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Tittel - norsk og engelsk
Occurrence of selected organic micropollutants and silver at wastewater treatment plants in Norway
Forekomster av utvalgte organiske mikroforurensninger og sølv fra avløpsrenseanlegg i Norge

Sammendrag – summary

På oppdrag fra Klima- og forurensningsdirektoratet (KLIF) har Norsk institutt for vannforskning (NIVA) og Norsk institutt for luftforskning (NILU) foretatt prøveinnsamling og analysert bisfenol A, tetrabrombisfenol A, bromerte flammehemmere (polybromerte difenyletere og hexabromsyklodekan), perfluorerte forbindelser, kort- og medium-kjedede parafiner, syntetiske musk forbindelser og sølv i utløp og avvannet slam fra 8 norske avløpsrenseanlegg.

As an assignment from the Norwegian Climate and Pollution Agency (Klif), the Norwegian Institute for Water Research (NIVA) and the Norwegian Institute for Air Research (NILU) have analysed and reported concentrations of selected organic contaminants in the effluent from eight Norwegian wastewater treatment plants (WWTPs), and the level of these compounds in dewatered sludge from the same plants.

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Wastewater treatment plants Organic micropollutants Sludge Effluent water	Avløpsrenseanlegg Organiske mikroforurensninger Slam Utløpsvann

Preface

As an assignment from the Norwegian Climate and Pollution Agency (Klif), the Norwegian Institute for Water Research (NIVA) and the Norwegian Institute for Air Research (NILU) have analysed and reported concentrations of selected organic contaminants in the effluent from eight Norwegian wastewater treatment plants (WWTPs), and the level of these compounds in dewatered sludge from the same plants.

The contaminants analysed were bisphenol A, tetrabromobisphenol A, brominated flame retardants (polybrominated diphenyl ethers and hexabromocyclododecane), perfluorinated compounds, short- and medium chain chlorinated paraffins and synthetic musk compounds. In addition, silver was analysed.

NIVA has been KLIFs contract partner and administrative responsible for the project.

NIVA personnel involved have been: Katherine Langford (data handling and reporting), Tone Muthanna (data handling and reporting), Kevin V. Thomas (data handling and reporting), Zuliang Liao (sampling) and Helge Liltved (project manager).

The analytical work was performed by NILU in collaboration with The University of Umeå, Sweden. The following personnel have been involved: Martin Schlabach (project manager at NILU), Ellen Katrin Enge (responsible for sample preparation), Anders Borgen (ionic PFAS, SCCP, MCCP), Mebrat Ghebremeskel (volatile PFAS), Hans Gundersen (BFR), Henriette Leknes (BPA, TBBPA), Hilde Uggerud (Ag) and Peter Haglund, University of Umeå (OPFR).

Bård Nordbø has been Klif's project manager and contact person.

We will use the opportunity to thank all colleagues and collaborators for good cooperation in completing this project. A special thank you to the personnel at the WWTPs for their help in collecting and sending samples, and providing required information during the sampling period.

All data on chemical analysis are included in Appendix A.

Oslo, 2011-03-18

Helge Liltved
Research manager

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Summary

As an assignment from the Norwegian Climate and Pollution Agency (Klif), the Norwegian Institute for Water Research (NIVA) and the Norwegian Institute for Air Research (NILU) have analysed and reported concentrations of selected organic contaminants in the effluent from eight Norwegian wastewater treatment plants (WWTPs), and the level of these compounds in dewatered sludge from the same plants. Four of the plants were equipped with combined chemical and biological treatment (Bekkelaget, HIAS, RA-2 and VEAS), while the other four were only using chemical precipitation (Arendal RA, FREVAR, Sandefjord RA and TAU). Differences in treatment process applied in the WWTPs will influence the concentrations of the various contaminants found in effluent water and sludge according to chemical and physical characteristics of the contaminants.

The contaminants analysed were bisphenol A, tetrabromobisphenol A, brominated flame retardants (polybrominated diphenyl ethers and hexabromocyclododecane), perfluorinated compounds, short- and medium chain chlorinated paraffins and synthetic musk compounds. In addition, silver was analysed.

