod fillet and blood could potentially be used as an alternative to liver for monitoring PCBs (fillet) and chlorinated paraffins (fillet) and PFAS (blood).
- The use of other fish species and passive sampling or lowering the limit of quantification (LOQ) may mitigate the challenge of insufficient cod liver material.

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Appendix A
Quality assurance programme

Information on Quality Assurance

The chemical laboratories (NIVA and subcontractor Eurofins) and the biological laboratory (NIVA) have participated in the QUASIMEME international intercalibration exercises and other SLPs relevant to chemical and imposex analyses. The QUASIMEME exercises included nearly all the contaminants as well as imposex analysed in this programme.

The quality assurance programme is corresponding to the analyses of the 2013 samples (cf. Green *et al*. 2014). The results for QUASIMEME round 2014-2, FAPAS 202014 05100F and FAPAS 1264 apply to the 2014 samples. The results are acceptable.

NIVA participated in the last round of QUASIMEME Laboratory Performance Studies "imposex and intersex in Marine Snails BE1" in June-August 2012. Shell height, penis-length-male, penis-length-female, average-shell-height and female-male-ratio were measured. NIVA got the score satisfactory for all parameters except number of females for one sample, which got the score questionable. The score for VDSI was satisfactory for both samples tested.

In addition to the QUASIMEME exercises, certified reference materials (CRM) and in-house reference materials are analysed routinely with the MILKYS samples. It should be noted that for biota, the type of tissue used in the CRMs does not always match the target tissue for analysis. Uncertain values identified by the analytical laboratory or the reporting institute are flagged in the database. The results are also "screened" during the import to the database at NIVA and ICES.

Accreditation
The laboratories used for the chemical testing are accredited according to ISO 17025:2005, except for the PFCs.

Summary of quality control results
Standard Reference Materials (SRM) as well as in-house reference materials were analysed regularly (***Table 33***). Fish protein (DORM-4) was used as SRM for the control of the determination of metals. The SRM for determination of BDEs and HBCDDs in blue mussel and liver was *Folkehelsa* reference material Halibut 2012. For determination of PCBs, DDTs and PAHs in blue mussel, as well as PCBs, DDTs and BDEs in liver, QUASIMEME biota samples with known true value was applied. For bisphenol-A, canned peach reference material was used. For TBBPA, spiked fish oil was used for quality assurance, and for chlorinated paraffins and octyl-/nonylphenols, spiked fish meal was used. For organophosphorous flame retardants, spiked internal reference material was used.

***Table 33**. Summary of the quality control of results for the 2014 biota samples analysed in 2014-2015. The Standard Reference Materials (SRM) DORM-4* (fish protein) for blue mussel, fish liver and fish fillet.* Folkehelsa RM Halibut 2012** *were used for blue mussel and fish liver. The in-house reference materials were QUASIMEME samples QOR110BT (mussel tissue), QBC032BT and QOR108BT (fish liver) and QPH065BT (shellfish tissue). In addition, spiked fish oil, spiked fish meal and spiked internal reference material were analysed. The SRMs and in-house reference materials and quality assurance standards were analysed in series with the MILKYS samples, and measured several times (N) over a number of weeks (W). The values are reported in the following units: metals (mg/kg), BDE (pg/g mussel in soft body, µg/kg in liver), PCBs (µg/kg), DDTs (µg/kg), HBCDDs (pg/g), PAH (µg/kg), TBBPA (ng/sample), BPA (µg/kg), SCCP/MCCP (ng/sample) octyl-/nonylphenol (ng/sample), organophosphorus flame retardants (pg/sample) and PFCs (% recovery). Tissue types were: mussel soft body (SB), fish liver (LI) and fish fillet (MU).*

Code	Contaminant	Tissue type	SRM type	SRM value confidence interval	N	W	Mean value	Standard deviation
Ag	Silver	SB/LI	DORM-4	m	41	21	0.023	0.0028
As	Arsenic	SB/LI	DORM-4	6.80 ± 0.64	41	21	6.34	0.42
Cd	Cadmium	SB/LI	DORM-4	0.306 ± 0.015	42	21	0.295	0.020
Cr	Chromium	SB/LI	DORM-4	1.87 ± 0.16	41	21	1.89	0.22
Co	Cobalt	SB/LI	DORM-4	m	41	21	0.233	0.015
Cu	Copper	SB/LI	DORM-4	15.9 ± 0.9	42	21	13.89	0.95
Hg	Mercury	SB/MU	DORM-4	0.410 ± 0.055	43	23	0.36	0.034
Ni	Nickel	SB/LI	DORM-4	1.36 ± 0.22	41	21	1.27	0.15
Pb	Lead	SB/LI	DORM-4	0.416 ± 0.053	42	21	0.38	0.022
Zn	Zinc	SB/LI	DORM-4	52.2 ± 3.2	41	21	50.6	3.4
BDE28	2,2,4' Tribromodiphenylether	LI	QBC032BT	0.39	35	6	0.391	0.086
BDE100	2,2',4,4',6-Pentabromodiphenylether	LI	QBC032BT	6.91	35	6	6.14	1.48
BDE153	2,2',4,4'5,5'-Hexabromodiphenylether	LI	QBC032BT	0.86[1)]	35	6	0.71	0.15
BDE154	2,2',4,4',5,6'-Hexabromodiphenylether	LI	QBC032BT	1.68	35	6	1.87	0.35
BDE47	2,2',4,4',-Tetrabromodiphenylether	LI	QBC032BT	23.2[1)]	35	6	19.08	4.21
BDE99	2,2',4,4',5-Pentabromodiphenylether	LI	QBC032BT	0.01[1)]	35	6	0.008	0.002
BDE126	3,3',4,4',5'-Pentabromodiphenyl ether	LI	QBC032BT	m	35	6	0.023	0.009
BDE183	2,2',3,4,4,5',6-Heptabromodiphenylether	LI	QBC032BT	m	35	6	0.0078	0.0033
BDE196	2,2',3,3',4,4',5',6-Octabromodiphenyl ether	LI		m	35	6	m	m
BDE209	Decabromodiphenylether	LI	QBC032BT	m	35	6	0.031	0.006
BDE28	2,2,4' Tribromodiphenylether	SB	Folkehelsa RM Halibut 2012	35 ± 5.6	2	1	39	0.6
BDE100	2,2',4,4',6-Pentabromodiphenylether	SB	Folkehelsa RM Halibut 2012	92 ± 12	2	1	95	0.03
BDE153	2,2',4,4'5,5'-Hexabromodiphenylether	SB	Folkehelsa RM Halibut 2012	17 ± 3.2	2	1	19	0.6
BDE154	2,2',4,4',5,6'-Hexabromodiphenylether	SB	Folkehelsa RM Halibut 2012	86 ± 19	2	1	76	10
BDE47	2,2',4,4',-Tetrabromodiphenylether	SB	Folkehelsa RM Halibut 2012	544 ± 94	2	1	570	2
BDE99	2,2',4,4',5-Pentabromodiphenylether	SB	Folkehelsa RM Halibut 2012	26 ± 6.5	2	1	26	0.5
BDE126	3,3',4,4',5'-Pentabromodiphenyl ether	SB	Folkehelsa RM Halibut 2012	m	m	m	m	m
BDE183	2,2',3,4,4,5',6-Heptabromodiphenylether	SB	Folkehelsa RM Halibut 2012	0.55 ± 0.31	2	1	<3.5	2.7
BDE196	2,2',3,3',4,4',5',6-Octabromodiphenyl ether	SB	Folkehelsa RM Halibut 2012	m	m	m	m	m
BDE209	Decabromodiphenylether	SB	Folkehelsa RM Halibut 2012	21 ± 10	2	1	<44	19
CB101	PCB congener CB-101	SB	QOR110BT	3.25	24	11	3.25	2.98
CB118	PCB congener CB-118	SB	QOR110BT	2.20	24	11	2.09	0.19
CB138	PCB congener CB-138	SB	QOR110BT	7.93	24	11	5.70	0.27
CB153	PCB congener CB-153	SB	QOR110BT	4.46	24	11	8.44	0.53
CB180	PCB congener CB-180	SB	QOR110BT	0.48	24	11	0.56	0.04
CB28	PCB congener CB-28	SB	QOR110BT	0.37	24	11	0.46	0.05
CB52	PCB congener CB-52	SB	QOR110BT	1.11	24	11	1.29	0.12
DDEPP	4.4'-DDE	SB	QOR110BT	1.4	24	11	1.79	0.34
TDEPP	4.4'-DDD	SB	QOR110BT	0.59	24	11	0.42	0.16
DDTPP	4.4'-DDT	SB	QOR110BT	0.14[1)]	24	11	0.31	0.39
α-HBCDD	α-Hexabromocyclododecane	LI	Folkehelsa RM Halibut 2012	388 ± 70	2	2	556	41.7
ß-HBCDD	ß- Hexabromocyclododecane	LI	Folkehelsa RM Halibut 2012	6 ± 7.6	2	2	59	30
γ-HBCDD	γ- Hexabromocyclododecane	LI	Folkehelsa RM Halibut 2012	13 ± 1.4	1	1	38	-

Code	Contaminant	Tissue type	SRM type	SRM value confidence interval	N	W	Mean value	Standard deviation
CB101	PCB congener CB-101	LI	QOR108BT	63.7	46	11	57.58	9.14
CB118	PCB congener CB-118	LI	QOR108BT	69.9	46	11	60.96	7.87
CB138	PCB congener CB-138	LI	QOR108BT	219	46	11	161.89	18.32
CB153	PCB congener CB-153	LI	QOR108BT	204.77	46	11	200.27	24.00
CB180	PCB congener CB-180	LI	QOR108BT	45.5	46	11	40.98	4.37
CB28	PCB congener CB-28	LI	QOR108BT	10.5	46	11	10.27	1.30
CB52	PCB congener CB-52	LI	QOR108BT	23.7	46	11	22.19	2.02
DDEPP	4.4'-DDE	LI	QOR108BT	83.1	46	11	73.54	12.36
TDEPP	4.4'-DDD	LI	QOR108BT	26.7	46	11	19.66	7.32
DDTPP	4.4'-DDT	LI	QOR108BT	0.83[1)]	46	11	m	m
ACNE	Acenaphthene	SB	QPH065BT	0.77	30	15	0.61	0.16
ACNLE	Acenaphthylene	SB	QPH065BT	0.45	30	15	0.68	0.3
ANT	Anthracene	SB	QPH065BT	0.75	30	15	1.81	0.47
BAP	benzo[a]pyrene	SB	QPH065BT	1.50	30	15	1.78	0.49
BBJF	Benzo[b+j]fluoranthene	SB	QPH065BT	4.99	30	15	4.65	1.11
BKF	Benzo[k]fluoranthene	SB	QPH065BT	2.00	30	15	3.17	0.61
BAA	Benzo[a]anthracene	SB	QPH065BT	5.26	30	15	5.07	0.72
CHR	Chrysene	SB	QPH065BT	7.19	30	15	6.81	0.81
DBA3A	Dibenzo[ac,ah]anthracene	SB	QPH065BT	0.43	30	15	0.39	0.11
FLE	Fluorene	SB	QPH065BT	1.59	30	15	0.77	0.20
FLU	Fluoranthene	SB	QPH065BT	13.8	30	15	16.02	2.76
ICDP	Indeno[1,2,3-cd]pyrene	SB	QPH065BT	1.52	30	15	1.13	0.43
NAP	Naphthalene	SB	QPH065BT	5.05	30	15	4.1	1.19
PA	Phenanthrene	SB	QPH065BT	8.18	30	15	7.72	0.89
BGHIP	Benzo(g,h,i)perylene	SB	QPH065BT	2.39	30	15	1.85	0.58
PYR	Pyrene	SB	QPH065BT	11.1	30	15	13.84	2.37
TBBPA	Tetrabromobisphenol-A	LI/SB	Internal RM (spiked fish oil)	m	20	17	1.48	0.1
BPA	Bisphenol-A	LI/SB	Peach, canned	30.26 ± 1.66	27	17	30.10	2.39
SCCP	C10-C13 Chlorinated paraffine	LI/SB	Internal RM (spiked fish meal)	m	8	17	1833	572
MCCP	C13-C17 Chlorinated paraffine	LI/SB	Internal RM (spiked fish meal)	m	7	17	4263	1368
	4-n-nonylphenol	LI/SB	Internal RM (spiked fish meal)	50	17	19	40.7	0.84
	4-n-octylphenol	LI/SB	Internal RM (spiked fish meal)	50	17	19	38.5	1.13
	4-Nonylphenol	LI/SB	Internal RM (spiked fish meal)	m	m	m	m	m
	4-tert-octylphenol	LI/SB	Internal RM (spiked fish meal)	50	17	19	41.9	6.37
TIBP	Triisobutylphosphate	LI/SB	Internal RM (spiked)	5565.2	10	5	4569.1	359.3
TBP	Tributylphosphate	LI/SB	Internal RM (spiked)	5217.4	10	5	5010	201.1
TCEP	Tris(2-chloroethyl)phosphate	LI/SB	Internal RM (spiked)	5217.4	10	5	4916.6	86.3
TCPP	Tris(2-chloro-isopropyl)phosphate	LI/SB	Internal RM (spiked)	5565.2	10	5	5610	320
TDCP	Tris(1,3-chloro-isopropyl)phosphate	LI/SB	Internal RM (spiked)	5217.4	10	5	5241.7	302.3
TBEP	Tris(2-butoxyethyl)phosphate	LI/SB	Internal RM (spiked)	5217.4	10	5	5101.4	249.4
TPhP	Triphenylphosphate	LI/SB	Internal RM (spiked)	5217.4	10	5	5321.3	467.6
EHDPP	2-Ethylhexyl-diphenylphosphate	LI/SB	Internal RM (spiked)	5217.4	10	5	5407.8	405.8
TEHP	Tris(2-ethylhexyl) phosphate	LI/SB	Internal RM (spiked)	5217.4	10	5	4472.3	561.9
ToCrP	o-Tricresylphosphate	LI/SB	Internal RM (spiked)	5217.4	10	5	5478.3	362.6
TCrP	Tricresylphosphate	LI/SB	Internal RM (spiked)	5165.3	10	5	5553	584.9
PFBS	Perfluorobutane sulphonate	LI		100 %[2)]	8	m	98	5.6
PFHxA	Perfluorohexane acid	LI		100 %[2)]	8	m	98	7.2
PFHpA	Perfluoroheptane acid	LI		100 %[2)]	8	m	98	5.0
PFOA	Perfluorooctane acid	LI		100 %[2)]	8	m	104	8.9
PFNA	Perfluorononane acid	LI		100 %[2)]	8	m	101	8.0
PFOS	Perfluorooctane sulphonate	LI		100 %[2)]	8	m	103	4.5
PFOSA	Perfluorooctane sulphone amide	LI		100 %[2)]	8	m	101	6.4
PFHxS	Perfluorohexane sulphonate	LI		100 %[2)]	8	m	98	6.1
PFDA	Perfluorodecanoic acid	LI		100 %[2)]	8	m	95	8.2
PFUDA	Perfluoroundecanoic acid	LI		100 %[2)]	8	m	94	6.7
PFDS	Perfluorodecanesulphonate	LI		100 %[2)]	8	m	87	7.4

* National Research Council Canada, Division of Chemistry, Marine Analytical Chemistry Standards.

** BCR, Community Bureau of Reference, Commission of the European Communities.

1) Not certified value.

2) Recovery of spiked control sample

Appendix B
Abbreviations

Abbreviation[1]	English	*Norwegian*	Param. group
ELEMENTS			
Al	aluminium	*aluminium*	I-MET
Ag	Silver	*sølv*	I-MET
As	arsenic	*arsen*	I-MET
Ba	barium	*barium*	I-MET
Cd	cadmium	*kadmium*	I-MET
Ce	cerium	*serium*	I-MET
Co	cobalt	*kobolt*	I-MET
Cr	chromium	*krom*	I-MET
Cu	copper	*kobber*	I-MET
Fe	iron	*jern*	I-MET
Hg	mercury	*kvikksølv*	I-MET
La	lanthanum	*lantan*	I-MET
Li	lithium	*litium*	I-MET
Mn	manganese	*mangan*	I-MET
Mo	molybdenum	*molybden*	I-MET
Nd	neodymium	*neodym*	I-MET
Ni	nickel	*nikkel*	I-MET
Pb	lead	*bly*	I-MET
Pb210	lead-210	*bly-210*	I-RNC
Pr	praseodymium	*praseodym*	I-MET
Se	selenium	*selen*	I-MET
Sn	tin	*tinn*	I-MET
Ti	titanium	*titan*	I-MET
V	vanadium	*vanadium*	I-MET
Zn	zinc	*sink*	I-MET
METAL COMPOUNDS			
TBT	tributyltin (formulation basis =TBTIN*2.44)	*tributyltinn (formula basis =TBTIN*2.44)*	O-MET
MBTIN (MBT)	monobutyltin	*monobutyltinn*	O-MET
MBTIN (MBT)	monobutyltin	*monobutyltinn*	O-MET
MOT	monooctyltin	*monooktyltinn*	O-MET
MPTIN	monophenyltin	*monofenyltinn*	O-MET
DBTIN	dibutyltin (di-n-butyltin)	*dibutyltinn (di-n-butyltinn)*	O-MET
DOT	dioctyltin	*dioktyltinn*	O-MET
DPTIN	diphenyltin	*difenyltinn*	O-MET
TBTIN	tributyltin (=TBT*0.40984)	*tributyltinn (=TBT*0.40984)*	O-MET
TCHT	tricyclohexyl-stannylium	*tricyclohexyl-stannylium*	O-MET
TPTIN (TPhT)	triphenyltin	*trifenyltinn*	O-MET
TTBT	tetrabutyltin	*tetrabutyltinn*	O-MET
PAHs			
PAH	polycyclic aromatic hydrocarbons	*polysykliske aromatiske hydrokarboner*	
ACNE [3]	acenaphthene	*acenaften*	PAH
ACNLE [3]	acenaphthylene	*acenaftylen*	PAH

Abbreviation[1]	English	*Norwegian*	Param. group
ANT [3]	anthracene	*antracen*	PAH
BAA [3, 4]	benzo[*a*]anthracene	*benzo[a]antracen*	PAH
BAP [3, 4]	benzo[*a*]pyrene	*benzo[a]pyren*	PAH
BBF [3, 4]	benzo[*b*]fluoranthene	*benzo[b]fluoranten*	PAH
BBJF [3, 4]	benzo[*j*]fluoranthene	*benzo[j]fluoranten*	PAH
BBJKF [3, 4]	benzo[*b,j,k*]fluoranthene	*benzo[b,j,k]fluoranten*	PAH
BBJKF [3, 4]	benzo[b+j,k]fluoranthene	*benzo[b+j,k]fluoranten*	PAH
BBKF [3, 4]	benzo[*b+k*]fluoranthene	*benzo[b+k]fluoranten*	PAH
BEP	benzo[*e*]pyrene	*benzo[e]pyren*	PAH
BGHIP [3]	benzo[*ghi*]perylene	*benzo[ghi]perylen*	PAH
BIPN [2]	biphenyl	*bifenyl*	PAH
BJKF [3, 4]	benzo[*j,k*]fluoranthene	*benzo[j,k]fluorantren*	PAH
BKF [3, 4]	benzo[*k*]fluoranthene	*benzo[k]fluorantren*	PAH
CHR [3, 4]	chrysene	*chrysen*	PAH
CHRTR [3, 4]	chrysene+triphenylene	*chrysen+trifenylen*	PAH
COR	coronene	*coronen*	PAH
DBAHA [3, 4]	dibenz[*a,h*]anthracene	*dibenz[a,h]anthracen*	PAH
DBA3A [3, 4]	dibenz[*a,c/a,h*]anthracene	*dibenz[a,c/a,h]antracen*	PAH
DBP [4, 6]	dibenzopyrenes	*dibenzopyren*	PAH
DBT	dibenzothiophene	*dibenzothiofen*	PAH
DBTC1	C_1-dibenzothiophenes	*C_1-dibenzotiofen*	PAH
DBTC2	C_2-dibenzothiophenes	*C_2-dibenzotiofen*	PAH
DBTC3	C_3-dibenzothiophenes	*C_3-dibenzotiofen*	PAH
FLE [3]	fluorene	*fluoren*	PAH
FLU [3]	fluoranthene	*fluoranten*	PAH
ICDP [3, 4]	indeno[*1,2,3-cd*]pyrene	*indeno[1,2,3-cd]pyren*	PAH
NAP [2]	naphthalene	*naftalen*	PAH
NAPC1 [2]	C_1-naphthalenes	*C_1-naftalen*	PAH
NAPC2 [2]	C_2-naphthalenes	*C_2-naftalen*	PAH
NAPC3 [2]	C_3-naphthalenes	*C_3-naftalen*	PAH
NAP1M [2]	1-methylnaphthalene	*1-metylnaftalen*	PAH
NAP2M [2]	2-methylnaphthalene	*2-metylnaftalen*	PAH
NAPD2 [2]	1,6-dimethylnaphthalene	*1,6-dimetylnaftalen*	PAH
NAPD3 [2]	1,5-dimethylnaphthalene	*1,5-dimetylnaftalen*	PAH
NAPDI [2]	2,6-dimethylnaphthalene	*2,6-dimetylnaftalen*	PAH
NAPT2 [2]	2,3,6-trimethylnaphthalene	*2,3,6-trimetylnaftalen*	PAH
NAPT3 [2]	1,2,4-trimethylnaphthalene	*1,2,4-trimetylnaftalen*	PAH
NAPT4 [2]	1,2,3-trimethylnaphthalene	*1,2,3-trimetylnaftalen*	PAH
NAPTM [2]	2,3,5-trimethylnaphthalene	*2,3,5-trimetylnaftalen*	PAH
NPD	collective term for naphthalenes, phenanthrenes and dibenzothiophenes	*Samme betegnelse for naftalen, fenantren og dibenzotiofens*	PAH
PA [3]	phenanthrene	*fenantren*	PAH
PAC1	C_1-phenanthrenes	*C_1-fenantren*	PAH
PAC2	C_2-phenanthrenes	*C_2-fenantren*	PAH
PAC3	C_3-phenanthrenes	*C_3-fenantren*	PAH
PAM1	1-methylphenanthrene	*1-metylfenantren*	PAH
PAM2	2-methylphenanthrene	*2-metylfenantren*	PAH