The concentrations found were compared to published predicted no-effect concentrations (PNEC) and Norwegian standards for the classification of surface waters and sediments. The comparison was used to indicate the risks posed by the concentrations of the various compounds in the effluents and sludge. The mass transport of the various pollutants from the WWTPs was estimated by multiplying the effluent concentrations with average flowrate.

The most important findings can be summarised as follows:

- All of the compounds selected for screening were detected in the dewatered sludge. With the exception of HBCD and TBBPA, all of the compounds were detected in the WWTPs effluents at measurable concentrations, and are therefore being released into the Norwegian aquatic environment.
- The levels detected pose very little risk to microorganisms in biological treatment plants, and for many of the compounds, it also appears that they pose little risk to the receiving aquatic environments.
- It is possible that the OPFRs, TBEP and TCP pose a risk to aquatic organisms at the point of discharge at certain locations. Paucity of ecotoxicity data for deca-BDE, and a lack of understanding of silver speciation, makes an assessment of the environmental risk they pose difficult.
- The levels of individual chemicals detected in sludge pose little direct risk to soil dwelling organisms.
- Risks associated with persistence, bioaccumulation and the combined effect of mixtures of chemicals below PNEC values, have not been possible to estimate in the current study.

Bisphenol A and Tetrabromobisphenol A

Bisphenol A (BPA) was detected in all but one of the WWTP effluent samples collected at a median concentration of 327 ng/L. The median BPA concentration in sludge was 425 ng/g. Tetrabromobisphenol A (TBBPA) was not detected at a concentration higher than the limit of detection (LOD) in any of the effluent samples collected. The concentration of TBBPA in sludge ranged from below the LOD value to129 ng/g. There is little risk to the WWTP

microorganisms from the levels of BPA measured in the final effluent. Effluent concentrations at certain locations were above the predicted no-effect concentration (PNEC) for water and marine water. The risk to the aquatic environment from releases of TBBPA from WWTP is low since the levels detected were all below the limits of detection. There is little risk to soil dwelling organisms from the levels determined in sludge if dried sludge should be disposed on land.

Polybrominated diphenyl ether (PBDE)

Deca-BDE (BDE-209) was the dominant congener in both sludge and effluent samples with significantly higher concentrations detected in sludge samples. The median concentrations for BDE-209 were 335 ng/g and 2.1 ng/L in sludge and effluent, respectively. The sum-PBDE concentration measured in sludge varied for different WWTP, from 68 to 427 ng/g as median values. The congener distribution in sludge shows the same distribution pattern as has been frequently reported elsewhere in the literature, with the highest concentrations measured for BDE-209, 154, 153, 100, 99 and 47. In the effluent samples, the 2 penta congeners BDE-99 and BDE-100, and the tetra congener BDE-47, were the most frequently detected compounds, with the exception of BDE-209. Comparing the median sum PBDE level in sludge to the lowest $\text{PNEC}_{\text{Soil}}$ value suggests that there is little risk to soil dwelling organisms.

Hexabromocyclododecane (HBCD)

All 3 stereoisomers of hexabromocyclododecane (HBCD) were detected in sludge samples with the highest median value measured for γ-HBCD (4.5 ng/g). However, the highest maximum value was measured for α-HBCD (71 ng/g). Effluent concentrations were typically < LOD, indicating little risk posed by HBCD to WWTP microorganisms or aquatic organisms in receiving waters.

Perfluorinated compounds (PFCs)

6:2 FTS was detected in all but one of the effluent samples at a median concentration of 5.8 ng/L. Very little 6:2 FTS was sorbed to sludge. 8:2 FTS was not detected in any sample of sludge or effluent. PFOS was detected in all samples. 8:2 FTOH was detected in all sludge samples and had the highest median value of all of the perfluorinated compounds studied. Both of the fluorotelomer alcohols were less predominant in the effluent samples with median values for 8:2 FTOH and 10:2 FTOH of 1.7 ng/L and <LOD, respectively. Data for two additional fluorotelomer compounds, 4:2 and 6:2 FTOH, were also included for comparison. These were less frequently detected.