Abbreviation[1]	English	*Norwegian*	Param. group
PADM1	3,6-dimethylphenanthrene	*3,6-dimetylfenantren*	PAH
PADM2	9,10-dimethylphenanthrene	*9,10-dimetylfenantren*	PAH
PER	perylene	*perylen*	PAH
PYR [3]	pyrene	*pyren*	PAH
DI-Σn	sum of "n" dicyclic "PAH"s (footnote 2)	*sum "n" disykliske "PAH" (fotnote 2)*	
P-Σn/P_S	sum "n" PAH (DI-Σn not included, footnote 3)	*sum "n" PAH (DI-Σn ikke inkludert, fotnote 3)*	
PK-Σn/PK_S	sum carcinogen PAHs (footnote 4)	*sum kreftfremkallende PAH (fotnote 4)*	
PAHΣΣ	dI-Σn + P-Σn etc.	*dI-Σn + P-Σn mm.*	
SPAH	"total" PAH, specific compounds not quantified (outdated analytical method)	*"total" PAH, spesifikk forbindelser ikke kvantifisert (foreldet metode)*	
BAP_P	% BAP of PAHΣΣ	*% BAP av PAHΣΣ*	
BAPPP	% BAP of P-Σn	*% BAP av P-Σn*	
BPK_P	% BAP of PK_Sn	*% BAP av PK_Sn*	
PKn_P	% PK_Sn of PAHΣΣ	*% PK_Sn av PAHΣΣ*	
PKnPP	% PK_Sn of P-Σn	*% PK_Sn av P-Σn*	
PCBs			
PCB	**polychlorinated biphenyls**	***polyklorerte bifenyler***	
CB	individual chlorobiphenyls (CB)	*enkelte klorobifenyl*	
CB28	CB28 (IUPAC)	*CB28 (IUPAC)*	OC-CB
CB31	CB31 (IUPAC)	*CB31 (IUPAC)*	OC-CB
CB44	CB44 (IUPAC)	*CB44 (IUPAC)*	OC-CB
CB52	CB52 (IUPAC)	*CB52 (IUPAC)*	OC-CB
CB77 [5]	CB77 (IUPAC)	*CB77 (IUPAC)*	OC-CB
CB81 [5]	CB81 (IUPAC)	*CB81 (IUPAC)*	OC-CB
CB95	CB95 (IUPAC)	*CB95 (IUPAC)*	OC-CB
CB101	CB101 (IUPAC)	*CB101 (IUPAC)*	OC-CB
CB105	CB105 (IUPAC)	*CB105 (IUPAC)*	OC-CB
CB110	CB110 (IUPAC)	*CB110 (IUPAC)*	OC-CB
CB118	CB118 (IUPAC)	*CB118 (IUPAC)*	OC-CB
CB126 [5]	CB126 (IUPAC)	*CB126 (IUPAC)*	OC-CB
CB128	CB128 (IUPAC)	*CB128 (IUPAC)*	OC-CB
CB138	CB138 (IUPAC)	*CB138 (IUPAC)*	OC-CB
CB149	CB149 (IUPAC)	*CB149 (IUPAC)*	OC-CB
CB153	CB153 (IUPAC)	*CB153 (IUPAC)*	OC-CB
CB156	CB156 (IUPAC)	*CB156 (IUPAC)*	OC-CB
CB169 [5]	CB169 (IUPAC)	*CB169 (IUPAC)*	OC-CB
CB170	CB170 (IUPAC)	*CB170 (IUPAC)*	OC-CB
CB180	CB180 (IUPAC)	*CB180 (IUPAC)*	OC-CB
CB194	CB194 (IUPAC)	*CB194 (IUPAC)*	OC-CB
CB209	CB209 (IUPAC)	*CB209 (IUPAC)*	OC-CB
CB-Σ7	CB: 28+52+101+118+138+153+180	*CB: 28+52+101+118+138+153+180*	
CB-ΣΣ	sum of CBs, includes CB-Σ7	*sum CBer, inkluderer CB-Σ7*	

Abbreviation[1]	English	*Norwegian*	Param. group
TECBW	sum of CB-toxicity equivalents after WHO model, see **TEQ**	*sum CB- toksisitets ekvivalenter etter WHO modell, se **TEQ***	
TECBS	sum of CB-toxicity equivalents after SAFE model, see **TEQ**	*sum CB-toksisitets ekvivalenter etter SAFE modell, se **TEQ***	
PCN	polychlorinated naphthalenes	*polyklorerte naftalen*	
DIOXINs			
TCDD	2, 3, 7, 8-tetrachloro-dibenzo dioxin	*2, 3, 7, 8-tetrakloro-dibenzo dioksin*	OC-DX
CDDST	sum of tetrachloro-dibenzo dioxins	*sum tetrakloro-dibenzo dioksiner*	
CDD1N	1, 2, 3, 7, 8-pentachloro-dibenzo dioxin	*1, 2, 3, 7, 8-pentakloro-dibenzo dioksin*	OC-DX
CDDSN	sum of pentachloro-dibenzo dioxins	*sum pentakloro-dibenzo dioksiner*	
CDD4X	1, 2, 3, 4, 7, 8-hexachloro-dibenzo dioxin	*1, 2, 3, 4, 7, 8-heksakloro-dibenzo dioksin*	OC-DX
CDD6X	1, 2, 3, 6, 7, 8-hexachloro-dibenzo dioxin	*1, 2, 3, 6, 7, 8-heksakloro-dibenzo dioksin*	OC-DX
CDD9X	1, 2, 3, 7, 8, 9-hexachloro-dibenzo dioxin	*1, 2, 3, 7, 8, 9-heksakloro-dibenzo dioksin*	OC-DX
CDDSX	sum of hexachloro-dibenzo dioxins	*sum heksakloro-dibenzo dioksiner*	
CDD6P	1, 2, 3, 4, 6, 7, 8-heptachloro-dibenzo dioxin	*1, 2, 3, 4, 6, 7, 8-heptakloro-dibenzo dioksin*	OC-DX
CDDSP	sum of heptachloro-dibenzo dioxins	*sum heptakloro-dibenzo dioksiner*	
CDDO	Octachloro-dibenzo dioxin	*Oktakloro-dibenzo dioksin*	OC-DX
PCDD	sum of polychlorinated dibenzo-p-dioxins	*sum polyklorinaterte-dibenzo-p-dioksiner*	
CDF2T	2, 3, 7, 8-tetrachloro-dibenzofuran	*2, 3, 7, 8-tetrakloro-dibenzofuran*	OC-DX
CDFST	sum of tetrachloro-dibenzofurans	*sum tetrakloro-dibenzofuraner*	
CDFDN	1, 2, 3, 7, 8/1, 2, 3, 4, 8-pentachloro-dibenzofuran	*1, 2, 3, 7, 8/1, 2, 3, 4, 8-pentakloro-dibenzofuran*	OC-DX
CDF2N	2, 3, 4, 7, 8-pentachloro-dibenzofuran	*2, 3, 4, 7, 8-pentakloro-dibenzofuran*	OC-DX
CDFSN	sum of pentachloro-dibenzofurans	*sum pentakloro-dibenzofuraner*	
CDFDX	1, 2, 3, 4, 7, 8/1, 2, 3, 4, 7, 9-hexachloro-dibenzofuran	*1, 2, 3, 4, 7, 8/1, 2, 3, 4, 7, 9-heksakloro-dibenzofuran*	OC-DX
CDF6X	1, 2, 3, 6, 7, 8-hexachloro-dibenzofuran	*1, 2, 3, 6, 7, 8-heksakloro-dibenzofuran*	OC-DX
CDF9X	1, 2, 3, 7, 8, 9-hexachloro-dibenzofuran	*1, 2, 3, 7, 8, 9-heksakloro-dibenzofuran*	OC-DX
CDF4X	2, 3, 4, 6, 7, 8-hexachloro-dibenzofuran	*2, 3, 4, 6, 7, 8-heksakloro-dibenzofuran*	OC-DX

Abbreviation[1]	English	*Norwegian*	Param. group
CDFSX	sum of hexachloro-dibenzofurans	*sum heksakloro-dibenzofuraner*	
CDF6P	1, 2, 3, 4, 6, 7, 8-heptachloro-dibenzofuran	*1, 2, 3, 4, 6, 7, 8-heptakloro-dibenzofuran*	OC-DX
CDF9P	1, 2, 3, 4, 7, 8, 9-heptachloro-dibenzofuran	*1, 2, 3, 4, 7, 8, 9-heptakloro-dibenzofuran*	OC-DX
CDFSP	sum of heptachloro-dibenzofurans	*sum heptakloro-dibenzofuraner*	OC-DX
CDFO	octachloro-dibenzofurans	*octakloro-dibenzofuran*	OC-DX
PCDF	sum of polychlorinated dibenzo-furans	*sum polyklorinated dibenzo-furaner*	
CDDFS	sum of PCDD and PCDF	*sum PCDD og PCDF*	
TCDDN	sum of TCDD-toxicity equivalents after Nordic model, see **TEQ**	*sum TCDD- toksisitets ekvivalenter etter Nordisk modell, se **TEQ***	
TCDDI	sum of TCDD-toxicity equivalents after international model, see **TEQ**	*sum TCDD-toksisitets ekvivalenter etter internasjonale modell, se **TEQ***	
BIOICIDES			
ALD	aldrin	*aldrin*	OC-DN
DIELD	dieldrin	*dieldrin*	OC-DN
ENDA	endrin	*endrin*	OC-DN
CCDAN	cis-chlordane (=α-chlordane)	*cis-klordan (=α-klordan)*	OC-DN
TCDAN	trans-chlordane (=γ-chlordane)	*trans-klordan (=γ-klordan)*	OC-DN
OCDAN	oxy-chlordane	*oksy-klordan*	OC-DN
TNONC	trans-nonachlor	*trans-nonaklor*	OC-DN
TCDAN	trans-chlordane	*trans-klordan*	OC-DN
Triclosan	5-chloro-2-2,4-dichlorophenoxy)phenol	*5-kloro-2-2,4-diklorofenoxy)fenol*	OC-CL
Diuron	3-(3,4-dichlorophenyl)-1,1-dimethylurea	*3-(3,4-diklorofenyl)-1,1-dimetylurea*	OC-CL
Irgarol	a triazine (nitrogen containing heterocycle)	*en triazin (nitrogen holdig heterosykle)*	
OCS	octachlorostyrene	*oktaklorstyren*	OC-CL
QCB	pentachlorobenzene	*pentaklorbenzen*	OC-CL
DDD	dichlorodiphenyldichloroethane 1,1-dichloro-2,2-bis-(4-chlorophenyl)ethane	*diklordifenyldikloretan* *1,1-dikloro-2,2-bis-(4-klorofenyl)etan*	OC-DD
DDE	dichlorodiphenyldichloroethylene (principle metabolite of DDT) 1,1-*bis*-(4-chlorophenyl)-2,2-dichloroethene*	*diklordifenyldikloretylen* *(hovedmetabolitt av DDT)* *1,1-bis-(4-klorofenyl)-2,2-dikloroeten*	OC-DD
DDT	dichlorodiphenyltrichloroethane 1,1,1-trichloro-2,2-bis-(4-chlorophenyl)ethane	*diklordifenyltrikloretan* *1,1,1-trikloro-2,2-bis-(4-klorofenyl)etan*	OC-DD

Abbreviation[1]	English	*Norwegian*	Param. group
DDEOP	o,p'-DDE	*o,p'-DDE*	OC-DD
DDEPP	p,p'-DDE	*p,p'-DDE*	OC-DD
DDTOP	o,p'-DDT	*o,p'-DDT*	OC-DD
DDTPP	p,p'-DDT	*p,p'-DDT*	OC-DD
TDEPP	p,p'-DDD	*p,p'-DDD*	OC-DD
DDTEP	p,p'-DDE + p,p'-DDT	*p,p'-DDE + p,p'-DDT*	OC-DD
DD-nΣ	sum of DDT and metabolites, n = number of compounds	*sum DDT og metabolitter, n = antall forbindelser*	OC-DD
HCB	hexachlorobenzene	*heksaklorbenzen*	OC-CL
HCHG	Lindane γ HCH = gamma hexachlorocyclohexane (γ BHC = gamma benzenehexachloride, outdated synonym)	*Lindan* *γ HCH = gamma heksaklorsykloheksan* *(γ BHC = gamma benzenheksaklorid, foreldet betegnelse)*	OC-HC
HCHA	α HCH = alpha HCH	*α HCH = alpha HCH*	OC-HC
HCHB	β HCH = beta HCH	*β HCH = beta HCH*	OC-HC
HC-nΣ	sum of HCHs, n = count	*sum av HCHs, n = antall*	
EOCl	extractable organically bound chlorine	*ekstraherbart organisk bundet klor*	OC-CL
EPOCl	extractable persistent organically bound chlorine	*ekstraherbart persistent organisk bundet klor*	OC-CL
PBDEs			
PBDE	**polybrominated diphenyl ethers**	***polybromerte difenyletere***	OC-BR
BDE	brominated diphenyl ethers		OC-BR
BDE28	2,4,4'-tribromodiphenyl ether	*2,4,4'-tribromdifenyleter*	OC-BR
BDE47	2,2',4,4'-tetrabromodiphenyl ether	*2,2',4,4'-tetrabromdifenyleter*	OC-BR
BDE49*	2,2',4,5'- tetrabromodiphenyl ether	*2,2',4,5'- tetrabromdifenyleter*	OC-BR
BDE66*	2,3',4',6- tetrabromodiphenyl ether	*2,3',4',6- tetrabromdifenyleter*	OC-BR
BDE71*	2,3',4',6- tetrabromodiphenyl ether	*2,3',4',6- tetrabromdifenyleter*	OC-BR
BDE77	3,3',4,4'-tetrabromodiphenyl ether	*3,3',4,4'-tetrabromdifenyleter*	OC-BR
BDE85	2,2',3,4,4'-pentabromodiphenyl ether	*2,2',3,4,4'-pentabromdifenyleter*	OC-BR
BDE99	2,2',4,4',5-pentabromodiphenyl ether	*2,2',4,4',5-pentabromdifenyleter*	OC-BR
BDE100	2,2',4,4',6-pentabromodiphenyl ether	*2,2',4,4',6-pentabromdifenyleter*	OC-BR
BDE119	2,3',4,4',6-pentabromodiphenyl ether	*2,3',4,4',6-pentabromdifenyleter*	OC-BR
BDE126	3,3',4,4',5'-pentabromodiphenyl ether	3,3',4,4',5'-*pentabromdifenyleter*	OC-BR

Abbreviation[1]	English	*Norwegian*	Param. group
BDE138	2,2',3,4,4',5'-hexabromodiphenyl ether	*2,2',3,4,4',5'-heksabromdifenyleter*	OC-BR
BDE153	2,2',4,4',5,5'-hexabromodiphenyl ether	*2,2',4,4',5,5'-heksabromdifenyleter*	OC-BR
BDE154	2,2',4,4',5,6'-hexabromodiphenyl ether	*2,2',4,4',5,6'-heksabromdifenyleter*	OC-BR
BDE183	2,2',3,4,4',5',6-heptabromodiphenyl ether	*2,2',3,4,4',5',6-heptabromdifenyleter*	OC-BR
BDE196	2,2',3,3',4,4',5',6-octabromodiphenyl ether	*2,2',3,3',4,4',5',6-octabromdifenyleter*	OC-BR
BDE205	2,2',3,3',4,4',5,5',6'-nonabromodiphenyl ether	*2,2',3,3',4,4',5,5',6'-nonabromdifenyleter*	OC-BR
BDE209	decabromodiphenyl ether	*Dekabromdifenyleter*	OC-BR
BDE5S	sum of BDE -85, -99, -100, -119	*sum av BDE -85, -99, -100, -119*	OC-BR
BDESS	sum of all BDEs	*sum av alle BDEer*	OC-BR
HBCDD	hexabromocyclododecane (1 2 5 6 9 10 hexabromocyclododecane)	*heksabromsyklododekan (1 2 5 6 9 10 heksabromsyklododekan)*	OC-BR
HBCDA	α–hexabromocyclododecane	*α–heksabromsyklododekan*	OC-BR
HBCDB	β-hexabromocyclododecane	*β-heksabromsyklododekan*	OC-BR
HBCDG	γ-hexabromocyclododecane	*γ-heksabromsyklododekan*	OC-BR
TBBPA	tetrabrombisphenol A	*tetrabrombisfenol A*	OC-CP
BPA	bisphenol A	*bisfenol A*	OC-CP
PFAS	**perfluorinated alkylated substances**	***Perfluoralkylerte stoffer***	
PFBS	perfluorobutane sulfonate	*perfluorbutan sulfonat*	PFAS
PFDCA	perfluorodecanoic acid	*perfluordekansyre*	PFAS
PFDCS	ammonium henicosafluorodecanesulphonate	*ammonium henikosafluordekansulfonat*	PFAS
PFHxA	perfluorohexanoic acid	*perfluorhexansyre*	PFAS
PFHpA	perfluoroheptanoic acid	*perfluorheptansyre*	PFAS
PFOA	perfluorooctanoic acid	*perfluoroktansyre*	PFAS
PFNA	perfluorononanoic acid	*perfluornonansyre*	PFAS
PFOS	perfluoroctanoic sulfonate	*perfluoroktansulfonat*	PFAS
PFOSA	perfluoroctanesulfonic amide	*perfluoroktansulfonamid*	PFAS
PFUDA	perfluoroundecanoic acid	*perfluorundekansyre*	PFAS
SCCP	short chain chlorinated paraffins, C_{10-13}	*kortkjedete klorerte parafiner, C_{10-13}*	
MCCP	medium chain chlorinated, C_{14-17} paraffins	*mediumkjedete klorerte parafiner, C_{14-17}*	
Alkylphenols	**phenols/chlorophenols**	***fenoler/klorfenoler***	
4-n-NP	4-n-nonylphenol	*4-n-nonylfenol*	
4-n-OP	4-n-octylphenol	*4-n-oktylfenol*	

Abbreviation[1]	English	*Norwegian*	Param. group
4-t-NP	4-tert-nonylphenol	*4-tert-nonylfenol*	
4-t-OP	4-tert-octylphenol	*4-tert-oktylfenol*	
PFRs	**Phosphorus Flame Retardants**	***Fosforflammehemmere***	
TIBP	tri-*iso*-butylphosphate	*tri-iso-butylfosfat*	
TBP	tributylphosphate	*tributylfosfat*	
TCEP	tri(2-chloroethyl)phosphate	*tri(2-kloretyl)fosfat*	
TCPP	tri(1-chloro-2-propyl)phosphate	*tri(1-klor-2-propyl)fosfat*	
TDCP	tri(1,3-dichloro-2-propyl)phosphate	*tri(1,3-diklor-2-propyl)fosfat*	
TBEP	tri(2-butoxyethyl)phosphate	*tri(2-butokysetyl)fosfat*	
TPhP	triphenylphosphate	*trifenylfosfat*	
EHDPP	2-ethylhexyl-di-phenylphosphate	*2-etylheksyl-difenylfosfat*	
V6	tetrekis(2-chlorethyl)dichloroisopentyldiphosphate	*tetrakis-(2-kloroetyl)diklorisopentyldifosfat*	
DBPhP	dibutylphenylphosphate	*dibutylfenylfosfat*	
BdPhP	butyldiphenylphosphate	*butyldifenylfosfat*	
TEHP	tris(2-etylhexyl)phosphate	*tris(2-etylheksyl)fosfat*	
ToCrP	tris-o-cresylphosphate	*tris-o-kresylfosfat*	
TCrP	tricresyl phosphate	*trikresylfosfat*	
	stable isotopes	***stabile isotoper***	
C/N	$\delta^{13}C$ /$\delta^{15}N$	$\delta^{13}C$ /$\delta^{15}N$	
Delta15N	$\delta^{15}N$	$\delta^{15}N$	
Delta13C	$\delta^{13}C$	$\delta^{13}C$	
	phthalates/organic esters	***phtalater/organiske estere***	
BBP	benzylbutylphthalate	*benzylbutylftalat*	
DBP[6]	dibutylphthalate	*dibutylftalat*	
DBPA	dibutyladipat	*dibutyladipat*	
DEHA	diethylhexcyladipate	*dietylheksyladipat*	
DEHP	di(2-ethylhexyl)-phthalate	*di(2-etylhexyl)-ftalat*	
DEP	dietylphthale	*dietylftalat*	
DEPA	diethyladipat	*dietyladipat*	
DIBP	diisobutylphthalate	*diisobutylftalat*	
DIDP	diisodectylyphthalate	*diisodekylftalat*	
DIHP	diisoheptylphthalate	*diisoheptylftalat*	
DINCH	1,2-Cyclohexane dicarboxylic acid diisononyl ester	*1,2-sykloheksan dikarboksylik syre diisononyl ester*	
DIPA	diisobutyl adipate	*diisobutyladipat*	
DMP	dimethylphthalate	*dimetylftalat*	
DNOP	di-n-octylphthalte	*di-n-oktylftalt*	
DPF	diphenylphthalate	*difenylftalat*	
SDD	dinonylphthalte+diisononylphthalate	*dinonylftalat+diisononylftalat*	
TBP	tributylphosphate	*tributylfosfat*	

Abbreviation[1]	English	*Norwegian*	Param. group
TOA	tributyl-o-acetylcitrate	*tributyl-o-acetylcitrate*	
Triclosan	triclosan	*triklosan*	
[not defined]	dodecylfenol	*dodecylfenol*	
Diuron	Duiron	*Durion*	
Irgarol	Irgarol	*Irgarol*	
NTOT	total organic nitrogen	*total organisk nitrogen*	I-NUT
CTOT	total organic carbon	*total organisk karbon*	O-MAJ
CORG	organic carbon	*organisk karbon*	O-MAJ
GSAMT	grain size	*kornfordeling*	P-PHY
MOCON	moisture content	*vanninnhold*	P-PHY
Specific biological effects methods			
ALAD	δ-aminolevulinic acid dehydrase inhibition	*δ-aminolevulinsyre dehydrase*	BEM
CYP1A	cytochrome P450 1A-protein	*cytokrom P450 1A-protein*	BEM
EROD-activity	Cytochrome P4501A-activity (CYP1A/P4501A1, EROD)	*cytokrom P450 1A-aktivitet*	BEM
OH-pyrene	Pyrene metabolite	*pyren metabolitt*	BEM
VSDI	Vas Deferens Sequence Index		BEM
INSTITUTES			
EFDH	Eurofins [DK]	*Eurofins [DK]*	
EFNO	Eurofins [N, Moss]	*Eurofins [N, Moss]*	
EFGFA	Eurofins [DE, GFA]	*Eurofins [DE, GFA]*	
EFSofia	Eurofins [DE, Sofia]	*Eurofins [DE, Sofia]*	
FIER	Institute for Nutrition, Fisheries Directorate	*Fiskeridirektoratets Ernæringsinstitutt*	
FORC	FORCE Institutes, Div. for Isotope Technique and Analysis [DK]	*FORCE Institutterne, Div. for Isotopteknik og Analyse [DK]*	
GALG	GALAB Laboratories Gmbh [D]	*GALAB Laboratories Gmbh [D]*	
IFEN	Institute for Energy Technology	*Institutt for energiteknikk*	
IMRN	Institute of Marine Research (IMR)	*Havforskningsinstituttet*	
NACE	Nordic Analytical Center	*Nordisk Analyse Center*	
NILU	Norwegian Institute for Air Research	*Norsk institutt for luftforskning*	
NIVA	Norwegian Institute for Water Research	*Norsk institutt for vannforskning*	
SERI	Swedish Environmental Research Institute	*Institutionen för vatten- och luftvårdsforskning*	

Abbreviation[1]	English	*Norwegian*	Param. group
SIIF	Fondation for Scientific and Industrial Research at the Norwegian Institute of Technology-SINTEF (a division, previously: Center for Industrial Research SI)	*Stiftelsen for industriell og teknisk forskning ved Norges tekniske høgskole- SINTEF (en avdeling, tidligere: Senter for industriforskning SI)*	
VETN	Norwegian Veterinary Institute	*Veterinærinstituttet*	
VKID	Water Quality Institute [DK]	*Vannkvalitetsintitutt [DK]*	

1) After: ICES Environmental Data Reporting Formats. International Council for the Exploration of the Sea. July 1996 and supplementary codes related to non-ortho and mono-ortho PCBs and “dioxins” (ICES pers. comm.)

2) Indicates "PAH" compounds that are dicyclic and not truly PAHs typically identified during the analyses of PAH, include naphthalenes and "biphenyls".

3) Indicates the sum of tri- to hexacyclic PAH compounds named in EPA protocol 8310 minus naphthalene (dicyclic), so that the Klif classification system can be applied

4) Indicates PAH compounds potentially cancerogenic for humans according to IARC (1987, updated 14.August 2007 at http://monographs.iarc.fr/ENG/Classification/crthgr01.php), i.e., categories 1, 2A, and 2B (are, possibly and probably carcinogenic). NB.: the update includes Chrysene as cancerogenic and hence, KPAH with Chrysene should not be used in Klif's classification system for this sum-variable (Molvær *et al.* 1997).