Organophosphate flame retardants (OPFR)

All but two of the organophosphate flame retardants (OPFR) were detected in all of the sludge samples collected. DPhBP and TEHP were the two compounds that were not detected at quantifiable concentrations. The pattern of compounds was generally similar between all of the WWTPs, with TCPP, TBEP and EHDPP dominating (Figure 9). TBEP was present at the highest concentration in the majority of samples with a median concentration of 3,660 ng/g. This was followed by TCPP (median=2,580 ng/g), EHDPP (median=2,300 ng/g) and TCP (median= 460 ng/g). Median concentrations of the other OPFRs were typically below 200 ng/g. TCPP was present at the highest median concentration (600 ng/L) in WWTP effluent. The OPFR pattern in effluent was dominated by TCPP, TIBP and TBEP. Four of the effluent samples collected contained concentrations of TBEP greater than the $\text{PNEC}_{\text{Marine water}}$ of 1,300 ng/L, whilst seven of the samples contained TCP concentrations significantly above the $\text{PNEC}_{\text{Marine water}}$ of 3.2 ng/L, suggesting the potential for risks to aquatic organisms at the point of discharge. The median sludge concentration for TCPP, TBEP, TCP and TPhP was greater than the $\text{PNEC}_{\text{Soil}}$, however the concentration in soil would be expected to be below the $\text{PNEC}_{\text{Soil}}$ following moderate dilution upon application to land.

Short and medium chained paraffins (SCCP and MCCP)
Short chain chlorinated paraffins (SCCP) were detected in 65% of the effluent samples collected at a median concentration of 102 ng/L. SCCP were detected in all sludge samples at a median concentration of 416 ng/g. Medium chain chlorinated paraffins (MCCP) were only detected at quantifiable levels in 13% of the samples analysed. MCCP were detected in all of the sludge samples collected at levels of between 14 and 7,000 ng/g. The data suggest little or no risk to various environmental compartments from the levels determined when compared with relevant PNEC data.

Silver
Silver was measured in concentrations above the limit of detection in all of the effluents samples and in all but one of the sludge samples. Concentrations in effluent ranged from 0.01 to 0.49 µg/L, with a median concentration of 0.05 µg/L. Concentrations in sludge ranged from <0.01 to 9.55 µg/g, with a median concentration of 2.52 µg/g. With increased focus on the occurrence of silver in both WWTP effluent and sludge due to its increasing use, there is a need for an improved understanding of the speciation of silver in WWTPs and the toxicity of these species before the environmental risks can be assessed.

Synthetic musk compounds
AHTN and HHCB were detected at quantifiable concentrations in all of the samples analysed. WWTP effluent concentrations of HHCB were much higher than those for AHTN. A similar pattern was observed in sewage sludge with both compounds being detected at quantifiable concentrations in all samples. Again HHCB was present at much higher concentrations than AHTN. The data suggest little or no risk to various environmental compartments from the levels determined when compared with relevant PNEC data.

Sammendrag

På oppdrag fra Klima- og forurensningsdirektoratet (KLIF) har Norsk institutt for vannforskning (NIVA) og Norsk institutt for luftforskning (NILU) foretatt prøveinnsamling og analysert bisfenol A, tetrabrombisfenol A, bromerte flammehemmere (polybromerte difenyletere og hexabromsyklodekan), perfluorerte forbindelser, parafiner med korte og middels lange kjeder, syntetiske musk forbindelser og sølv i utløp og avvannet slam fra 8 norske avløpsrenseanlegg. Fire av renseanleggene er utstyrt med kombinert kjemisk og biologisk rensing (Bekkelaget, HIAS, RA-2 and VEAS), mens de andre fire bare benytter kjemisk felling som hovedrensemetode (Arendal RA, FREVAR, Sandefjord RA and TAU). Forskjeller i renseprosess i de ulike renseanleggene vil innvirke på konsentrasjonene av de ulike forbindelsene i utløpsvann og slam avhengig av forbindelsenes fysiske og kjemiske egenskaper.