5) Indicates non ortho- co-planer PCB compounds i.e., those that lack Cl in positions 1, 1', 5, and 5'

6) DBP is ambiguous; a code for both a PAH and an phthalate. DBP as a PAH was only measured in 1992 whereas DBP as an phthalate has been measure in 2012 and 2013. A correction in the data base is needed in this regard.

*) The Pesticide Index, second edition. The Royal Society of Chemistry, 1991.

Other abbreviations *andre forkortelser*

	English	*Norwegian*
TEQ	"Toxicity equivalency factors" for the most toxic compounds within the following groups:	*"Toxisitetsekvivalentfaktorer" for de giftigste forbindelsene innen følgende grupper.*
	• polychlorinated dibenzo-p-dioxins and dibenzofurans (**PCDD/PCDFs**). Equivalents calculated after Nordic model (Ahlborg 1989) [1] or international model (Int./EPA, cf. Van den Berg *et al.* 1998) [2]	• *polyklorerte dibenzo-p-dioksiner og dibenzofuraner (**PCDD/PCDF**). Ekvivalentberegning etter nordisk modell (Ahlborg 1989) [1] eller etter internasjonal modell (Int./EPA, cf. Van den Berg et al. 1998) [2]*
	• non-ortho and mono-ortho substituted chlorobiphenyls after WHO model (Ahlborg *et al.* 1994) [3] or Safe (1994, cf. NILU pers. comm.)	• *non-orto og mono-orto substituerte klorobifenyler etter WHO modell (Ahlborg et al. 1994) [3] eller Safe (1994, cf. NILU pers. medd.)*
ppm	parts per million, mg/kg	*deler pr. milliondeler, mg/kg*
ppb	parts per billion, μg/kg	*deler pr. milliarddeler, μg/kg*
ppp	parts per trillion, ng/kg	*deler pr. tusen-milliarddeler, ng/kg*
d.w.	dry weight basis	*tørrvekt basis*
w.w.	wet weight or fresh weight basis	*våtvekt eller friskvekt basis*

[1]) Ahlborg, U.G., 1989. Nordic risk assessment of PCDDs and PCDFs. Chemosphere 19:603-608.

[2]) Van den Berg, Birnbaum, L, Bosveld, A. T. C. and co-workers, 1998. Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. Environ Hlth. Perspect. 106:775-792.

[3]) Ahlborg, U.G., Becking G.B., Birnbaum, L.S., Brouwer, A, Derks, H.J.G.M., Feely, M., Golor, G., Hanberg, A., Larsen, J.C., J.C., Liem, A.K.G., Safe, S.H., Schlatter, C., Wärn, F., Younes, M., Yrjänheikki, E., 1994. Toxic equivalency factors for dioxin-like PCBs. Report on a WHO-ECEH and IPSC consultation, December 1993. Chemosphere 28:1049-1067.

Appendix C
Classification of environmental quality

***Table 34**. Norwegian Environment Agency classification system of contaminants in blue mussel and fish (Molvær et al. 1997) and proposed revisions (shaded) for Class I concentrations (Knutzen & Green 2001) used in this report.*

Contaminant			Classification (upper limit for Classes I-IV) *Degree of pollution*				
			I *Insignificant*	II *Moderate*	III *Marked*	IV *Severe*	V *Extreme*
Blue mussel							
Arsenic (As)	mg/kg	w.w. [2]	<2	6	20	40	>40
	mg/kg	d.w.	<10	30	100	200	>200
Cadmium (Cd)	mg/kg	w.w. [2]	<0.4	1	4	8	>8
	mg/kg	d.w.	<2	5	20	40	>40
Copper (Cu)	mg/kg	w.w. [2]	<2	6	20	40	>40
	mg/kg	d.w.	<10	30	100	200	>200
Chromium (Cr)	mg/kg	w.w. [2]	<0.6	2	6	12	>12
	mg/kg	d.w.	<3	10	30	60	>60
Lead (Pb)	mg/kg	w.w. [2]	<0.6	3	8	20	>20
	mg/kg	d.w.	<3	15	40	100	>100
Mercury (Hg)	mg/kg	w.w. [2]	<0.04	0.1	0.3	0.8	>0.8
	mg/kg	d.w.	<0.2	0.5	1.5	4	>4
Nickel (Ni)	mg/kg	w.w. [2]	<1	5	10	20	>20
	mg/kg	d.w.	<5	25	50	100	>100
Silver (Ag)	mg/kg	w.w. [2]	<0.06	0.2	0.4	1	>1
	mg/kg	d.w.	<0.3	1	2	5	>5
Zinc (Zn)	mg/kg	w.w. [2]	<40	80	200	500	>500
	mg/kg	d.w.	<200	400	1000	2500	>2500
TBT [1]	mg/kg	d.w.	<0.1	0.5	2	5	>5
∑PCB-7	µg/kg	w.w.	<3 [5]	15	40	100	>100
		d.w. [2]	<15 [2]	75	200	500	>500
∑DDT [11]	µg/kg	w.w.	<2	5	10	30	>30
		d.w. [2]	<10	25	50	150	>150
∑HCH [12]	µg/kg	w.w.	<1	3	10	30	>30
		d.w. [2]	<5	15	50	150	>150
HCB	µg/kg	w.w.	<0.1	0.3	1	5	>5
		d.w. [2]	<0.5	1.5	5	25	>25
∑PAH [13]	µg/kg	w.w.	<50	200	2000	5000	>5000
		d.w. [2]	<250	1000	10000	25000	>25000
∑KPAH	µg/kg	w.w.	<10	30	100	300	>300
		d.w. [2]	<50	150	500	1500	>1500
B[*a*]P	µg/kg	w.w.	<1	3	10	30	>30
		d.w. [2]	<5	15	50	150	>150
$TE_{PCDF/D}$ [3]	µg/t [4]	w.w.	<0.2	0.5	1.5	3	>3
Cod, fillet							
Mercury (Hg)	mg/kg	w.w.	<0.1	0.3	0.5	1	>1
∑PCB-7	µg/kg	w.w.	<3 [6]	20	50	150	>150
∑DDT [11]	µg/kg	w.w.	<1	3	10	25	>25
∑HCH [12]	µg/kg	w.w.	<0.3 [7]	2	5	15	>15
HCB	µg/kg	w.w.	<0.2	0.5	2	5	>5
$TE_{PCDF/D}$	ng/kg	w.w.	< 0.1	0.3	1	2	> 2
Cod, liver							
∑PCB-7	µg/kg	w.w.	<500	1500	4000	10000	>10000
∑DDT [11]	µg/kg	w.w.	<200 [8]	500	1500	3000	>3000

Contaminant			Classification (upper limit for Classes I-IV) *Degree of pollution*				
			I *Insignificant*	II *Moderate*	III *Marked*	IV *Severe*	V *Extreme*
∑HCH[12)]	µg/kg	w.w.	<30 [9)]	200	500	1000	>1000
HCB	µg/kg	w.w.	<20	50	200	400	>400
$TE_{PCDF/D}$ [3)]	µg/t [4)]	w.w.	<10 [10)]	40	100	300	>300
Flounder, fillet							
∑PCB-7	µg/kg	w.w.	<5	20	50	150	>150
∑DDT[11)]	µg/kg	w.w.	<2	4	15	40	>40
∑HCH[12)]	µg/kg	w.w.	<1	3	10	30	>30
HCB	µg/kg	w.w.	<0.2	0.5	2	5	>5
$TE_{PCDF/D}$	ng/kg	w.w.	<0.1	0.3	1	3	>3

[1]) Tributyltin on a formula basis

[2]) Conversion assuming 20% dry weight

[3]) TCDDN (Appendix B)

[4]) µg/t = µg/ton = g/1000 kg (Appendix B)

[5]) Blue mussel-ΣPCB7: Decrease limit from 4 to 3

[6]) Cod fillet-ΣPCB7: Decrease limit from 5 to 3

[7]) Cod fillet-ΣHCH: Decrease limit from 0.5 to 0.3

[8]) Cod liver-ΣDDT: Proposal to either increase limit from 200 to 300 or, preferably, replace ΣDDT with p,p'-DDE and keep the limit (Knutzen & Green 2001)

[9]) Cod liver-ΣHCH: Decrease limit from 50 to 30

[10]) Cod liver: TEPCDD/PCDF: Decrease limit from 15 to 10

[11]) Used in this investigation also for ppDDE

[12]) Used in this investigation also for γ-HCH (lindane)

[13]) The sum of tri- to hexacyclic PAH compounds named in EPA protocol 8310 minus naphthalene (dicyclic)-totalling 15 compounds, so that the Klif classification system can be applied

***Table 35**. OSPAR classification of vas deferens sequence index (VDSI) in dog whelk (OSPAR 2013). For this report, the short name for each class ("Insignificant", "Moderate", etc) has been adopted from the Norwegian Environment Agency classification system. OSPAR has a sixth class, not shown here and not applied in this report, that indicates that* dog whelks were absent or expired.

		Classification (upper limit for Classes A-E) *Degree of pollution*				
		A *Insignificant* [1)]	B *Moderate* [2)]	C *Marked* [3)]	D *Severe* [4)]	E *Extreme* [5)]
VDSI		0.3	2	4	5	>5

[1]) The level of imposex in the more sensitive gastropod species is close to zero (0-30 % of females have imposex) indicating exposure to TBT concentrations close to zero, which is the objective in the OSPAR Hazardous Substances Strategy. [Author's note: this level marks OSPAR's Background Assessment Criteria (BAC)]

[2]) The level of imposex in the more sensitive gastropod species (30--100 % of females have imposex) indicates exposure to TBT concentrations below the exotoxicological assessment cirteria (EAC) derived by OSPAR for TBT. For example, adverse effects in the more sensitive taxa of the ecosystem caused by long-term expsosure to TBT are predicted to be unlikely to occur.

[3]) The level of imposex in the more sensitive gastropod species indicates exposure to TBT concentrations higher than EAC derived for TBT. For example, there is a risk of adverse effects such as reduced growth and recruitment, in the more sensitive taxa of the ecosystem caused by long-term expsosure to TBT.

[4]) The reproductive capacity in the populations of the more sensitive gastropod species, such as *Nucella lapillus,* is afffected as a result of the presence of sterile females, but some repoductively capable females remain. For example, there is evidence of adverse effets that can be dreictly associated with the exposure to TBT.

[5]) Polulations of the more sensitive gastropod species, such as *Nucella lapillus, are unable to reproduce. The majority of, if not all, females within the population have been sterilized*

***Table 36**. Provisional "high background levels" of selected contaminants, in **mg/kg dry weight** (blue mussel) and **mg/kg wet weight** (blue mussel and fish) used in this report. The respective "high background" limits are from Knutzen & Skei (1990) with mostly minor adjustments (Knutzen & Green 1995, 2001; Molvær et al. 1997, Green & Knutzen 2003), except for dab where the suggested limit is based on CEMP-data (Knutzen & Green 1995) and PFOS, PFOSA and S_BDE (Green et al. 2009 and Bakke et al. 2008, see footnote). Especially uncertain values are marked with "?".*

Cont.	Blue mussel [1]		Cod [1]	
			liver	fillet
	mg/kg d.w.	mg/kg w.w.	mg/kg w.w.	mg/kg w.w.
Lead	3.0 [2)]	0.6 [3)]	0.1	
Cadmium	2.0 [2)]	0.4 [3)]	0.3	
Copper	10.0 [2)]	2.0 [3)]	20.0	
Mercury	0.2 [2)]	0.04 [3)]		0.1 [2)]
Zinc	200.0 [2)]	40.0 [3)]	30.0	
∑PCB-7 [8)]	0.015 [3, 9)]	0.003 [2 9)]	0.50 [2)]	0.003 [9)]
ppDDE	0.010 [3)]	0.002 [6)]	0.2 [9)]	
γ HCH	0.005 [3)]	0.001 [6)]	0.03 [9)]	0.0003 [9)]
HCB	0.0005 [3)]	0.0001 [2)]	0.02 [2)]	
TCDDN	0.000001 [3)] 0.0000002 [2)]		0.00001 [9)]	
PFOS [10)]			0.05	
PFOSA [11)]			0.01	
S_BDE [12)]			0.05	

[1]) Respectively: *Mytilus edulis, Gadus morhua, Platichthys flesus* and *Limanda limanda*

[2]) From the Norwegian Environment Agency Class I ("good") (Molvær *et al.* 1997)

[3]) Conversion assuming 20% dry weight

[4]) Approximately 25% of ΣPCB-7 (Knutzen & Green 1995)

[5]) 1.5-2 times 75% quartile (cf. Annex B in Knutzen & Green 1995)

[6]) Assumed equal to limit for ΣDDT or ΣHCH, respectively, from the Norwegian Pollution Control Authority Environmental Class I ("good") (Molvær *et al.* 1997). Hence, limits for ppDDE and γHCH are probably too high (lacking sufficient and reliable reference values)

[7]) Mean plus 2 times standard deviation (cf. Annex B in Knutzen & Green 1995)

[8]) Estimated as sum of 7 individual PCB compounds (CB-28, -52, -101, -118, -138, -153 and -180) and assumed to be ca. 50% and 70% of total PCB for blue mussel and cod/flatfish, respectively

[9]) Flounder liver: Decrease limit from 5 to 3 and from 2 to 1 for ΣPCB7 and p,p-DDE, respectively, with regard to revisions suggested by Knutzen & Green (2001) and Green & Knutzen (2003)

[10]) PFOS in cod liver. Background: West coast, Lofoten: 1-49 µg/kg w.w. (Green *et al.* 2009), Barentshav: 3 – 8 µg/kg w.w. (Bakke *et al.* 2008). Conclusion: 50 µg/kg w.w.

[11]) PFOSA in cod liver. Background: West coast, Lofoten: 1.9-6.1 µg/kg w.w. (Green *et al.* 2009), Barentshav: 3 – 8 µg/kg w.w. (Bakke *et al.* 2008). Conclusion: 10 µg/kg w.w.

[12]) Sum_BDE in cod liver. Background: Norwegian coast, exposed and remote from heavily populated areas: average 12-36 µg/kg w.w. (Green *et al.* 2009). Conclusion: 50 µg/kg w.w.

Appendix D
Map of stations

Nominel station positions 1981-2014
(cf. Appendix E)

Appendix D (cont.) Map of stations

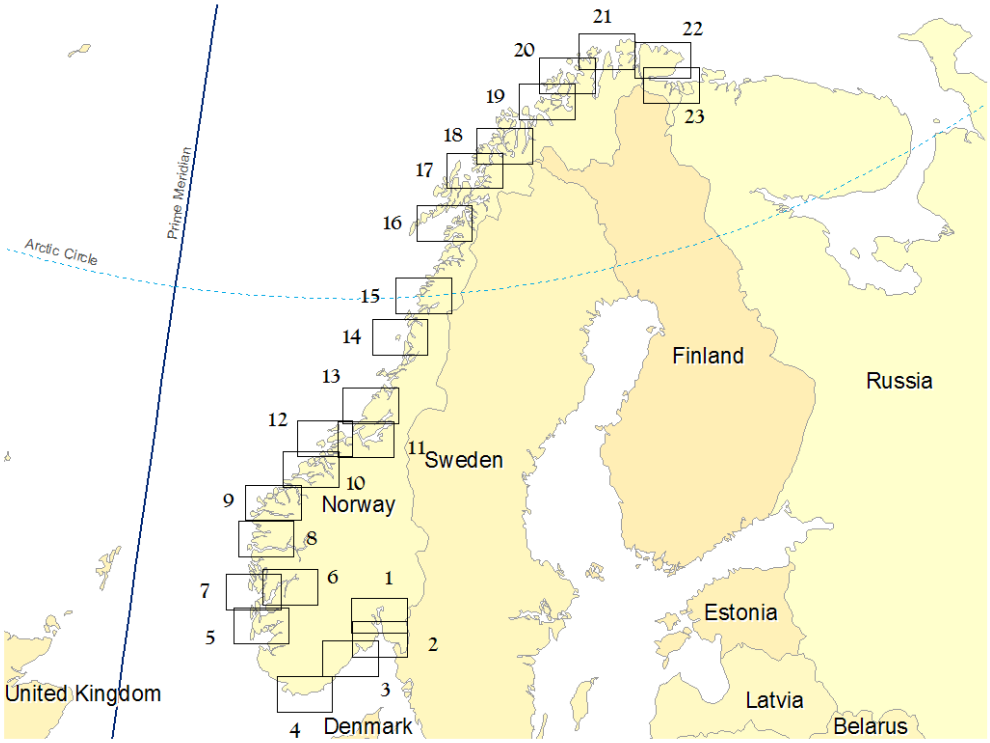
NOTES

The station's nominal position is plotted, and not the specific positions that may have differed from one year to another. The maps are generated using ArcGIS version 9.1.

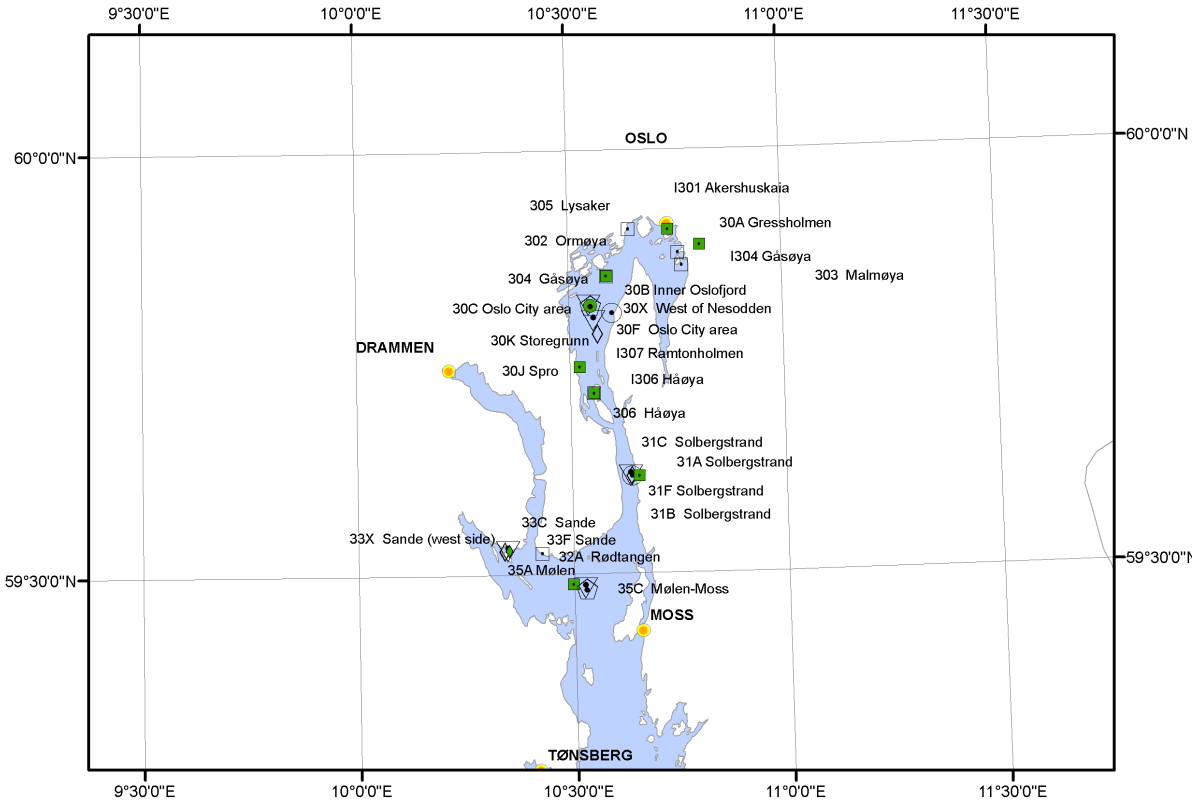
The following symbols and codes apply:

All years	2014	Explanation	Station code
		Sediment	<number>S
		Blue mussel	<number>A
		Blue mussel	I<number/letter> [1]
		Blue mussel	R<number/letter> [1]
		Dog whelk	<number>F
		Prawn	<number>C
		Atlantic cod	<number>A
		Flatfish	<number>D/E
		Other round fish	
		Town or city	

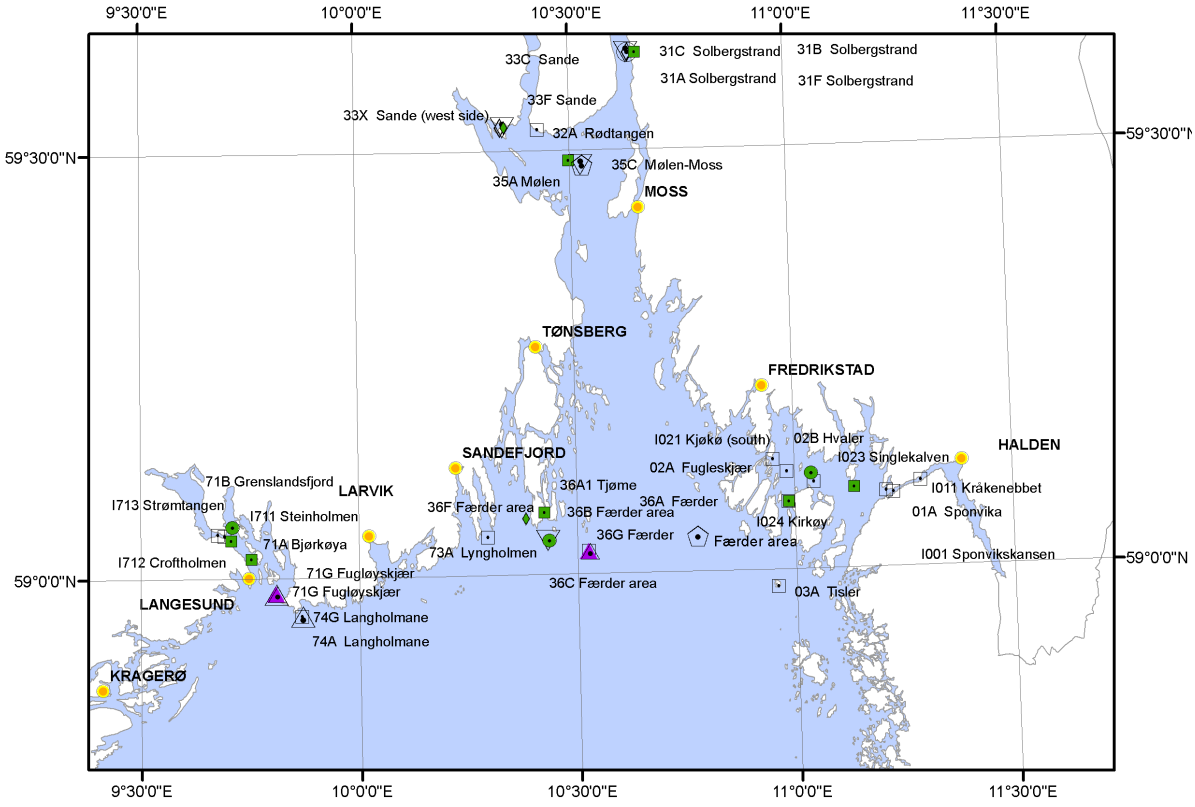
1) Supplementary station used in the blue mussel pollution (I) or reference (R) index of the Norwegian Environment Agency (cf. Green *et al.* 2011b).



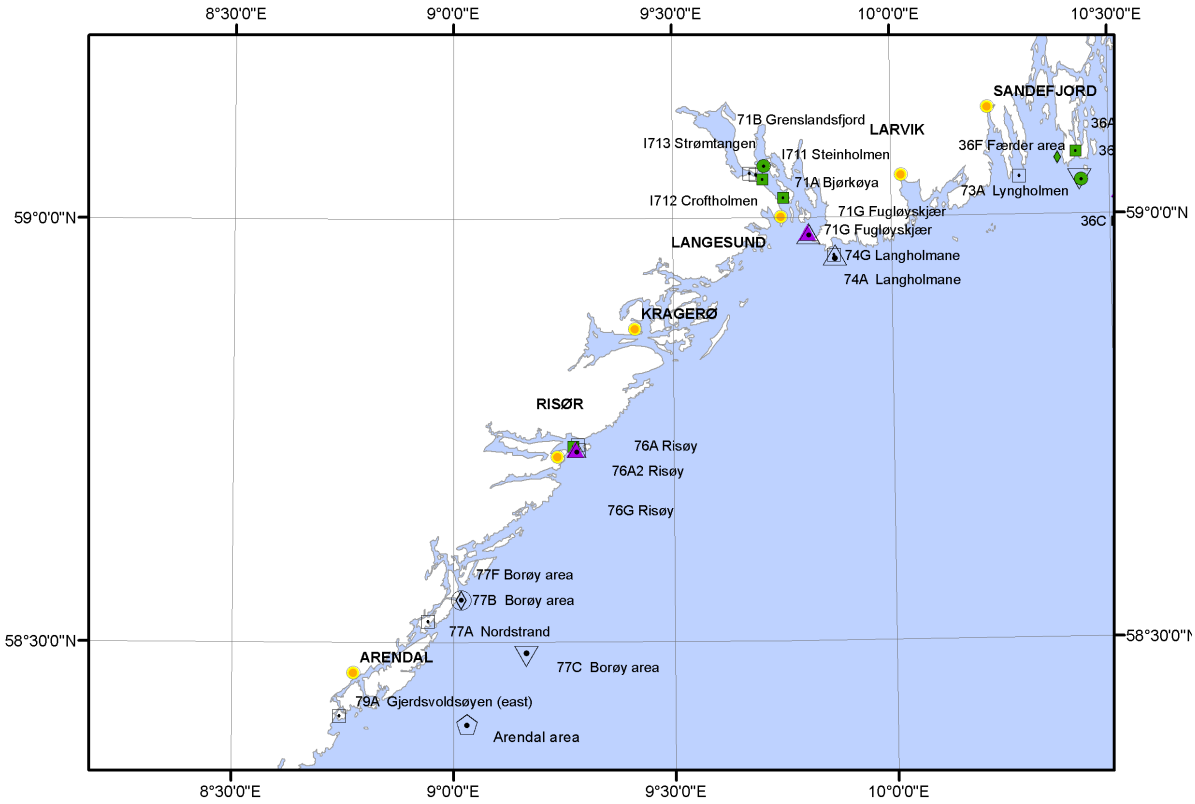
MILKYS stations Norway. Numbers indicate map references that follow.
Note: distance between two lines of latitude is 15 nautical miles (= 27.8 km).


9°30'0"E
10°0'0"E
10°30'0"E
11°0'0"E
11°30'0"E
60°0'0"N
59°30'0"N
OSLO
I301 Akershuskaia
305 Lysaker
30A Gressholmen
302 Ormøya
I304 Gåsøya
303 Malmøya
304 Gåsøya
30B Inner Oslofjord
30C Oslo City area
30X West of Nesodden
30F Oslo City area
DRAMMEN
30K Storegrunn
I307 Ramtonholmen
30J Spro
I306 Håøya
306 Håøya
31C Solbergstrand
31A Solbergstrand
31F Solbergstrand
31B Solbergstrand
33C Sande
33X Sande (west side)
33F Sande
32A Rødtangen
35A Mølen
35C Mølen-Moss
MOSS
TØNSBERG

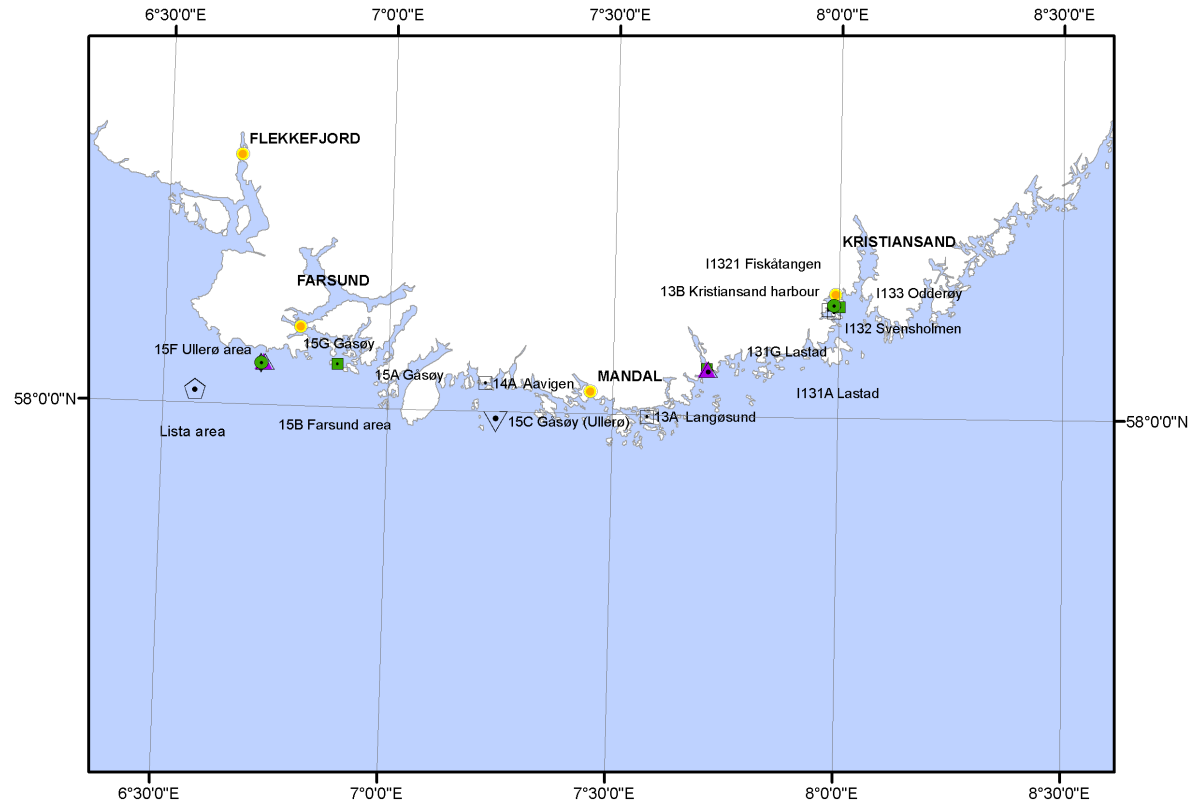
MAP 1


9°30'0"E
10°0'0"E
10°30'0"E
11°0'0"E
11°30'0"E
59°30'0"N
59°0'0"N
33C Sande
31C Solbergstrand
31B Solbergstrand
31A Solbergstrand
31F Solbergstrand
33F Sande
33X Sande (west side)
32A Rødtangen
35C Mølen-Moss
35A Mølen
MOSS
TØNSBERG
FREDRIKSTAD
I021 Kjøkø (south)
02B Hvaler
I023 Singlekalven
HALDEN
SANDEFJORD
02A Fugleskjær
I011 Kråkenebbet
71B Grenslandsfjord
LARVIK
36A1 Tjøme
I713 Strømtangen
36F Færder area
36A Færder
01A Sponvika
I711 Steinholmen
36B Færder area
I024 Kirkøy
36G Færder
Færder area
71A Bjørkøya
73A Lyngholmen
I001 Sponvikskansen
I712 Croftholmen
71G Fugløyskjær
36C Færder area
03A Tisler
71G Fugløyskjær
LANGESUND
74G Langholmane
74A Langholmane
KRAGERØ

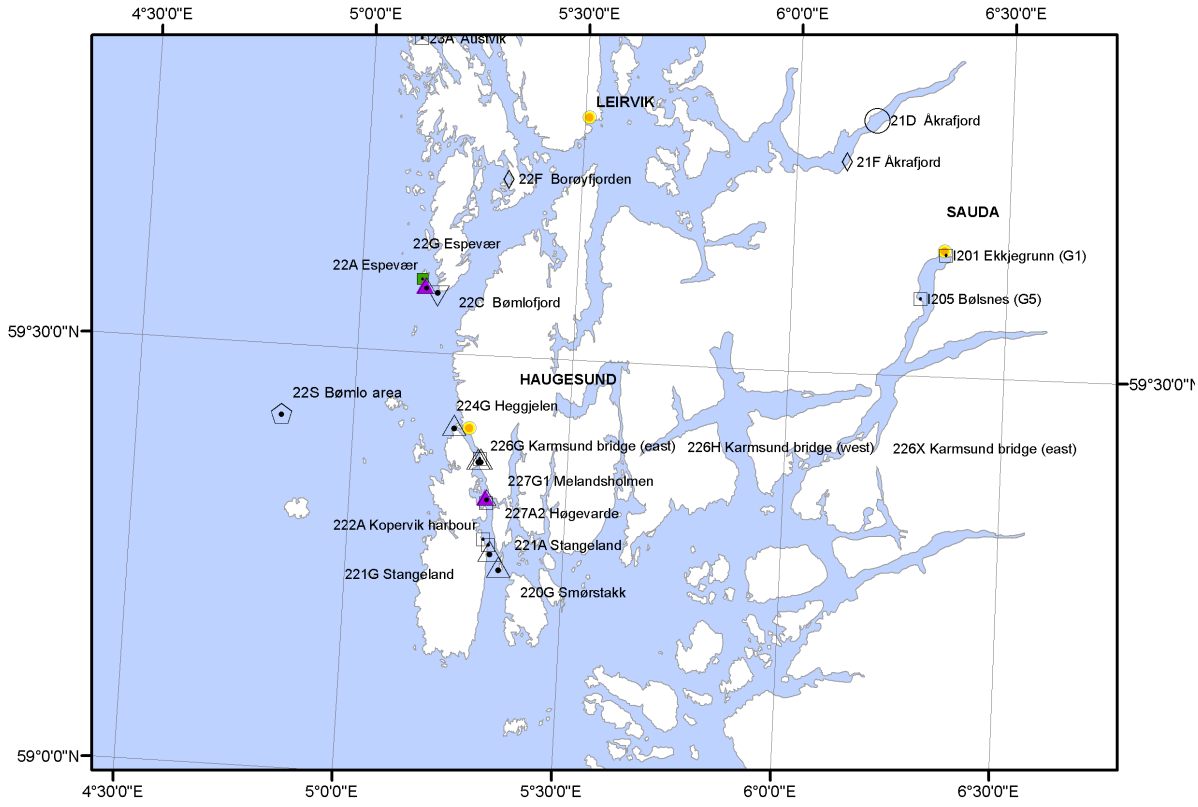
MAP 2


8°30'0"E
9°0'0"E
9°30'0"E
10°0'0"E
10°30'0"E
59°0'0"N
58°30'0"N
SANDEFJORD
LARVIK
71B Grenslandsfjord
I713 Strømtangen
I711 Steinholmen
36F Færder area
71A Bjørkøya
I712 Croftholmen
73A Lyngholmen
71G Fugløyskjær
71G Fugløyskjær
LANGESUND
74G Langholmane
74A Langholmane
KRAGERØ
RISØR
76A Risøy
76A2 Risøy
76G Risøy
77F Borøy area
77B Borøy area
77A Nordstrand
ARENDAL
77C Borøy area
79A Gjerdsvoldsøyen (east)
Arendal area

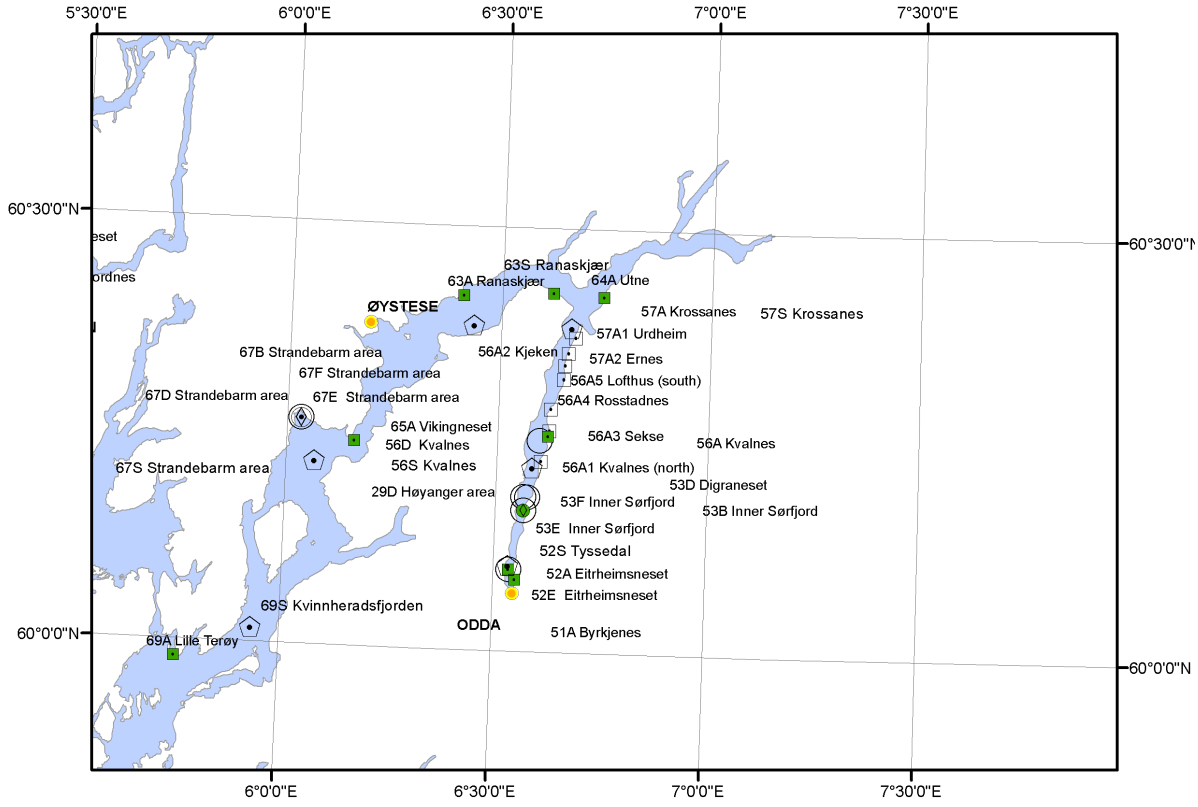
MAP 3


6°30'0"E
7°0'0"E
7°30'0"E
8°0'0"E
8°30'0"E
58°0'0"N
FLEKKEFJORD
KRISTIANSAND
I1321 Fiskåtangen
FARSUND
13B Kristiansand harbour
I133 Odderøy
I132 Svensholmen
15F Ullerø area
15G Gåsøy
131G Lastad
15A Gåsøy
14A Aavigen
MANDAL
I131A Lastad
Lista area
15B Farsund area
15C Gåsøy (Ullerø)
13A Langøsund

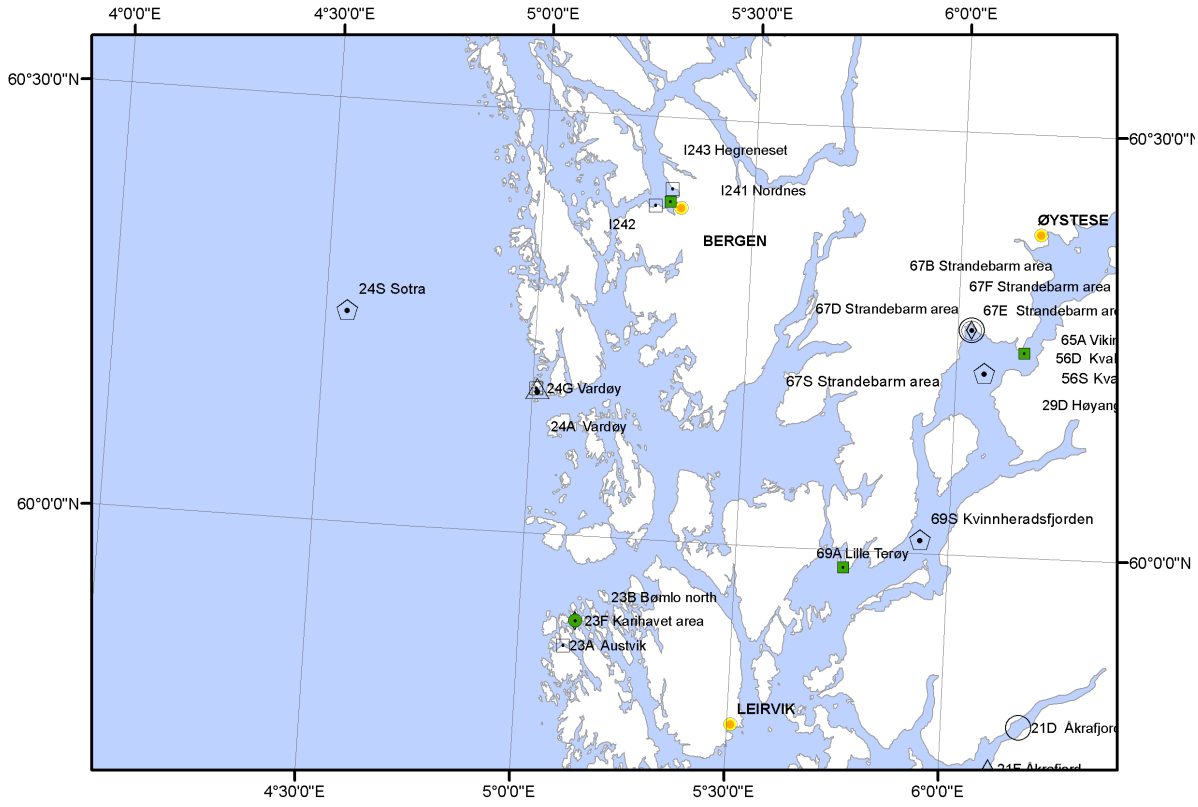
MAP 4


4°30'0"E
5°0'0"E
5°30'0"E
6°0'0"E
6°30'0"E
59°30'0"N
59°0'0"N
23A Austvik
LEIRVIK
21D Åkrafjord
21F Åkrafjord
22F Borøyfjorden
SAUDA
22G Espevær
22A Espevær
201 Ekkjegrunn (G1)
22C Bømlofjord
205 Bølsnes (G5)
HAUGESUND
22S Bømlo area
224G Heggjelen
226G Karmsund bridge (east)
226H Karmsund bridge (west)
226X Karmsund bridge (east)
227G1 Melandsholmen
227A2 Høgevarde
222A Kopervik harbour
221A Stangeland
221G Stangeland
220G Smørstakk

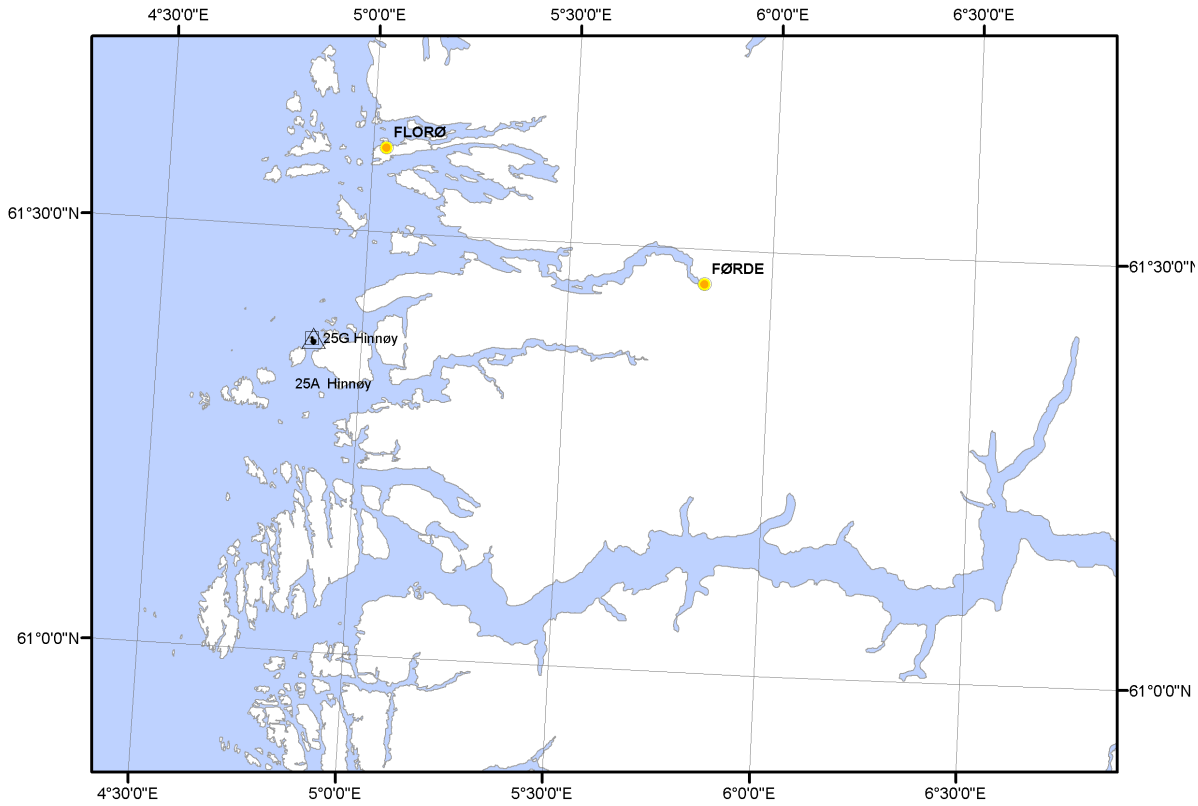
MAP 5


5°30'0"E
6°0'0"E
6°30'0"E
7°0'0"E
7°30'0"E
60°30'0"N
60°0'0"N
63S Ranaskjær
63A Ranaskjær
64A Utne
ØYSTESE
57A Krossanes
57S Krossanes
57A1 Urdheim
67B Strandebarm area
56A2 Kjeken
57A2 Ernes
67F Strandebarm area
56A5 Lofthus (south)
67D Strandebarm area
67E Strandebarm area
56A4 Rosstadnes
65A Vikingneset
56A3 Sekse
56A Kvalnes
56D Kvalnes
67S Strandebarm area
56S Kvalnes
56A1 Kvalnes (north)
53D Digraneset
29D Høyanger area
53F Inner Sørfjord
53B Inner Sørfjord
53E Inner Sørfjord
52S Tyssedal
52A Eitrheimsneset
52E Eitrheimsneset
69S Kvinnheradsfjorden
ODDA
69A Lille Terøy
51A Byrkjenes