De målte konsentrasjonene ble sammenliknet med publiserte ”predicted no-effect concentrations (PNEC)” og norske standarder for klassifisering av overflatevann og sedimenter. Sammenlikningen ble gjort for å indikere hvilken risiko hver av de enkelte forbindelsene representerer i avløpsvann og slam. Mengden per døgn som ble sluppet ut av de ulike forbindelsene ble beregnet ved å multiplisere konsentrasjon med gjennomsnittlig vannmengde per døgn.

De viktigste funnene i undersøkelsen kan oppsummeres som følger:

- Alle forbindelsene som inngikk i undersøkelsen ble påvist i avvannet slam. Med unntak av HBCD og TBBPA, ble også alle forbindelsene som inngikk i undersøkelsen funnet i utløpet fra ett eller flere renseanlegg, og blir derfor tilført norske vannforekomster.
- Konsentrasjonene som ble påvist representerer en veldig liten risiko for mikroorganismene i biologiske renseprosesser, og det synes også som de representerer en liten risiko for det akvatiske miljø.
- Det er mulig at OPFR, TBEP og TCP representerer en risiko nær utslippspunktet ved noen lokaliteter. Manglende økotoksikologiske data for deca-BDE, og manglende kunnskap om spesiering av sølv i avløpsvann, gjør det vanskelig å bestemme hvilken miljømessig risiko sølv representerer.
- Hver for seg representerer ikke de målte nivåene av kjemikalier i slam noen risiko for organismer i jord ved anvendelse i jordbruket.
- Eventuell risiko forbundet med persistens i miljøet, bioakkumulering og kombinerte effekter av kjemikaliene ble ikke vurdert nærmere i denne studien.

Bisfenol A (BPA) og tetrabrombisfenol A (TBBPA)
Bisfenol A (BPA) ble målt i samtlige utløpsprøver fra renseanleggene, med en medianverdi på 327 ng/L. Mediankonsentrasjonen i slam var 425 ng/g. Tetrabromobisfenol (TBBPA) ble ikke funnet i noen av utløpsprøvene. Analysemetoden har en nedre deteksjonsgrense på <15 ng/L. TBBPA-konsentrasjonen i slam varierte fra under deteksjonsgrensen til 129 ng/g. Konsentrasjonen av BPA målt i utløpsvannet fra renseanleggene utgjorde liten risiko for mikroorganismene i biologiske renseanlegg. Utløpsprøvene ved noen av renseanleggene oversteg PNEC-grensen for ferskvann og sjøvann. TBBPA utslipp fra renseanleggene utgjør liten risiko for det akvatiske miljøet siden alle konsentrasjonene var under deteksjonsgrensen.

Konsentrasjonene målt i slam representerer liten risiko for organismer i jord ved anvendelse av slam til jordforbedring.

Polybromerte difenyletere (PBDE)

Deca-BDE (BDE-209) var den dominerende formen av PBDE, både i utløpsvann og slam. Konsentrasjonene i slam var betydelig høyere enn i vann. I slam var medianverdien 335 ng/g, og i vann 2,1 ng/L. Summen av PBDE-konsentrasjonene i slam varierte mellom de ulike renseanleggene. Medianverdien varierte fra 68 til 427 ng/g. Mengden av de ulike formene for PBDE i slam var i overensstemmelse med hva som tidligere er rapportert, med høyeste konsentrasjoner for BDE-209, 154, 153, 100, 99 og 47. I utløpsprøvene ble 2-penta formene (BDE-99 og BDE-100) og tetra formen (BDE-47) identifisert som stoffene med høyeste konsentrasjoner etter BDE-209. Ved å sammenlikne medianverdiene for sum PBDE i slam med laveste $\mathrm{PNEC}_{\mathrm{Soil}}$ verdi, tilsier tallene at det er liten risiko for uheldige effekter på organismer i jord ved bruk av slam til jordforbedring.

Hexabromsyklododekan (HBCD)

Alle de tre stereoisomerne av hexabromsyklododekan (HBCD) ble funnet i slamprøvene. Den høyeste medianverdien ble funnet for γ-HBCD (4,5 ng/g), mens den høyeste maksimumskonsentrasjonene ble målt for α-HBCD (71 ng/g). Konsentrasjonen i utløpsprøvene var under deteksjonsgrensen, og utgjør dermed ingen risiko for akvatiske organismer i resipientene.

Perfluorerte forbindelser (PFCs)

6:2 FTS ble funnet i alle utløpsprøvene, bortsett fra i en prøve. Mediankonsentrasjonen for alle anleggene var 5,8 ng/L. Det ble funnet lave konsentrasjoner av 6:2 FTS i slamprøvene. 8:2 FTS ble ikke funnet i noen av slam- eller vannprøvene. PFOS ble funnet i alle slam- og utløpsprøvene. 8:2 FTOH ble funnet i alle slamprøvene, og hadde den høyeste medianverdien av alle de perfluorerte forbindelsene. De to fluortelomer alkoholene (8:2 FTOH og 10:2 FTOH) ble i liten grad funnet i vannprøvene, med medianverdi på 1,7 ng/L for 8:2 FTOH og under deteksjonsgrensen for 10:2 FTOH. Analyse av 4:2 FTOH og 6:2 FTOH ble også inkludert i undersøkelsen for om mulig å sammenlikne, men ble bare detektert i noen prøver.

Organofosfatflammehemmere (OPFRs)

Alle organofosfatflammehemmere, bortsett fra to, ble funnet i alle slamprøvene. DPhBP og TEHP var de to forbindelsene som ikke ble funnet i målbare konsentrasjoner. Sammensetningen av OPFR forbindelser var generelt sammenliknbar mellom alle avløpsrenseanleggene, med TCPP, TBEP og EHDPP som de dominerende forbindelsene (Figure 9). TBEP ble funnet i høyest konsentrasjon i de fleste prøvene, med en mediankonsentrasjon på 3660 ng/g. Denne ble fulgt av TCPP (median=2580 ng/g), EHDPP (median=2300 ng/g) og TCP (median=460ng/g). Mediankonsentrasjonen av de andre OPFR var typisk lavere enn 200 ng/g. TCPP ble funnet i høyest konsentrasjon i utløpsvannet (median=600 ng/L). Sammensetningen i utløpsvann ble dominert av TCPP, TIBP og TBEP. Fire av avløpsprøvene inneholdt konsentrasjoner av TBEP som overskred $\mathrm{PNEC}_{\mathrm{Marine\ water}}$ verdien på 1300 ng/L, mens syv av prøve hadde TCP konsentrasjoner som var signifikant høyere enn $\mathrm{PNEC}_{\mathrm{Marine\ water}}$ på 3,2 ng/L, noe som tilsier en potensiell risiko overfor akvatiske organismer ved utslippspunktet. Mediankonsentrasjonen i slam for TCPP, TBEP, TCP og TPhP var høyere enn $\mathrm{PNEC}_{\mathrm{Soil.}}$ Imidlertid forventes konsentrasjonen og bli lavere enn $\mathrm{PNEC}_{\mathrm{Soil}}$ ved moderat fortynning ved bruk på jordbruksarealer.

Parafiner med korte og middels lange kjeder (SCCP og MCCP)
Parafiner med korte kjeder (SCCP) ble påvist i 65% av avløpsprøvene med en mediankonsentrasjon på 102 ng/L. SCCP ble funnet i alle slamprøver med en mediankonsentrasjon på 416 ng/g. Parafiner med middles lange kjeder (MCCP) ble bare påvist i 13% av utløpsprøvene. MCCP ble påvist i alle slamprøvene i nivåer mellom 14 og 7000 ng/g. Sammenlikning med PNEC data tilsier konsentrasjonsnivåene som ble funnet liten eller ingen risiko ved utslipp eller ved bruk av slam.