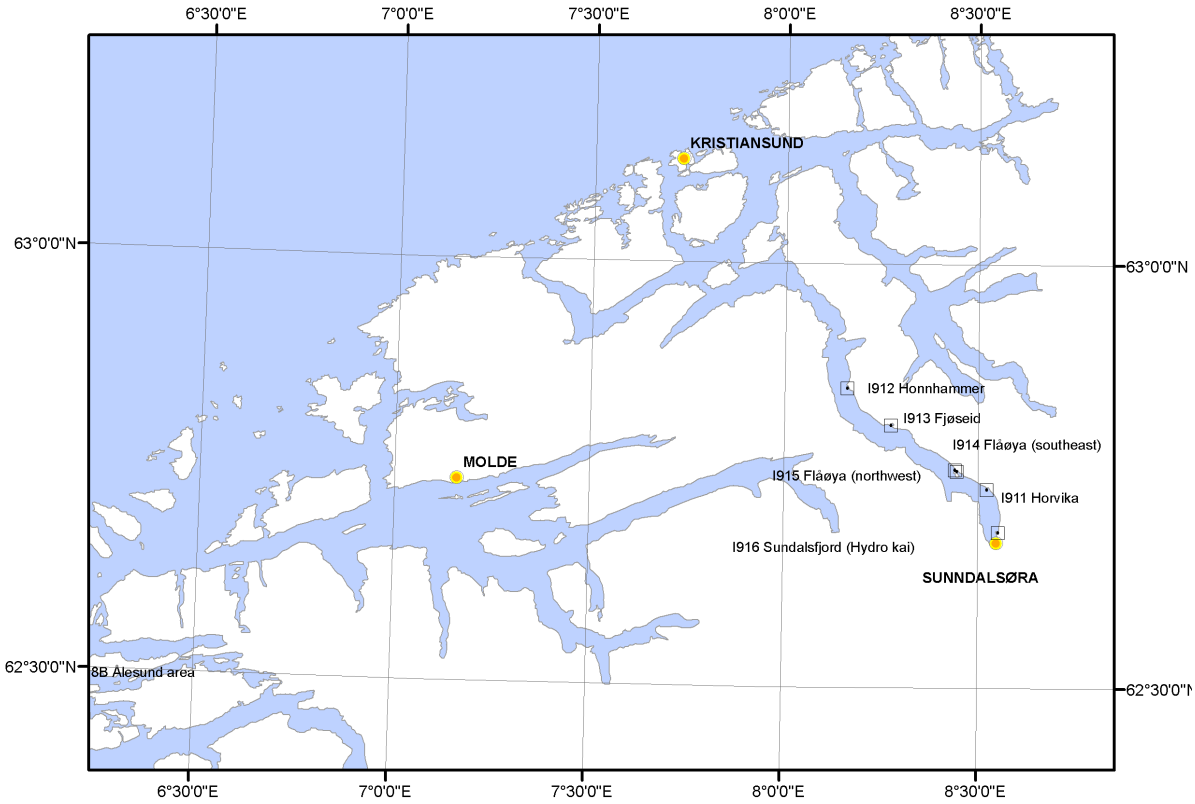
MAP 6


4°0'0"E
4°30'0"E
5°0'0"E
5°30'0"E
6°0'0"E
60°30'0"N
60°0'0"N
I243 Hegreneset
I241 Nordnes
I242
BERGEN
ØYSTESE
67B Strandebarm area
67F Strandebarm area
67D Strandebarm area
67E Strandebarm ar
65A Vikir
56D Kval
56S Kva
29D Høyan
24S Sotra
24G Vardøy
24A Vardøy
67S Strandebarm area
69S Kvinnheradsfjorden
69A Lille Terøy
23B Bømlo north
23F Karihavet area
23A Austvik
LEIRVIK
21D Åkrafjor

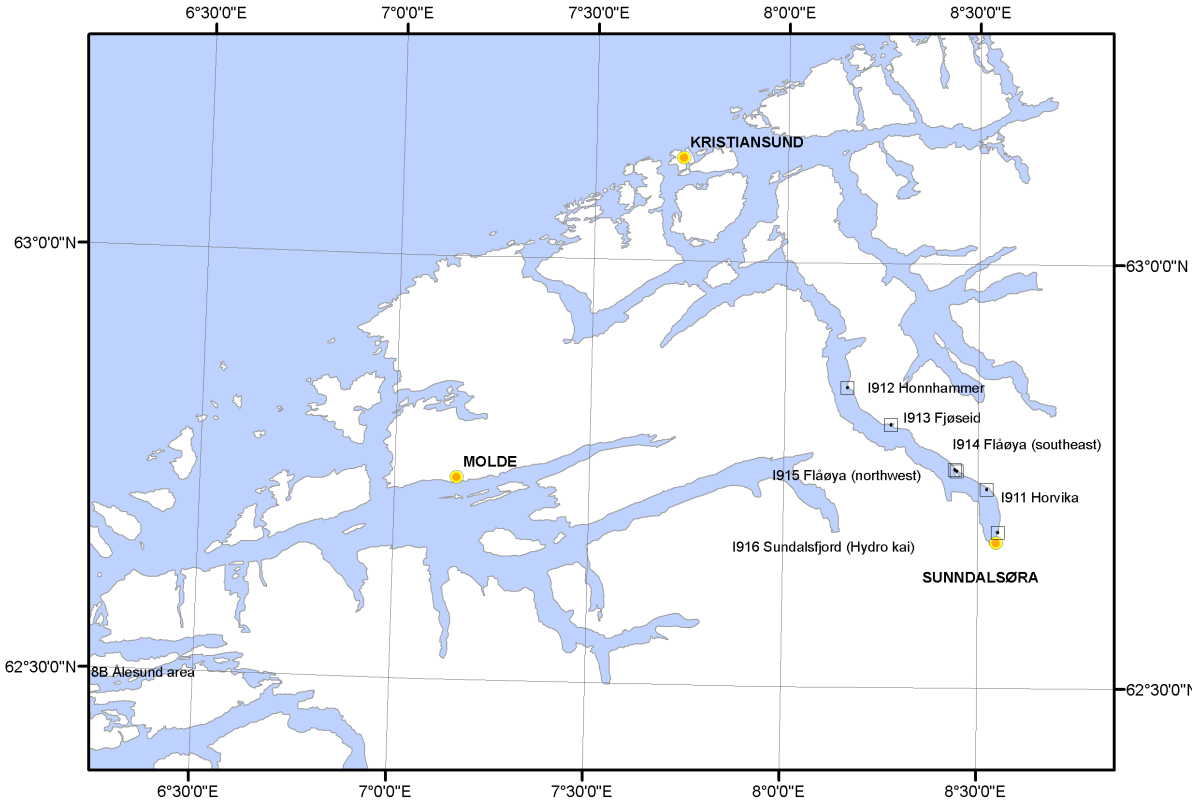
MAP 7


4°30'0"E
5°0'0"E
5°30'0"E
6°0'0"E
6°30'0"E
61°30'0"N
61°0'0"N
FLORØ
FØRDE
25G Hinnøy
25A Hinnøy

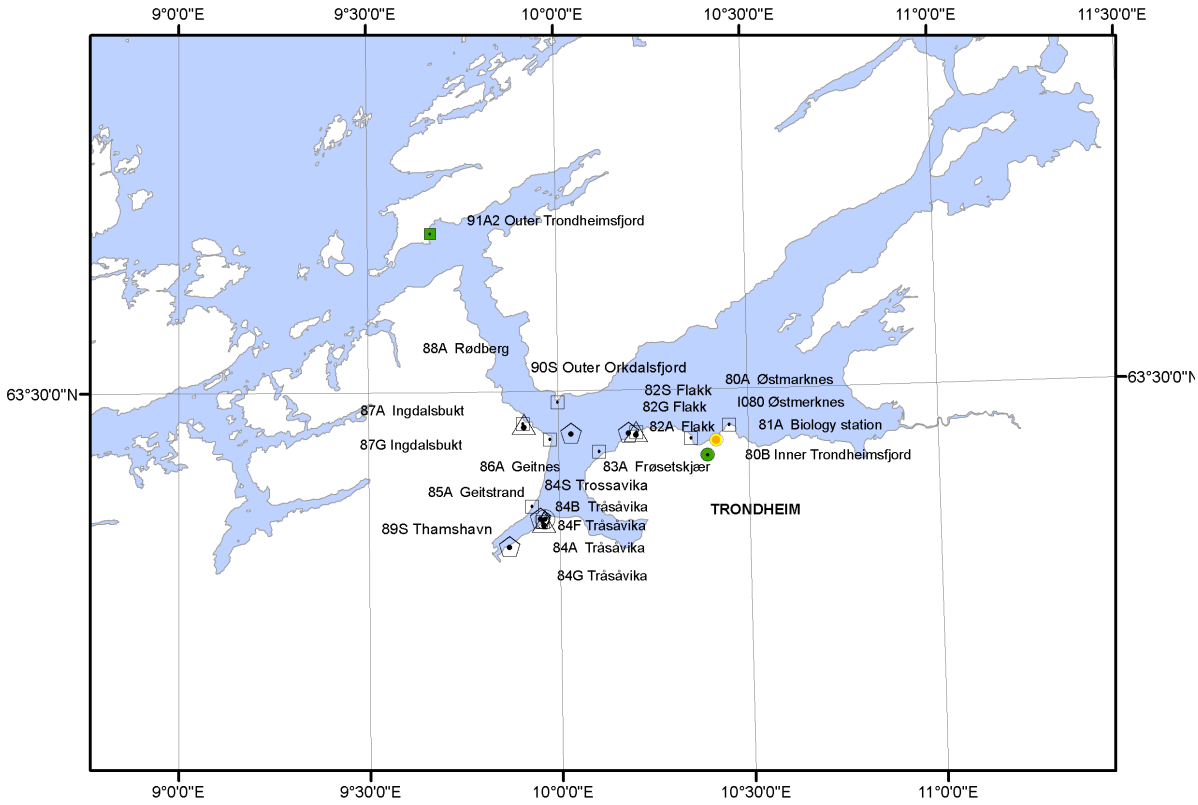
MAP 8


6°30'0"E
7°0'0"E
7°30'0"E
8°0'0"E
8°30'0"E
63°0'0"N
62°30'0"N
KRISTIANSUND
MOLDE
I912 Honnhammer
I913 Fjøseid
I914 Flåøya (southeast)
I915 Flåøya (northwest)
I911 Horvika
I916 Sundalsfjord (Hydro kai)
SUNNDALSØRA
8B Ålesund area

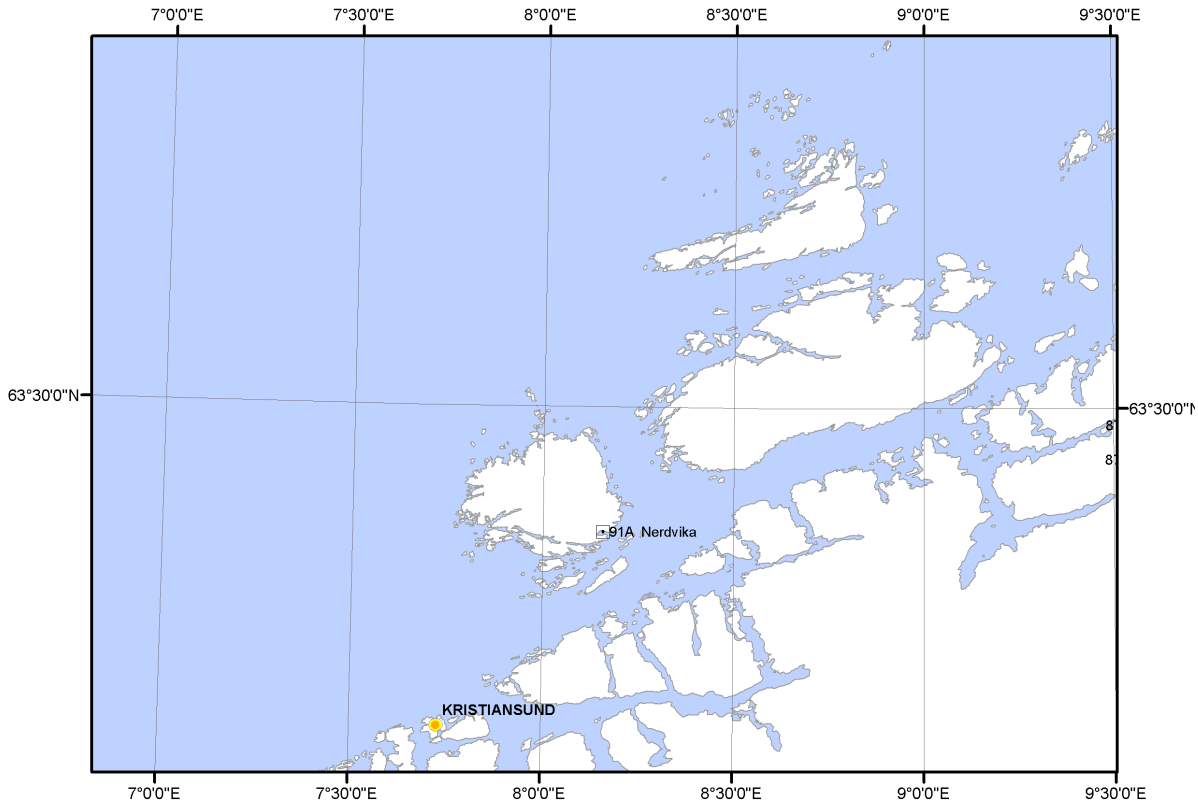
MAP 9


6°30'0"E
7°0'0"E
7°30'0"E
8°0'0"E
8°30'0"E
63°0'0"N
62°30'0"N
KRISTIANSUND
MOLDE
I912 Honnhammer
I913 Fjøseid
I914 Flåøya (southeast)
I915 Flåøya (northwest)
I911 Horvika
I916 Sundalsfjord (Hydro kai)
SUNNDALSØRA
8B Ålesund area

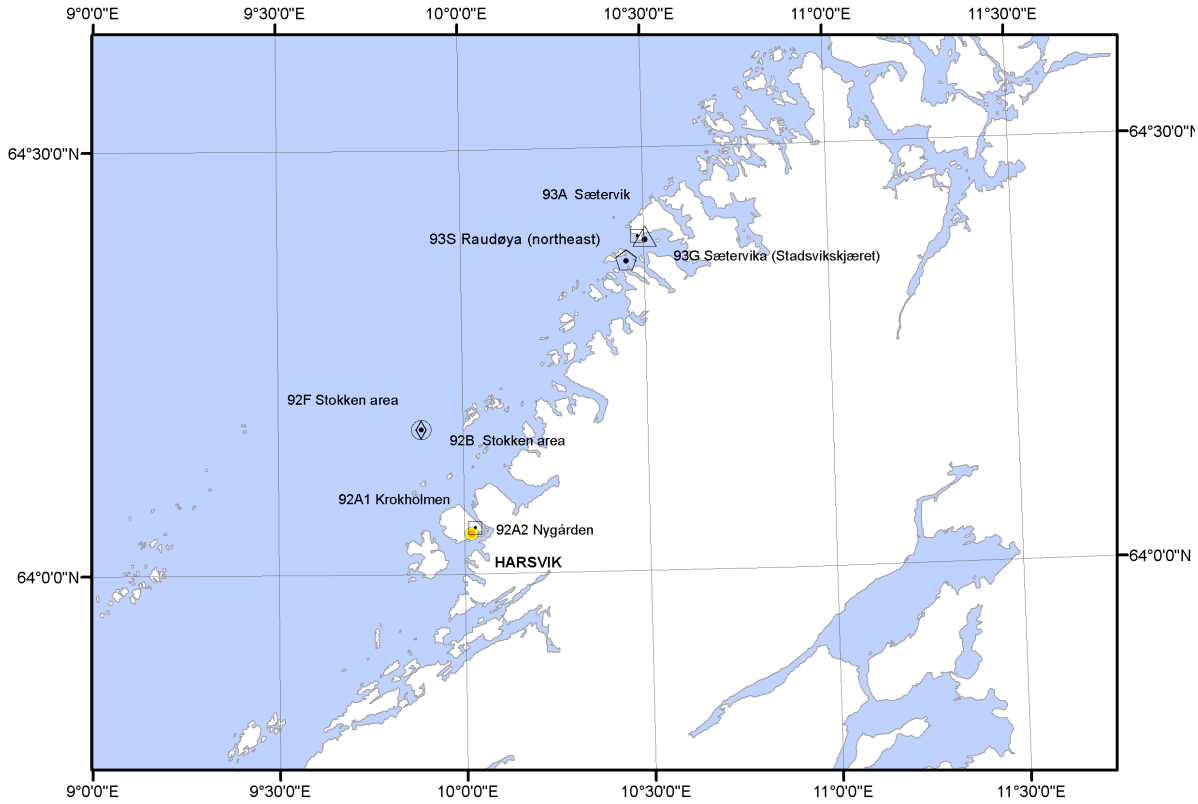
MAP 10


9°0'0"E
9°30'0"E
10°0'0"E
10°30'0"E
11°0'0"E
11°30'0"E
63°30'0"N
91A2 Outer Trondheimsfjord
88A Rødberg
90S Outer Orkdalsfjord
80A Østmarknes
82S Flakk
82G Flakk
I080 Østmerknes
87A Ingdalsbukt
82A Flakk
81A Biology station
87G Ingdalsbukt
80B Inner Trondheimsfjord
86A Geitnes
83A Frøsetskjær
85A Geitstrand
84S Trossavika
84B Tråsåvika
TRONDHEIM
89S Thamshavn
84F Tråsåvika
84A Tråsåvika
84G Tråsåvika

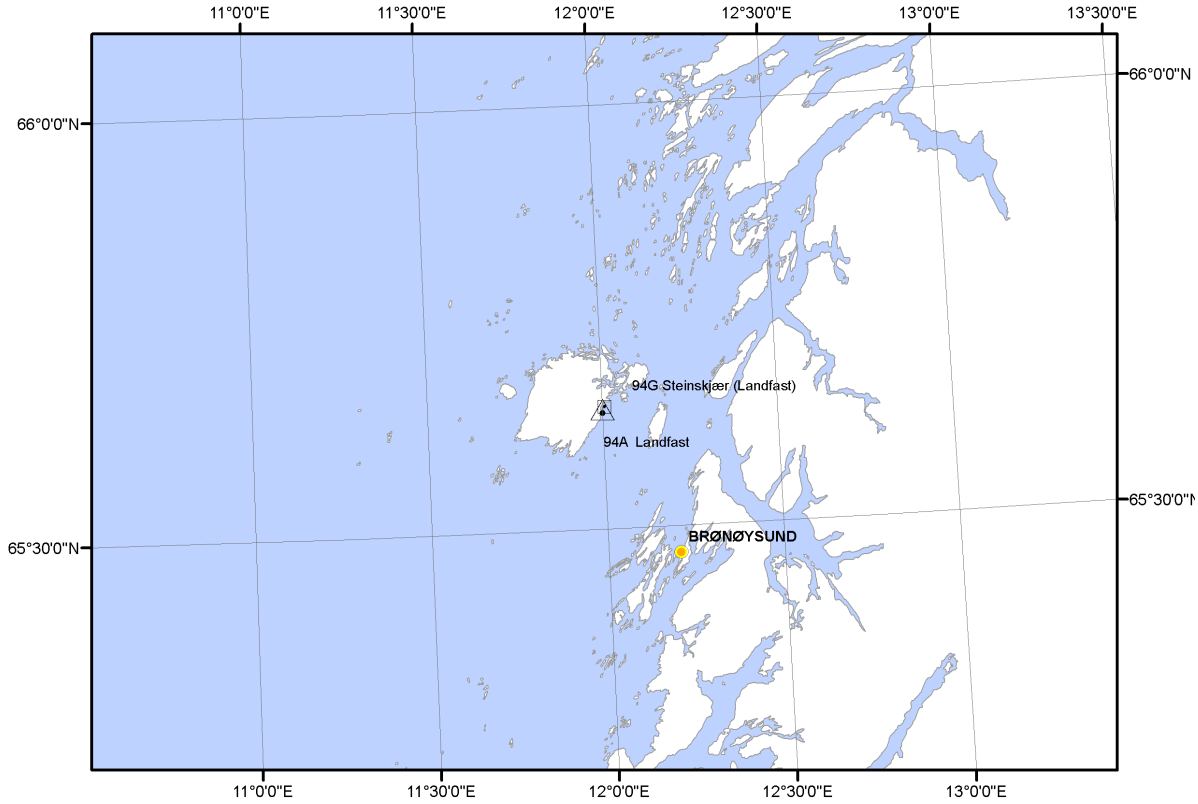
MAP 11


7°0'0"E
7°30'0"E
8°0'0"E
8°30'0"E
9°0'0"E
9°30'0"E
63°30'0"N
91A Nerdvika
KRISTIANSUND

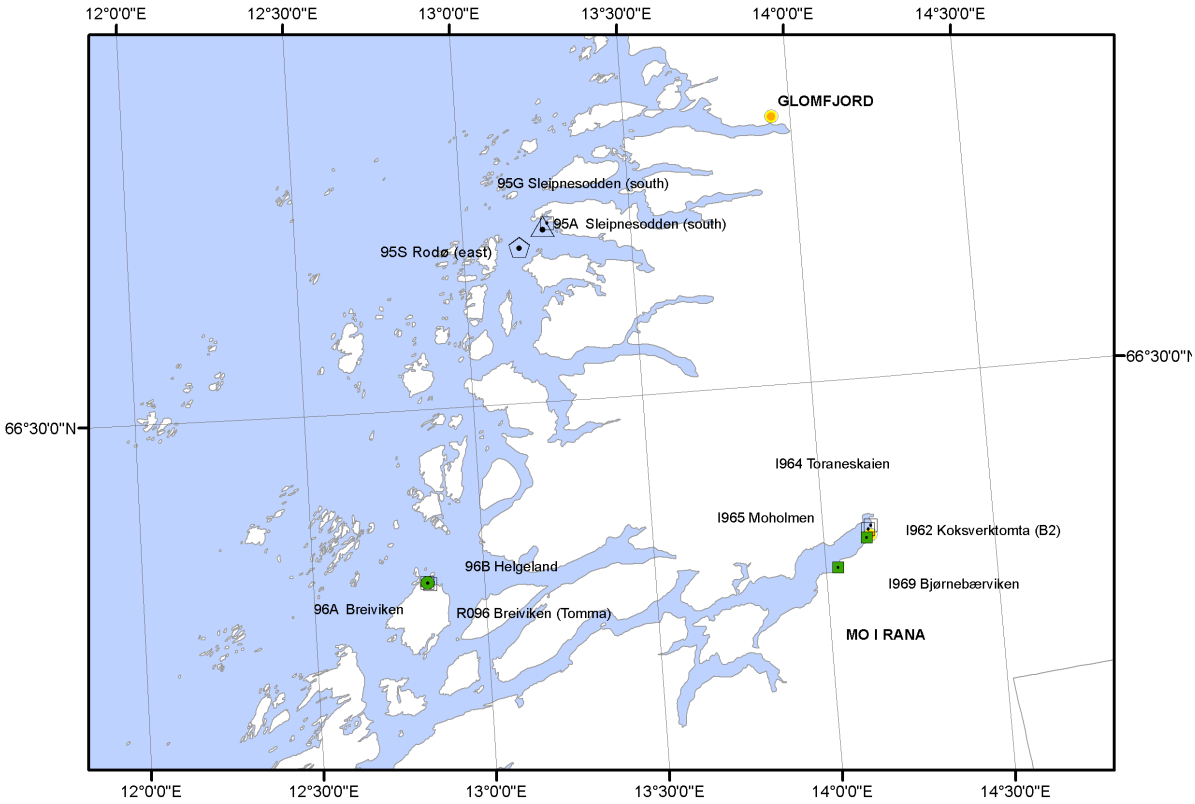
MAP 12


9°0'0"E
9°30'0"E
10°0'0"E
10°30'0"E
11°0'0"E
11°30'0"E
64°30'0"N
64°0'0"N
93A Sætervik
93S Raudøya (northeast)
93G Sætervika (Stadsvikskjæret)
92F Stokken area
92B Stokken area
92A1 Krokhölmen
92A2 Nygården
HARSVIK

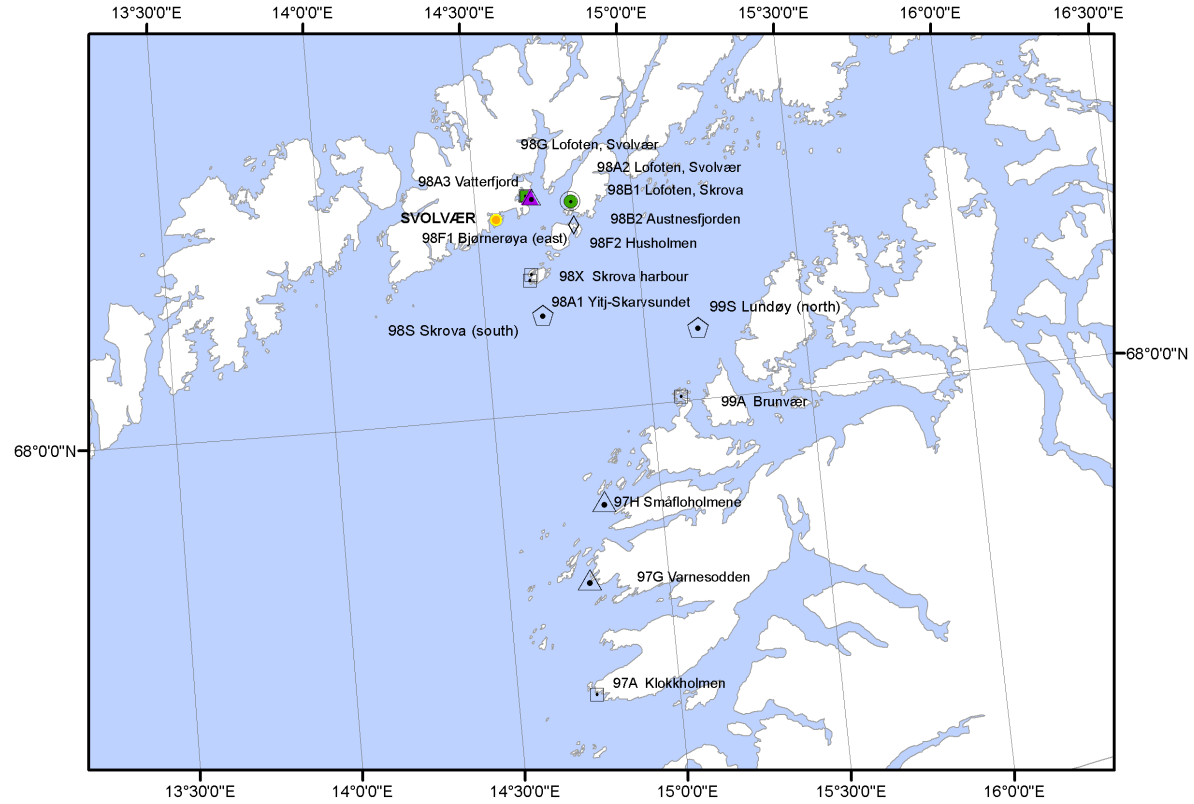
MAP 13


11°0'0"E
11°30'0"E
12°0'0"E
12°30'0"E
13°0'0"E
13°30'0"E
66°0'0"N
65°30'0"N
94G Steinskjær (Landfast)
94A Landfast
BRØNØYSUND

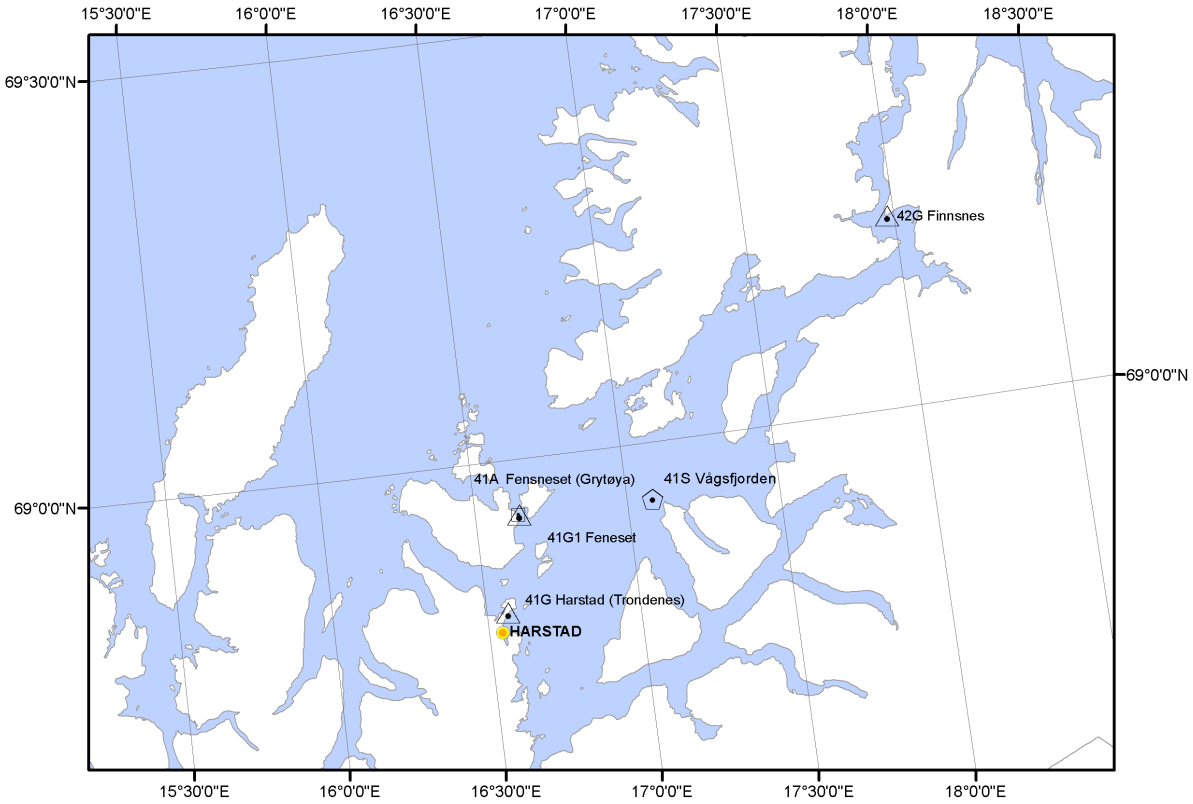
MAP 14


12°0'0"E
12°30'0"E
13°0'0"E
13°30'0"E
14°0'0"E
14°30'0"E
66°30'0"N
GLOMFJORD
95G Sleipnesodden (south)
95A Sleipnesodden (south)
95S Rodø (east)
I964 Toraneskaien
I965 Moholmen
I962 Koksverktomta (B2)
I969 Bjørnebærviken
96B Helgeland
96A Breiviken
R096 Breiviken (Tomma)
MO I RANA

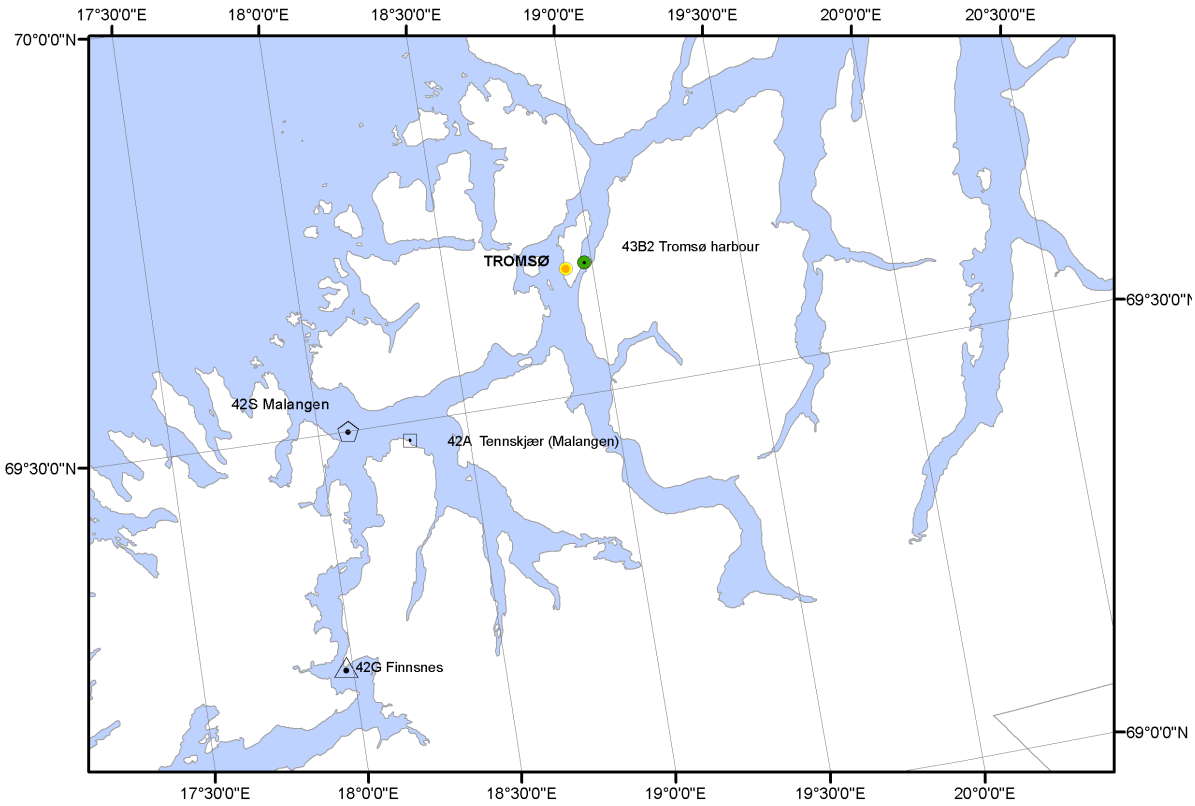
MAP 15


13°30'0"E
14°0'0"E
14°30'0"E
15°0'0"E
15°30'0"E
16°0'0"E
16°30'0"E
68°0'0"N
98G Lofoten, Svolvær
98A2 Lofoten, Svolvær
98A3 Vatterfjord
98B1 Lofoten, Skrova
SVOLVÆR
98B2 Austnesfjorden
98F1 Bjørnerøya (east)
98F2 Husholmen
98X Skrova harbour
98A1 Yitj-Skarvsundet
99S Lundøy (north)
98S Skrova (south)
99A Brunvær
97H Småfloholmene
97G Varnesodden
97A Klokkholmen

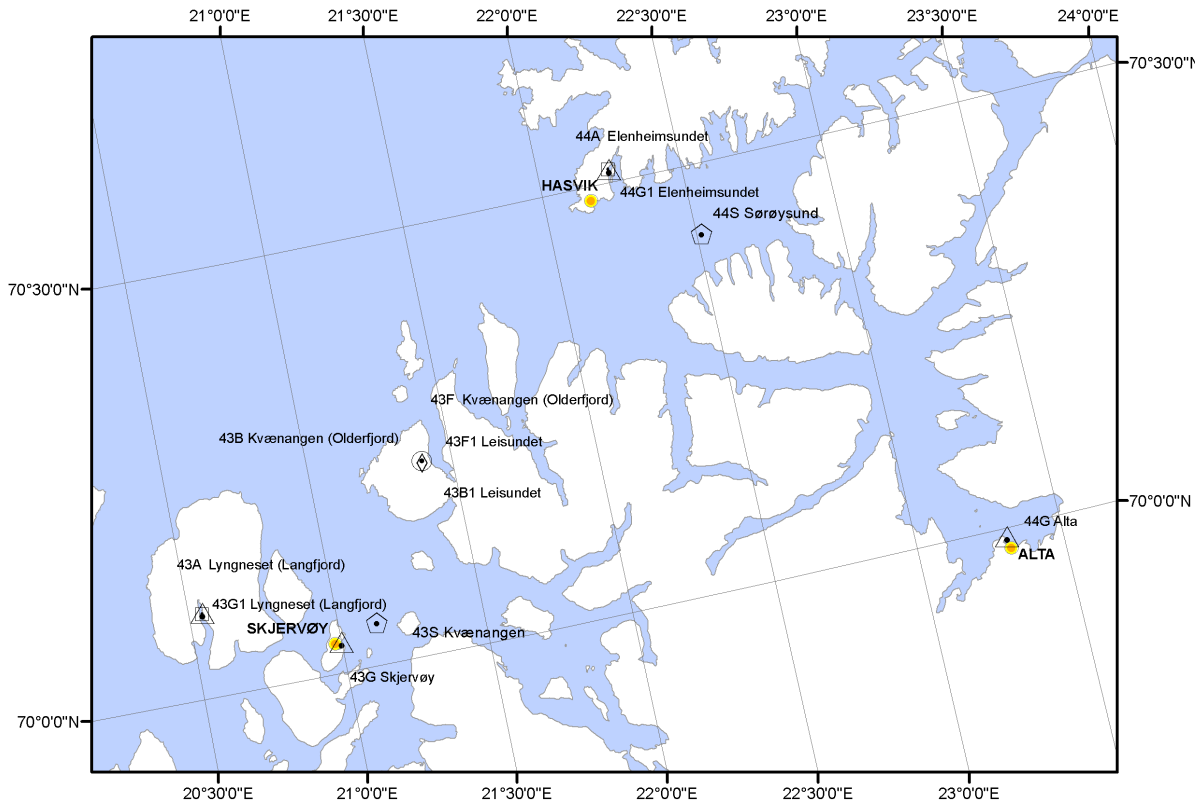
MAP 16


15°30'0"E
16°0'0"E
16°30'0"E
17°0'0"E
17°30'0"E
18°0'0"E
18°30'0"E
69°30'0"N
69°0'0"N
42G Finnsnes
41A Fensneset (Grytøya)
41S Vågsfjorden
41G1 Feneset
41G Harstad (Trondenes)
HARSTAD

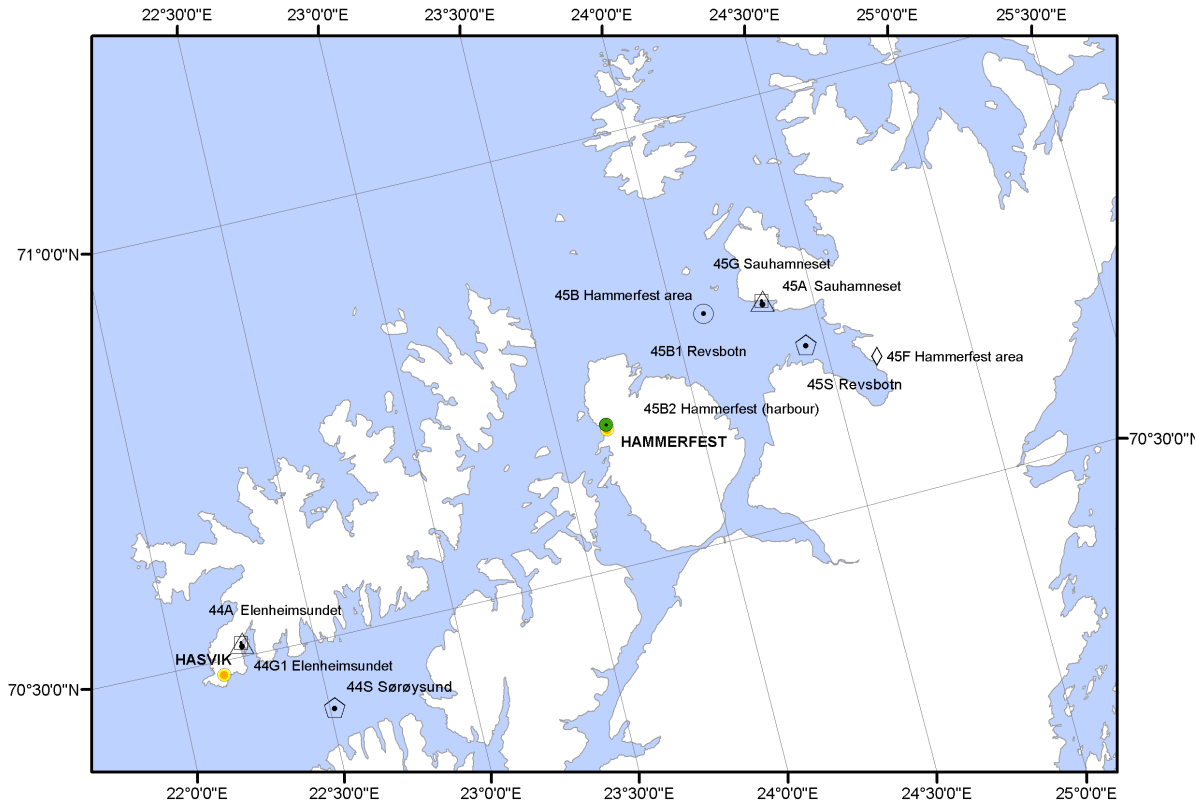
MAP 17


17°30'0"E
18°0'0"E
18°30'0"E
19°0'0"E
19°30'0"E
20°0'0"E
20°30'0"E
70°0'0"N
69°30'0"N
69°0'0"N
TROMSØ
43B2 Tromsø harbour
42S Malangen
42A Tennskjær (Malangen)
42G Finnsnes

MAP 18


21°0'0"E
21°30'0"E
22°0'0"E
22°30'0"E
23°0'0"E
23°30'0"E
24°0'0"E
70°30'0"N
70°0'0"N
20°30'0"E
44A Elenheimsundet
HASVIK
44G1 Elenheimsundet
44S Sørøysund
43F Kvænangen (Olderfjord)
43B Kvænangen (Olderfjord)
43F1 Leisundet
43B1 Leisundet
44G Alta
ALTA
43A Lyngneset (Langfjord)
43G1 Lyngneset (Langfjord)
SKJERVØY
43S Kvænangen
43G Skjervøy

MAP 19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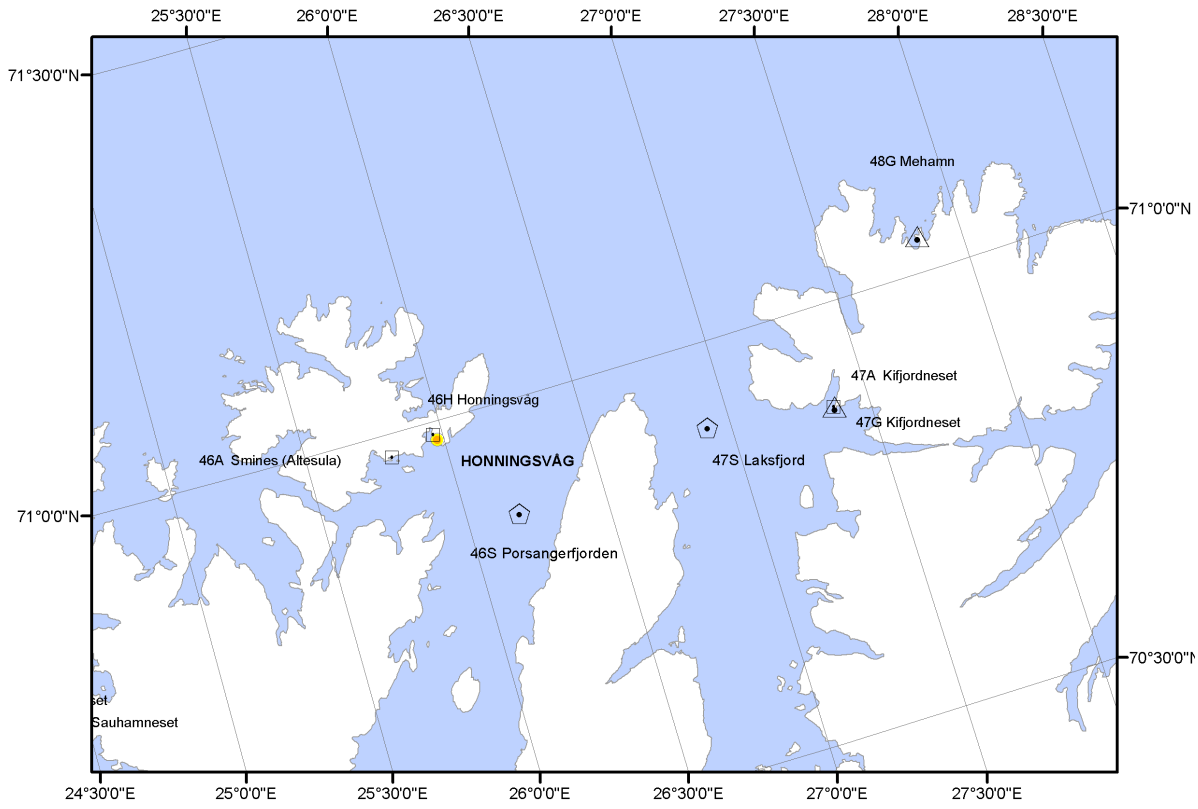
23°0'0"E
23°30'0"E
24°0'0"E
24°30'0"E
25°0'0"E
25°30'0"E
71°0'0"N
70°30'0"N
22°0'0"E
45G Sauhamneset
45A Sauhamneset
45B Hammerfest area
45B1 Revsbotn
45F Hammerfest area
45S Revsbotn
45B2 Hammerfest (harbour)
HAMMERFEST
44A Elenheimsundet
HASVIK
44G1 Elenheimsundet
44S Sørøysund

MAP 20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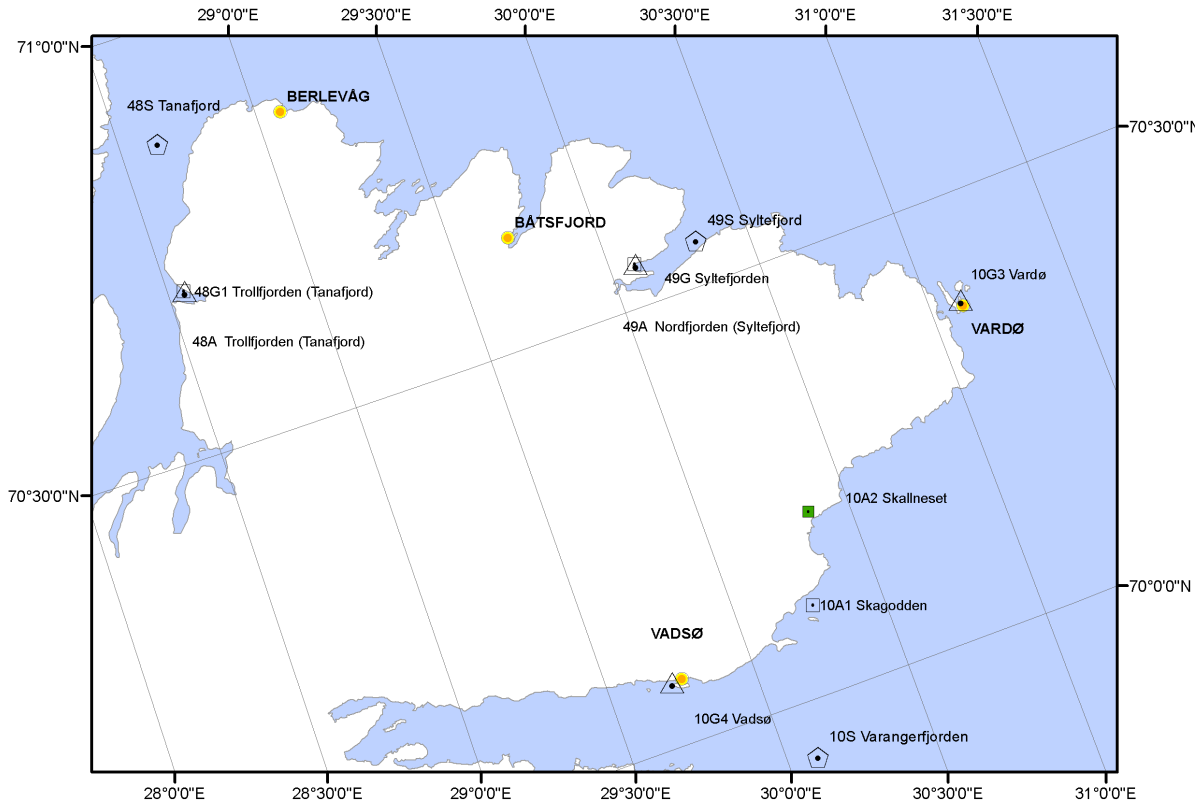
26°30'0"E
27°0'0"E
27°30'0"E
28°0'0"E
28°30'0"E
71°30'0"N
71°0'0"N
70°30'0"N
48G Mehamn
47A Kifjordneset
47G Kifjordneset
46H Honningsvåg
46A Smines (Altesula)
HONNINGSVÅG
47S Laksfjord
46S Porsangerfjorden
Sauhamneset
24°30'0"E
25°0'0"E
25°30'0"E
26°0'0"E
26°30'0"E
27°0'0"E
27°30'0"E

MAP 21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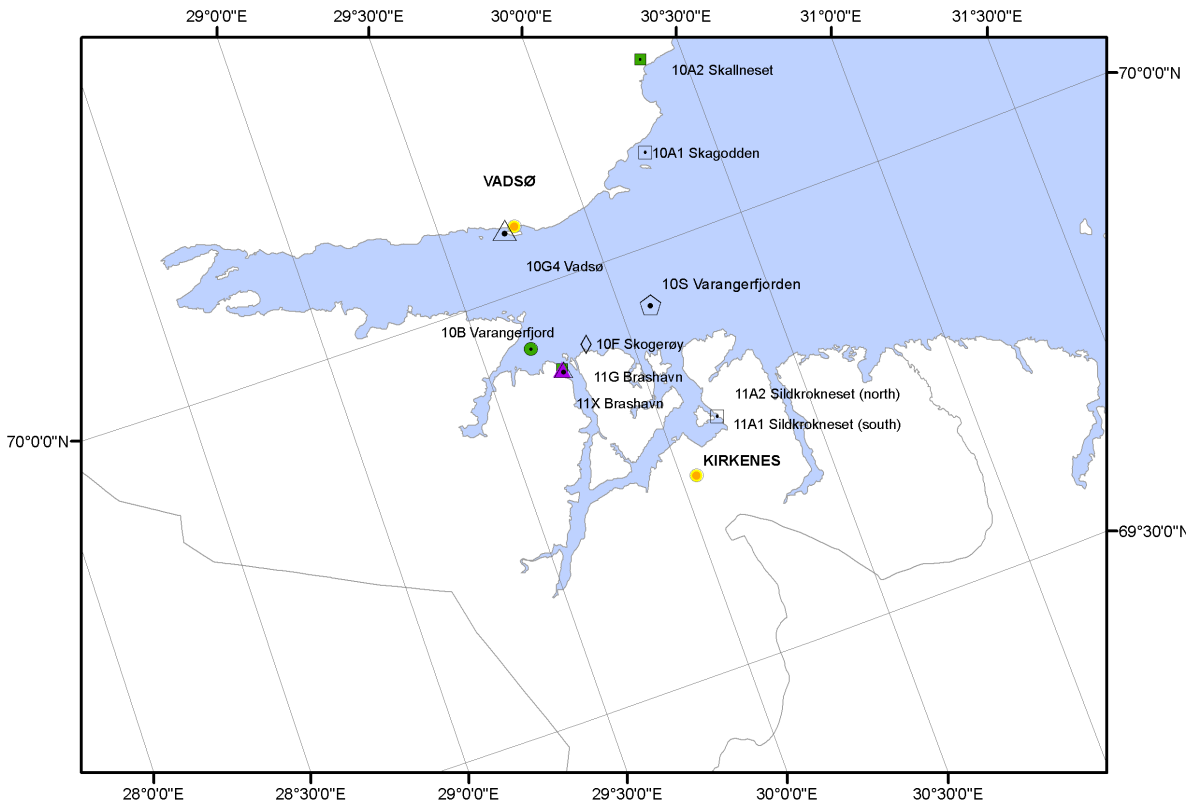
29°30'0"E
30°0'0"E
30°30'0"E
31°0'0"E
31°30'0"E
71°0'0"N
70°30'0"N
70°0'0"N
48S Tanafjord
BERLEVÅG
BÅTSFJORD
49S Syltefjord
49G Syltefjorden
10G3 Vardø
VARDØ
48G1 Trollfjorden (Tanafjord)
48A Trollfjorden (Tanafjord)
49A Nordfjorden (Syltefjord)
10A2 Skallneset
10A1 Skagodden
VADSØ
10G4 Vadsø
10S Varangerfjorden
28°0'0"E
28°30'0"E
29°0'0"E
29°30'0"E
30°0'0"E
30°30'0"E
31°0'0"E

MAP 22


29°0'0"E
29°30'0"E
30°0'0"E
30°30'0"E
31°0'0"E
31°30'0"E
70°0'0"N
69°30'0"N
28°0'0"E
28°30'0"E
29°0'0"E
29°30'0"E
30°0'0"E
30°30'0"E
10A2 Skallneset
10A1 Skagodden
VADSØ
10G4 Vadsø
10S Varangerfjorden
10B Varangerfjord
10F Skogerøy
11G Brashavn
11X Brashavn
11A2 Sildkrokneset (north)
11A1 Sildkrokneset (south)
KIRKENES

MAP 23

Appendix E
Overview of materials and analyses 2013-2014

Nominal station positions are shown on maps in Appendix D

Year:
2013t - samples taken in 2013
2014p – samples planned in 2014
2014t – samples taken in 2014

Species:
Atlantic cod (*Gadus morhua***)**
Blue Mussel (*Mytilus edulis***)**
Dog whelk (*Nucella lapillus***)**
Periwinkle (*Littorina littorea)*

Tissue:
SB-Soft body tissue
LI-Liver tissue, in fish
MU-Muscle tissue, in fish
BL-Blood, in fish
BI-Bile, fish

Red numbers indicate Supplementary investigations funded by the Ministry of Climate and Environment and these involved additional analyses on samples from blue mussel stations 30A, I301, I304, 31A, 36A1, 71A, I712, 51A, 56A, 65A, 22A, 10A2 and 11X; cod stations 30B, 36B, 15B, 53B, 23B, 98B1 and 10B; as well as all analyses for blue mussel stations: 52A, 57A, 63A, 69A, I133, I306, I307

Overview follows on next page

Parameter-group codes (see Appendix B for descriptions of codes) 2014:

code	Description	Me-SB	NI/LI-SB	Gm-BI	Gm-BL	Gm-LI	Gm-MU
I-MET	metals [1)]	x				x	
I-MET	Hg	x					x
ISOTO	$\delta^{15}N$ and $\delta^{13}C$	x					x
O-BR	PBDEs [2)]	x				x	x
OC-CB	PCBs [3)]	x				x	
OC-CL	HCB	x				x	x
OC-CP	SCCP, MCCP	x				x	
OC-DD	DDT, DDE, DDD	x				x	
OC-HC	α-, γ-HCH	x				x	
O-FL	PFAS [4)]					x	
O-PAH	PAHs [5)]	x				x	
O-MET	TBT [6)]	x	x				
O-FTA	Phthalates [7)]					x	
O-PHE	Phenols [8)]	x				x	x
PFRs	PFRs [9)]	x	x			x	x
PHC	PHCs [10)]	x	x			x	x
BE	Biological effects met. [11)]		Imposex	OH-pyrene	ALA-D	EROD-activity, CYP1A [12)]	

[1)] Cadmium (Cd), copper (Cu), lead (Pb), zinc (Zn), silver (Ag), arsenic (As), chrome (Cr), nickel (Ni), cobalt (Co) and tin (Sn).

[2)] Polybrominated diphenyl ethers (PBDEs), including brominated flame retardants and includes a selection of: BDE28, BDE47, BDE49, BDE66, BDE71, BDE77, BDE85, BDE99, BDE100, BDE119, BDE138, BDE153, BDE154, BDE183, BDE205, HBCD.

[3)] Includes a selection of the congeners: CB-28,-52,-101,-105,-118,-138,-153,-156,-180, 209, 5-CB, OCS and, when dioxins are analysed, the non-orto-PCBs, i.e. CB-77, -81, -126, -169.

[4)] Includes: PFNA, PFOA, PFHpA, PFHxA, PFOS, PFBS, PFOSA.

[5)] Includes (with NPDs): ACNE, ACNLE, ANT, BAP, BBJF, BEP, BGHIP, BKF. BAA. CHR, DBA3A, DBT, DBTC1, DBTC2, DBTC3, FLE, FLU, ICDP, NAP, NAPC1, NAPC2, NAPC3, PA, PAC1, PAC2, PAC3, PER, PYR.

[6)] Includes: DBTIN, DPTIN, MBTIN, MPTIN, TBTIN, TPTIN.

[7)] O-FTA Phthalates, includes: BBP, DBPA, DEHA, DEHP, DEP, DEPA, DIBP, DIDP, DIHP, DINCH, DIPA, DMP, DNOP, DPF.

[8)] O-PHE phenols (octa non), includes: 4-n-NP, 4-n-OP, 4-t-NP, 4-t-OP.

[9)] PFRs – Phosphorus Flame Retardants and includes a selection of: TIBP, TBP, TCEP, TCPP, TDCP, TBEP, TPhP, EHDPP, V6, DBPhP, BdPhP, TEHP, ToCrP, TCrP.

[10)] PHC – phenols including BPA, TBBPA.

[11)] Biological effects methods.

[12)] Cod only.

Appendix E. Sampling and analyses for 2013-2014 -biota.

order	Year	Station	Station name	Latitude	Longitude	Species	Tissue	I-MET	O-MET	O-BR	OC-CB	OC-CL	OC-CP	OC-DD	OC-HC	O-FL	O-FTA	O-PAH	PFR	PHC	O-PHE	Triclosan	Diuron & Irgarol	BE	ISOTO
1	2013t	30B	Oslo City area	59.81667	10.55000	GADU MOR	BI																	14	
1	2014p	30B	Oslo City area	59.81667	10.55000	GADU MOR	BI																	15	
1	2014t	30B	Oslo City area	59.81667	10.55000	GADU MOR	BI																	15	
6	2013t	15B	Ullerø area	58.05000	6.71667	GADU MOR	BI																	14	
6	2014p	15B	Ullerø area	58.05000	6.71667	GADU MOR	BI																	15	
6	2014t	15B	Ullerø area	58.05000	6.71667	GADU MOR	BI																	15	
7	2013t	53B	Inner Sørfjord	60.16667	6.56667	GADU MOR	BI																	15	
7	2014p	53B	Inner Sørfjord	60.16667	6.56667	GADU MOR	BI																	15	
7	2014t	53B	Inner Sørfjord	60.16667	6.56667	GADU MOR	BI																	15	
8	2013t	23B	Karihavet area	59.90000	5.13333	GADU MOR	BI																	15	
8	2014p	23B	Karihavet area	59.90000	5.13333	GADU MOR	BI																	15	
8	2014t	23B	Karihavet area	59.90000	5.13333	GADU MOR	BI																	15	
1	2013t	30B	Oslo City area	59.81667	10.55000	GADU MOR	BL																	15	
1	2014p	30B	Oslo City area	59.81667	10.55000	GADU MOR	BL																	15	
1	2014t	30B	Oslo City area	59.81667	10.55000	GADU MOR	BL																	15	
7	2013t	53B	Inner Sørfjord	60.16667	6.56667	GADU MOR	BL																	15	
7	2014p	53B	Inner Sørfjord	60.16667	6.56667	GADU MOR	BL																	15	
7	2014t	53B	Inner Sørfjord	60.16667	6.56667	GADU MOR	BL																	15	
8	2013t	23B	Karihavet area	59.90000	5.13333	GADU MOR	BL																	15	
8	2014p	23B	Karihavet area	59.90000	5.13333	GADU MOR	BL																	15	
8	2014t	23B	Karihavet area	59.90000	5.13333	GADU MOR	BL																	15	
1	2013t	30B	Oslo City area	59.81667	10.55000	GADU MOR	LI	16		18	16	16	10	16	16	16	6		10	10	12	6	6	15	
1	2014p	30B	Oslo City area	59.81667	10.55000	GADU MOR	LI	15		15	15	15	15	15		15			15	15	15			15	
1	2014t	30B	Oslo City area	59.81667	10.55000	GADU MOR	LI	15		15	15	15	15	15		15			15	15	15			15	
2	2013t	36B	Færder area	59.04050	10.43583	GADU MOR	LI	10		10	10	10	3	10	10	10			3	3	3				
2	2014p	36B	Færder area	59.04050	10.43583	GADU MOR	LI	15		15	15	15	15	15		15			15	15	15				
2	2014t	36B	Færder area	59.04050	10.43583	GADU MOR	LI	15		15	15	15	15	15		15			15	15	15				
3	2013t	02B	Hvalerbassenget, Kirkøy nord	59.11250	11.03883	GADU MOR	LI	4		2	4		2						2	2	2				
3	2014p	02B	Hvalerbassenget, Kirkøy nord	59.11250	11.03883	GADU MOR	LI	15		15	15		15						15	15	15				
3	2014t	02B	Hvalerbassenget, Kirkøy nord	59.11250	11.03883	GADU MOR	LI	3		3	3		3						3	3	3				
4	2013t	71B	Grenlandsfjord, Brevik area	59.06117	9.70967	GADU MOR	LI	15		9			9						9	9	9				
4	2014p	71B	Grenlandsfjord, Brevik area	59.06117	9.70967	GADU MOR	LI	15		15			15						15	15	15				
4	2014t	71B	Grenlandsfjord, Brevik area	59.06117	9.70967	GADU MOR	LI	13		13			13						13	13	13				
5	2013t	13B	Kristiansand (harbour)	58.13283	7.98850	GADU MOR	LI	10		10	10		6			10	5		6	6	6	7	5		
5	2014p	13B	Kristiansand (harbour)	58.13283	7.98850	GADU MOR	LI	15		15	15		15			15			15	15	15				
5	2014t	13B	Kristiansand (harbour)	58.13283	7.98850	GADU MOR	LI	14		14	14		14			14			14	14	14				
6	2013t	15B	Ullerø area	58.05000	6.71667	GADU MOR	LI	15			15	15		15	15										
6	2014p	15B	Ullerø area	58.05000	6.71667	GADU MOR	LI	15			15	15		15											
6	2014t	15B	Ullerø area	58.05000	6.71667	GADU MOR	LI	15			15	15		15											
7	2013t	53B	Inner Sørfjord	60.16667	6.56667	GADU MOR	LI			6	6	6	6	6	6	6			6	5	6			15	
7	2014p	53B	Inner Sørfjord	60.16667	6.56667	GADU MOR	LI			15	15	15	15	15		15			15	15	15			15	
7	2014t	53B	Inner Sørfjord	60.16667	6.56667	GADU MOR	LI			9	9	9	9	17		9			7	9	7			9	
8	2013t	23B	Karihavet area	59.90000	5.13333	GADU MOR	LI	16		19	16	16	9	16	16	16			9	8	9			15	
8	2014p	23B	Karihavet area	59.90000	5.13333	GADU MOR	LI	15		15	15	15	15	15		15			15	15	15			15	
8	2014t	23B	Karihavet area	59.90000	5.13333	GADU MOR	LI	14		14	14	14	14	14		14			14	14	14			14	
9	2013t	28B	Ålesund, Hundsvær area	62.25167	5.86400	GADU MOR	LI	6		6	6		4				5		4	4	4	5	5		

order	Year	Station	Station name	Latitude	Longitude	Species	Tissue	I-MET	O-MET	O-BR	OC-CB	OC-CL	OC-CP	OC-DD	OC-HC	O-FL	O-FTA	O-PAH	PFR	PHC	O-PHE	Triclosan	Diuron & Irgarol	BE	ISOTO
9	2014p	28B	Ålesund, Hundsvær area	62.25167	5.86400	GADU MOR	LI	15		15	15		15						15	15	15				
9	2014t	28B	Ålesund, Hundsvær area	62.25167	5.86400	GADU MOR	LI	0		0	0		0						0	0	0				
10	2013t	80B	Inner Trondheimsfjord	63.45733	10.44950	GADU MOR	LI	15		15	15		15			15			15	14	15				
10	2014p	80B	Inner Trondheimsfjord	63.45733	10.44950	GADU MOR	LI	15		15	15		15			15			15	15	15				
10	2014t	80B	Inner Trondheimsfjord	63.45733	10.44950	GADU MOR	LI	15		15	15		15			15			15	15	15				
11	2013t	96B	Helgeland coast, Sandnessjøen area	66.29617	12.83367	GADU MOR	LI	15			15														
11	2014p	96B	Helgeland coast, Sandnessjøen area	66.29617	12.83367	GADU MOR	LI	15			15														
11	2014t	96B	Helgeland coast, Sandnessjøen area	66.29617	12.83367	GADU MOR	LI	15			15														
12	2013t	98B1	Lofoten, Skrova	68.24667	14.80333	GADU MOR	LI	15		16	15	15	3	15	15	15			1						
12	2014p	98B1	Lofoten, Skrova	68.24667	14.80333	GADU MOR	LI	15		15	15	15	15	15		15			15						
12	2014t	98B1	Lofoten, Skrova	68.24667	14.80333	GADU MOR	LI	8		8	8	8	8	8		8			8						
13	2013t	43B2	Tromsø harbour	70.30200	21.42683	GADU MOR	LI	15		15	15		15			15	9		15	15	15	9	9		
13	2014p	43B2	Tromsø harbour	70.30200	21.42683	GADU MOR	LI	15		15	15		15			15			15	15	15				
13	2014t	43B2	Tromsø harbour	70.30200	21.42683	GADU MOR	LI	15		15	15		15			15			15	15	15				
14	2013t	45B2	Hammerfest harbour	70.70000	24.48333	GADU MOR	LI	0			0														
14	2014p	45B2	Hammerfest harbour	70.70000	24.48333	GADU MOR	LI	15			15														
14	2014t	45B2	Hammerfest harbour	70.70000	24.48333	GADU MOR	LI	15			15														
15	2013t	10B	Varangerfjord	69.93333	29.66667	GADU MOR	LI	13			13	13		13	13										
15	2014p	10B	Varangerfjord	69.93333	29.66667	GADU MOR	LI	15			15	15		15										15	
15	2014t	10B	Varangerfjord	69.93333	29.66667	GADU MOR	LI	13			13	15		15										15	
1	2013t	30B	Oslo City area	59.81667	10.55000	GADU MOR	MU	15																	15
1	2014p	30B	Oslo City area	59.81667	10.55000	GADU MOR	MU	15																	15
1	2014t	30B	Oslo City area	59.81667	10.55000	GADU MOR	MU	15																	15
2	2013t	36B	Færder area	59.04050	10.43583	GADU MOR	MU	15																	15
2	2014p	36B	Færder area	59.04050	10.43583	GADU MOR	MU	15																	15
2	2014t	36B	Færder area	59.04050	10.43583	GADU MOR	MU	15																	15
3	2013t	02B	Hvalerbassenget, Kirkøy nord	59.11250	11.03883	GADU MOR	MU	18																	18
3	2014p	02B	Hvalerbassenget, Kirkøy nord	59.11250	11.03883	GADU MOR	MU	15																	15
3	2014t	02B	Hvalerbassenget, Kirkøy nord	59.11250	11.03883	GADU MOR	MU	8																	8
4	2013t	71B	Grenlandsfjord, Brevik area	59.06117	9.70967	GADU MOR	MU	15																	15
4	2014p	71B	Grenlandsfjord, Brevik area	59.06117	9.70967	GADU MOR	MU	15																	15
4	2014t	71B	Grenlandsfjord, Brevik area	59.06117	9.70967	GADU MOR	MU	15																	15
5	2013t	13B	Kristiansand (harbour)	58.13283	7.98850	GADU MOR	MU	15																	15
5	2014p	13B	Kristiansand (harbour)	58.13283	7.98850	GADU MOR	MU	15																	15
5	2014t	13B	Kristiansand (harbour)	58.13283	7.98850	GADU MOR	MU	15																	15
6	2013t	15B	Ullerø area	58.05000	6.71667	GADU MOR	MU	15																	15
6	2014p	15B	Ullerø area	58.05000	6.71667	GADU MOR	MU	15																	15
6	2014t	15B	Ullerø area	58.05000	6.71667	GADU MOR	MU	15																	15
7	2013t	53B	Inner Sørfjord	60.16667	6.56667	GADU MOR	MU	15																	13
7	2014p	53B	Inner Sørfjord	60.16667	6.56667	GADU MOR	MU	15																	15
7	2014t	53B	Inner Sørfjord	60.16667	6.56667	GADU MOR	MU	15																	15
8	2013t	23B	Karihavet area	59.90000	5.13333	GADU MOR	MU	18																	18
8	2014p	23B	Karihavet area	59.90000	5.13333	GADU MOR	MU	15																	15
8	2014t	23B	Karihavet area	59.90000	5.13333	GADU MOR	MU	15																	15
9	2013t	28B	Ålesund, Hundsvær area	62.25167	5.86400	GADU MOR	MU	7																	7
9	2014p	28B	Ålesund, Hundsvær area	62.25167	5.86400	GADU MOR	MU	15																	15
9	2014t	28B	Ålesund, Hundsvær area	62.25167	5.86400	GADU MOR	MU	0																	15
10	2013t	80B	Trondheim (harbour)	63.45733	10.44950	GADU MOR	MU	15																	15
10	2014p	80B	Trondheim (harbour)	63.45733	10.44950	GADU MOR	MU	15																	15
10	2014t	80B	Trondheim (harbour)	63.45733	10.44950	GADU MOR	MU	15																	15
11	2013t	96B	Helgeland coast, Sandnessjøen area	66.29617	12.83367	GADU MOR	MU	15																	15
11	2014p	96B	Helgeland coast, Sandnessjøen area	66.29617	12.83367	GADU MOR	MU	15																	15

order	Year	Station	Station name	Latitude	Longitude	Species	Tissue	I-MET	O-MET	O-BR	OC-CB	OC-CL	OC-CP	OC-DD	OC-HC	O-FL	O-FTA	O-PAH	PFR	PHC	O-PHE	Triclosan	Diuron & Irgarol	BE	ISOTO
11	2014t	96B	Helgeland coast, Sandnessjøen area	66.29617	12.83367	GADU MOR	MU	15																	15
12	2013t	98B1	Bjørnerøya (east)	68.24667	14.80333	GADU MOR	MU	15																	15
12	2014p	98B1	Bjørnerøya (east)	68.24667	14.80333	GADU MOR	MU	15																	15
12	2014t	98B1	Bjørnerøya (east)	68.24667	14.80333	GADU MOR	MU	15																	15
13	2013t	43B2	Trømsø (harbour)	70.30200	21.42683	GADU MOR	MU	15																	15
13	2014p	43B2	Trømsø (harbour)	70.30200	21.42683	GADU MOR	MU	15																	15
13	2014t	43B2	Trømsø (harbour)	70.30200	21.42683	GADU MOR	MU	15																	15
14	2013t	45B2	Hammerfest (harbour)	70.70000	24.48333	GADU MOR	MU	15																	0
14	2014p	45B2	Hammerfest (harbour)	70.70000	24.48333	GADU MOR	MU	15																	15
14	2014t	45B2	Hammerfest (harbour)	70.70000	24.48333	GADU MOR	MU	15																	15
15	2013t	10B	Varangerfjord	69.93333	29.66667	GADU MOR	MU	15																	15
15	2014p	10B	Varangerfjord	69.93333	29.66667	GADU MOR	MU	15																	15
15	2014t	10B	Varangerfjord	69.93333	29.66667	GADU MOR	MU	15																	15
2	2013t	71G	Fugløyskjær	58.98250	9.80833	LITT LIT	SB		1										1	1				1	
2	2014p	71G	Fugløyskjær	58.98250	9.80833	LITT LIT	SB		1										1	1				1	
2	2014t	71G	Fugløyskjær	58.98250	9.80833	LITT LIT	SB		1										1	1				1	3
1	2013t	I301	Akershuskaia	59.90533	10.73633	MYTI EDU	SB	3	3	2	3	3		3	3		3	3	3	3	2	2	2		
1	2014p	I301	Akershuskaia	59.90533	10.73633	MYTI EDU	SB	3	3		3	3		3				3	3	3					
1	2014t	I301	Akershuskaia	59.90533	10.73633	MYTI EDU	SB	3	3		3	3		3				3	3	3					
2	2013t	30A	Gressholmen	59.88667	10.80967	MYTI EDU	SB	3	3	3	3	3	3	3	3			3	3	3	3				3
2	2014p	30A	Gressholmen	59.88667	10.80967	MYTI EDU	SB	3	3	3	3	3	3	3				3	3	3	3				3
2	2014t	30A	Gressholmen	59.88667	10.80967	MYTI EDU	SB	3	3	3	3	3	3	3				3	3	3	3				3
3	2013t	I304	Gåsøya	59.85133	10.58900	MYTI EDU	SB	3	3	2	3	3		3	3		3	3	3	3	2		2		3
3	2014p	I304	Gåsøya	59.85133	10.58900	MYTI EDU	SB	3	3		3	3		3				3	3	3					3
3	2014t	I304	Gåsøya	59.85133	10.58900	MYTI EDU	SB	3	3		3	3		3				3	3	3				1	3
4	2013t	I306	Håøya	59.71333	10.55517	MYTI EDU	SB	3			3														
4	2014p	I306	Håøya	59.71333	10.55517	MYTI EDU	SB	3			3														
4	2014t	I306	Håøya	59.71333	10.55517	MYTI EDU	SB	3			3													1	
5	2013t	I307	Ramtonholmen	59.74450	10.52283	MYTI EDU	SB	3			3														
5	2014p	I307	Ramtonholmen	59.74450	10.52283	MYTI EDU	SB	3			3														
5	2014t	I307	Ramtonholmen	59.74450	10.52283	MYTI EDU	SB	3			3													1	
6	2013t	31A	Solbergstrand	59.61500	10.65667	MYTI EDU	SB	3	3	2	3	3		3	3				3	3	2		1		
6	2014p	31A	Solbergstrand	59.61500	10.65667	MYTI EDU	SB	3	3		3	3		3					3	3					
6	2014t	31A	Solbergstrand	59.61500	10.65667	MYTI EDU	SB	3	3		3	3		3					3	3					
7	2013t	35A	Mølen	59.48817	10.49800	MYTI EDU	SB	3		3	3										2		3		
7	2014p	35A	Mølen	59.48817	10.49800	MYTI EDU	SB	3	1		2							1	1	1					
7	2014t	35A	Mølen	59.48817	10.49800	MYTI EDU	SB	3	1		2							1	1	1					
8	2013t	36A1	Tjøme	59.07357	10.42522	MYTI EDU	SB	3	3	3	3	3	3	3						3	3				
8	2014p	36A1	Tjøme	59.07357	10.42522	MYTI EDU	SB	3	3	3	3		3	3					3	3	3				
8	2014t	36A1	Tjøme	59.07357	10.42522	MYTI EDU	SB	3	3	3	3		3	3					3	3	3				
9	2013t	I023	Singlekalven (south)	59.09500	11.13667	MYTI EDU	SB	3		3	3		3					3	3	3	3				3
9	2014p	I023	Singlekalven (south)	59.09500	11.13667	MYTI EDU	SB	3		3	3		3					3	3	3	3				3
9	2014t	I023	Singlekalven (south)	59.09500	11.13667	MYTI EDU	SB	3		3	3		3					3	3	3	3				3
10	2013t	I024	Kirkøy (north west)	59.08000	10.98633	MYTI EDU	SB	1																	
10	2014p	I024	Kirkøy (north west)	59.08000	10.98633	MYTI EDU	SB	3			3														
10	2014t	I024	Kirkøy (north west)	59.08000	10.98633	MYTI EDU	SB	3			3														
11	2013t	71A	Bjørkøya (Risøyodden)	59.02333	9.75367	MYTI EDU	SB	3		3	3	3	3	3	3			3	3	3	3				3
11	2014p	71A	Bjørkøya (Risøyodden)	59.02333	9.75367	MYTI EDU	SB	3		3		3	3	3				3	3	3	3				3
11	2014t	71A	Bjørkøya (Risøyodden)	59.02333	9.75367	MYTI EDU	SB	3		3		3	3	3				3	3	3	3				3
12	2013t	I712	Croftholmen	59.04533	9.70683	MYTI EDU	SB	2		2	2		2					2	2	2	2				2
12	2014p	I712	Croftholmen	59.04533	9.70683	MYTI EDU	SB	3		3		3	3	3				3	3	3	3				3
12	2014t	I712	Croftholmen	59.04533	9.70683	MYTI EDU	SB	1		1		1	1	1				1	1	1	1			1	1

order	Year	Station	Station name	Latitude	Longitude	Species	Tissue	I-MET	O-MET	O-BR	OC-CB	OC-CL	OC-CP	OC-DD	OC-HC	O-FL	O-FTA	O-PAH	PFR	PHC	O-PHE	Triclosan	Diuron & Irgarol	BE	ISOTO
13	2013t	76A2	Risøy	58.73270	9.28104	MYTI EDU	SB	3			3	3		3											
13	2014p	76A2	Risøy	58.73270	9.28104	MYTI EDU	SB	3			3	3		3											
13	2014t	76A2	Risøy	58.73270	9.28104	MYTI EDU	SB	3			3	3		3											
14	2013t	I131A	Lastad	58.05550	7.70867	MYTI EDU	SB	3										3							
14	2014p	I131A	Lastad	58.05550	7.70867	MYTI EDU	SB	3										3							
14	2014t	I131A	Lastad	58.05550	7.70867	MYTI EDU	SB	3										3							
15	2013t	I133	Odderøy	58.13167	8.00167	MYTI EDU	SB	3	3		3	3		3	3				3	3					
15	2014p	I133	Odderøy	58.13167	8.00167	MYTI EDU	SB	3	3		3	3		3					3	3					
15	2014t	I133	Odderøy	58.13167	8.00167	MYTI EDU	SB	3	3		3	3		3					3	3				1	
16	2013t	15A	Gåsøy (Ullerø)	58.05117	6.88600	MYTI EDU	SB	3			3														3
16	2014p	15A	Gåsøy (Ullerø)	58.05117	6.88600	MYTI EDU	SB	3			3														3
16	2014t	15A	Gåsøy (Ullerø)	58.05117	6.88600	MYTI EDU	SB	3			3														3
17	2013t	51A	Byrkjenes	60.08500	6.55167	MYTI EDU	SB	3			3	3		3	3		3					3	3		3
17	2014p	51A	Byrkjenes	60.08500	6.55167	MYTI EDU	SB	3			3	3		3											3
17	2014t	51A	Byrkjenes	60.08500	6.55167	MYTI EDU	SB	3			3	3		3											3
18	2013t	52A	Eitrheimsneset	60.09667	6.53667	MYTI EDU	SB	3			3	3		3	3										
18	2014p	52A	Eitrheimsneset	60.09667	6.53667	MYTI EDU	SB	3			3	3		3											
18	2014t	52A	Eitrheimsneset	60.09667	6.53667	MYTI EDU	SB	3			3	3		3											
19	2013t	56A	Kvalnes	60.25517	6.62000	MYTI EDU	SB	3			3	3		3	3		3					3	3		3
19	2014p	56A	Kvalnes	60.25517	6.62000	MYTI EDU	SB	3			3	3		3											3
19	2014t	56A	Kvalnes	60.25517	6.62000	MYTI EDU	SB	3			3	3		3											3
20	2013t	57A	Krossanes	60.42083	6.74217	MYTI EDU	SB	3			3	3		3	3										
20	2014p	57A	Krossanes	60.42083	6.74217	MYTI EDU	SB	3			3	3		3											
20	2014t	57A	Krossanes	60.42083	6.74217	MYTI EDU	SB	3			3	3		3											
21	2013t	63A	Ranaskjær	60.41833	6.40833	MYTI EDU	SB	3			3	3		3	3										
21	2014p	63A	Ranaskjær	60.41833	6.40833	MYTI EDU	SB	3			3	3		3											
21	2014t	63A	Ranaskjær	60.41833	6.40833	MYTI EDU	SB	3			3	3		3											
22	2013t	64A	Utne, Outer Sørfjord	60.42367	6.62217	MYTI EDU	SB	3			3			3											
22	2014p	64A	Utne, Outer Sørfjord	60.42367	6.62217	MYTI EDU	SB	3			3			3											
22	2014t	64A	Utne, Outer Sørfjord	60.42367	6.62217	MYTI EDU	SB	3			3			3											
23	2013t	65A	Vikingneset	60.24167	6.16000	MYTI EDU	SB	3			3	3		3	3										
23	2014p	65A	Vikingneset	60.24167	6.16000	MYTI EDU	SB	3			3	3		3											
23	2014t	65A	Vikingneset	60.24167	6.16000	MYTI EDU	SB	3			3	3		3											
24	2013t	69A	Lille Terøy	59.97983	5.75583	MYTI EDU	SB	3			3														
24	2014p	69A	Lille Terøy	59.97983	5.75583	MYTI EDU	SB	3			3														
24	2014t	69A	Lille Terøy	59.97983	5.75583	MYTI EDU	SB	3			3														
25	2013t	22A	Espevær (west)	59.58667	5.14167	MYTI EDU	SB	3	3		3	3		3	3				3	3					3
25	2014p	22A	Espevær (west)	59.58667	5.14167	MYTI EDU	SB	3	3		3	3		3					3	3					3
25	2014t	22A	Espevær (west)	59.58667	5.14167	MYTI EDU	SB	3	3		3	3		3					3	3					3
26	2013t	I241	Nordnes	60.40067	5.30167	MYTI EDU	SB	2		2	2		2						2	2	2				
26	2014p	I241	Nordnes	60.40067	5.30167	MYTI EDU	SB	3		3	3		3						3	3	3				
26	2014t	I241	Nordnes	60.40067	5.30167	MYTI EDU	SB	2		1	2		1						1	1	1			1	
27	2013t	26A2	Måløy	61.94050	5.12300	MYTI EDU	SB	3		3	3		3						3	3	3				3
27	2014p	26A2	Måløy	61.94050	5.12300	MYTI EDU	SB	3		3	3		3						3	3	3				3
27	2014t	26A2	Måløy	61.94050	5.12300	MYTI EDU	SB	3		3	3		3						3	3	3				3
28	2013t	91A2	Ørland, Outer Trondheimsfjord	63.68750	9.66783	MYTI EDU	SB	3		3	3		3						3	3	3				3
28	2014p	91A2	Ørland, Outer Trondheimsfjord	63.68750	9.66783	MYTI EDU	SB	3		3	3		3						3	3	3				3
28	2014t	91A2	Ørland, Outer Trondheimsfjord	63.68750	9.66783	MYTI EDU	SB	3		3	3		3						3	3	3				3
29	2013t	I965	Moholmen	66.31200	14.12583	MYTI EDU	SB	3										3							
29	2014p	I965	Moholmen	66.31200	14.12583	MYTI EDU	SB	3										3							
29	2014t	I965	Moholmen	66.31200	14.12583	MYTI EDU	SB	3										3						1	
31	2013t	97A2	Bodø harbour	67.29500	14.38800	MYTI EDU	SB	3		3	3		3						3	3	3				3

order	Year	Station	Station name	Latitude	Longitude	Species	Tissue	I-MET	O-MET	O-BR	OC-CB	OC-CL	OC-CP	OC-DD	OC-HC	O-FL	O-FTA	O-PAH	PFR	PHC	O-PHE	Triclosan	Diuron & Irgarol	BE	ISOTO
31	2014p	97A2	Bodø harbour	67.29500	14.38800	MYTI EDU	SB	3		3	3		3						3	3	3				3
31	2014t	97A2	Bodø harbour	67.29500	14.38800	MYTI EDU	SB	3		3	3		3						3	3	3				3
32	2013t	98A2	Husvaagen area	68.25767	14.66383	MYTI EDU	SB	3		3	3		3					3	3	3	3				3
32	2014p	98A2	Husvaagen area	68.25767	14.66383	MYTI EDU	SB	3		3	3		3					3	3	3	3				3
32	2014t	98A2	Husvaagen area	68.25767	14.66383	MYTI EDU	SB	3		3	3		3					3	3	3	3				3
33	2013t	10A2	Skallneset	70.20833	30.35833	MYTI EDU	SB	3			3	3		3	3										
33	2014p	10A2	Skallneset	70.20833	30.35833	MYTI EDU	SB	3			3	3		3											
33	2014t	10A2	Skallneset	70.20833	30.35833	MYTI EDU	SB	3			3	3		3											
34	2013t	11X	Brashavn	69.89867	29.74417	MYTI EDU	SB	3			3	3		3	3										3
34	2014p	11X	Brashavn	69.89867	29.74417	MYTI EDU	SB	3			3	3		3											3
34	2014t	11X	Brashavn	69.89867	29.74417	MYTI EDU	SB	3			3	3		3											3
#N/A	2013t	I969	Bjørnbærviken (B9)	66.27983	14.03550	MYTI EDU	SB	3										3							
#N/A	2014p	I969	Bjørnbærviken (B9)	66.27983	14.03550	MYTI EDU	SB	3										3							
#N/A	2014t	I969	Bjørnbærviken (B9)	66.27983	14.03550	MYTI EDU	SB	3										3							
1	2013t	36G	Færder	59.02717	10.52550	NUCE LAP	SB		1										1	1				1	
1	2014p	36G	Færder	59.02717	10.52550	NUCE LAP	SB		1										1	1				1	
1	2014t	36G	Færder	59.02717	10.52550	NUCE LAP	SB		1										1	1				1	
4	2013t	76G	Risøy	58.72800	9.27600	NUCE LAP	SB		1										1	1				1	
4	2014p	76G	Risøy	58.72800	9.27600	NUCE LAP	SB		1										1	1				1	
4	2014t	76G	Risøy	58.72800	9.27600	NUCE LAP	SB		1										1	1				1	
5	2013t	131G	Lastad	58.05550	7.70867	NUCE LAP	SB		1										1	1				1	
5	2014p	131G	Lastad	58.05550	7.70867	NUCE LAP	SB		1										1	1				1	
5	2014t	131G	Lastad	58.05550	7.70867	NUCE LAP	SB		1										1	1				1	
6	2013t	15G	Gåsøy (Ullerø)	58.05167	6.72167	NUCE LAP	SB		1										1	1				1	
6	2014p	15G	Gåsøy (Ullerø)	58.05167	6.72167	NUCE LAP	SB		1										1	1				1	
6	2014t	15G	Gåsøy (Ullerø)	58.05167	6.72167	NUCE LAP	SB		1										1	1				1	
7	2013t	227G1	Melandholmen	59.33350	5.31500	NUCE LAP	SB		1										1	1				1	
7	2014p	227G1	Melandholmen	59.33350	5.31500	NUCE LAP	SB		1										1	1				1	
7	2014t	227G1	Melandholmen	59.33350	5.31500	NUCE LAP	SB		1										1	1				1	
8	2013t	22G	Espevær (west)	59.57917	5.14833	NUCE LAP	SB		1										1	1				1	
8	2014p	22G	Espevær (west)	59.57917	5.14833	NUCE LAP	SB		1										1	1				1	
8	2014t	22G	Espevær (west)	59.57917	5.14833	NUCE LAP	SB		1										1	1				1	
9	2013t	98G	Svolvær area	68.25667	14.67667	NUCE LAP	SB		1										1	1				1	
9	2014p	98G	Svolvær area	68.25667	14.67667	NUCE LAP	SB		1										1	1				1	
9	2014t	98G	Svolvær area	68.25667	14.67667	NUCE LAP	SB		1										1	1				1	
10	2013t	11G	Brashavn	69.89867	29.74417	NUCE LAP	SB		1										1	1				1	
10	2014p	11G	Brashavn	69.89867	29.74417	NUCE LAP	SB		1										1	1				1	
10	2014t	11G	Brashavn	69.89867	29.74417	NUCE LAP	SB		1										1	1				1	

Appendix F
Temporal trend analyses of contaminants and biomarkers in biota 1981-2014

This Appendix is provided as an EXCEL file separate from this report but described below.

Only information for those time series that include data for either 2013 or 2014 is shown. The column headings are as follows:

Parameter Code: are described in **Appendix B**
IUPAC: Internation Union of Pure and Applied Chemistry (IUPAC) parameter name (if any).
CAS: Chemical Abstracts Services (CAS) parameter number (if any).
Parameter Name: Common name
Parameter Group: Parameters belong to one of 14 groups
Unit: µg/kg, mg/kg, ng/kg, etc.
Station Code
Station Name
Area: general area (if defined).
County
Water region: Water framework directive (WFD) water region
Water body ID: WFD water body identification
Water body name: WFD water body name

Species:
- **MYTI EDU**-Blue Mussel (*Mytilus edulis*)
- **LITT LIT**-Common periwinkle (*Littorina littorea*)
- **NUCE LAP**-Dog whelk (*Nucella lapillus*)
- **GADU MOR**-Atlantic cod (*Gadus morhua*)

Tissue:
- **SB**-Soft body tissue
- **LI**-Liver tissue
- **MU**-Muscle tissue
- **BL**-Blood
- **BI**-Bile

Basis: wet weight (**W**), dry weight (**D**) or lipid weight (**L**).
[Year columns]: median value for years 1981-2014.
Sample count [year]: number of samples analysed The first number within the parentheses indicates the number of pooled samples included. The second number within the parentheses indicates for mussels the total number of individuals used in all pooled samples and for cod the number individuals in each pooled sample.
SD [year]: standard deviation.
Class [year]: Norwegian Environment Agency classification (**1**, **2**, **3**, **4** or **5,** corresponding to the agency's Classes I, II, III, IV or V, repsectively) or below (**6**) or above (**7**) presumed "high background" concentration (see Appendix C).
EAC [year]: below (**<EAC**) or above (**>EAC**) OSPARs Environmental Assessment Criteria (EAC).
EQS [year]: below (UQ) or above (OQ) EU's Environmental Quality Standard (EQS). Note: the EU EQRs are based on the whole organism whereas monitoring of fish in MILKYS is on a particular tissue. Hence, comparison is only relevant if it is assumed that the concentration found is the same for all tissues in the fish.

OC: Overconcentration expressed as quotient of median of last year and upper limit to the agency's Class I or presumed "high background" ("m" missing background value).
Trend p(long)[year]: The statistical significance (p)[year] of the trend for the entire time series.
Detectable % change(long)[year]: the percent change that can be detected with 90 % confidence.
First Year(long)[year]: first year in time series.
Last Year(long)[year]: last year in time series.
Number of Years(long)[year]: number of years with data.

Trend p(short)[year]: The statistical significance (p)[year] of the trend for the last 10-year sampling period.
Detectable % change(short)[year]: the percent change that can be detected with 90 % confidence.
First Year(short)[year]: first year in time series for the last 10-year sampling period.
Last Year(short)[year]: last year in time series for the last 10-year sampling period.
Number of Years(short)[year]: number of years with data in time series for the last 10-year sampling period.

Trends [year]: levels and trends in concentrations of contaminants monitored. Classification is based on observed concentrations in cod, blue mussel, dog whelk and periwinkle. Tissues: soft body (SB), muscle (MU), liver (LI) and whole organism (WO). The classification system of the Norwegian Environment Agency is used for biota (Molvær *et al*. 1997: Classes: I (blue), II (green), III (yellow), IV (orange) and V (red) (see Appendix C). For biota, trend analyses were done on time series with five or more years. An upward (🡅) or downward (🡇) arrow indicates statistically significant trends, whereas a zero (O) indicates no trend. A small filled square (▪) indicates that chemical analysis was performed, but either the results were insufficient to do a trend analysis. Results marked with a star (★) indicate that there is insufficient data above the detection limit to perform a trend analysis. The result from the trend analysis for the entire time series (long term) is shown before the slash "/", and the result for the last 10 years (short term) is shown after the slash. Dark grey indicates concentrations higher than estimated high background levels. Light grey indicates concentrations lower than high background levels. Note: Class limits for ΣDDT are used for ppDDE. Note: the Trend for the previous year are based on time series where the last year has been excluded.

TREND_CHANGE_[year]-[year]: indicates the difference (if any) between the year-before-last results and the last year's results.
CLASS_CHANGE_[year]-[year]: indicates the difference (if any) between the year-before-last results and the last year's results.
EQS_CHANGE_[year]-[year]: indicates the difference (if any) between the year-before-last results and the last year's results.
EAC_CHANGE_[year]-[year]: indicates the difference (if any) between the year-before-last results and the last year's results.

Note on detection limit in trend analyses: half of the limit is used, however if a substance is included as part of a sum (e.g. PCB-7) then null is used. Note, that the number of such cases and position in a times series may affect whether or not a trend analyses can be applied (see Chapter 2.7.2).

Appendix G
Passive sampling result-tables

As part of the batch of analysis of samplers from the 2014-2015 survey, a QA spiked samplers was analysed for substances of interest. This allows us to gauge the performance of the extraction and analysis over time. The table below (***Table 37***) show the contaminant concentrations measured in the QA spiked sampler. For most substances concentrations measured are close to the mean concentrations from the six QA spiked samplers analysed previously. Some substances (e.g. BDEs 196 and 209) exhibit wider deviations and this is shows the challenge in analysing these substances.

Table 37*. Comparison of concentrations of substances of interest measured in the two QA spiked samplers with data from the initial evaluation of the QA spiked samplers.*

Substance	Mean concentration in ng g^{-1} (% RSD) [4]	QA Spike (ng g^{-1})
Alkylphenols [1]		
4-t-OP	79 (12)	79
4-t-NP	289 (10)	293
4-n-OP	72 (18)	73
4-n-NP	64 (4)	62
HBCD [2]		
α-HBCD	2.5 (11)	2.7
β-HBCD	2.7 (13)	2.9
γ-HBCD	2.3 (21)	2.8
PBDEs [3]		
BDE 47	4.5 (9)	4.4
BDE99	4.4 (12)	4.6
BDE100	3.0 (8)	2.7
BDE126	2.3 (11)	-
BDE153	2.2 (14)	2.0
BDE154	2.0 (15)	1.5
BDE183	2.2 (21)	2.0
BDE196	1.7 (20)	1.2
BDE209	4.1 (18)	3.4

1) 4-t-OP: para-t-octylphenol; 4-t-NP : para-t-nonylphenol; 4-n-OP: para-n-octylphenol; 4-n-NP : para-n-nonylphenol
2) HBCD: Hexabromocyclododecane
3) PBDE: Polybrominated diphenyl ether
4) Mean concentration in the first six QA spiked samplers

The table below (***Table 38***) shows Water Framework Directive Environmental Quality Standards for substances of interest for the passive sampling work. These have been set for the "Whole Water" (as opposed to passive samplers measuring the freely dissolved concentration).

Table 38*. Annual average and maximum acceptable concentration environmental quality standard set by the European Union's Water Framework Directive (2013/39/EU).*

	Water Framework Directive EQS (μg L^{-1})	
	AA-EQS	**MAC-EQS**
Octylphenol*	0.01	Not applicable
Nonylphenol**	0.3	2.0
PBDEs***		0.014
HBCD	0.0008	0.05

*with CAS number 1806-26-4 (including compound with CAS number 140-66-9)
**with CAS number 25154 (including compounds with CAS numbers 104-40-5 and84852-15-3)
***only tetra, penta, hexa and heptabromodiphenyl ether (CAS numbers 40088-47-9, 32534-81-9, 36483-60-0, 68928-80-3)

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