Sølv
Sølv ble funnet i alle utløpsprøvene i konsentrasjoner fra 0,01 til 49 µg/L, med medianverdi på 0,05 µg/L. Sølv ble også funnet i alle slamprøvene med konsentrasjoner fra <0,01 til 9,55 µg/g, med medianverdi på 2,52 µg/g. Sølv er i økende grad brukt i forbruksvarer, noe som gjenspeiles i konsentrasjonene i avløpsvann og slam. Det er behov for mer kunnskap om de ulike formene av sølv som forekommer i avløpsvann og slam, og toksisiteten til disse for å kunne bestemme miljørisiko.

Syntetiske muskforbindelser
AHTN og HHCB ble identifisert i alle utløpsvann- og slamprøvene. I utløpsvannprøvene ble HHCB funnet i mye høyere konsentrasjoner enn AHTN. Tilsvarende ble funnet i slam med betydelig høyere konsentrasjoner av HHCB enn AHTN. De målte konsentrasjonene synes ikke å representere en miljørisiko i forhold til relevante PNEC-verdier.

1. Introduction

The discharge of organic matter and nutrients from Norwegian domestic wastewater treatment plants (WWTPs) is regulated on a national level to limit the total load to the environment, thereby minimizing potential problems with high oxygen consumption and eutrophication in the receiving waters. So far, no such regulations exist regarding organic micropollutants in final effluents from Norwegian WWTPs. WWTPs are designed and dimensioned to achieve their prescribed removal of organic matter and nutrients, where the discharge limits are given by the sensitivity of the local recipient and the size of the particular treatment plant.

A number of studies have focused on the removal and effluent concentrations of organic micropollutants, such as polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), nonylphenols and their ethoxylates, phthalates and linear alkylbenzene sulphonates (LAS) at WWTPs applying different types of biological treatments (McNally et al., 1998; Fauser et al., 2003; Marttinen et al., 2003; Blanchard et al., 2004; Katsoyiannis and Samara, 2004, Vogelsang et al. 2006). Other contaminants, such as bisphenol A, tetrabromobisphenol A, brominated flame retardants (polybrominated diphenyl ethers and hexabromocyclododecane), perfluorinated compounds, short and medium chain chlorinated paraffins and synthetic musk compounds have received relatively less attention. These compounds were focused on in the present study by collecting and analysing effluent and sludge samples from eight Norwegian WWTPs. The study was performed to improve our understanding about the concentrations of these substances in effluent from WWTPs, and to estimate and evaluate the possible impact on the environment.

Bisphenol A and tetrabromobisphenol A
Bisphenol A (BPA; 4,4’-Isopropylidenediphenol [CAS no. 80-05-7]) is used to make polycarbonate plastics and epoxy resins as well as other applications. BPA based polycarbonate and epoxy resin products are widespread in society and there is concern over the use of BPA since it is known to be estrogenic. BPA is not produced in Norway, however around 28 tonnes of the chemical is used annually (www.miljøstatus.no). In Norway BPA is considered a priority hazardous substance with the objective of significantly reducing releases into the environment by 2020. Tetrabromobisphenol A (TBBPA; [79-94-7]) is a flame retardant that is a derivative of BPA. TBBPA is included with BPA since under certain circumstances TBBPA can be transformed to BPA. TBBPA is a priority hazardous substance in Norway with around 290 tonnes being used annually.

Hexabromocyclododecane (HBCD)
Hexabromocyclododecane (HBCD; [3194-55-6]) is a brominated flame retardant that exists as 3 distereoisomers. The disteroisomers have different water solubilities and Log K_{ow} values, which impact their fate and behaviour during wastewater treatment processes. Their water solubilities are 48.8, 14.7 and 2.1 µg/L for α, β and γ- HBCD respectively. Their Log K_{ow} are not so different being 5.07, 5.12 and 5.47 for α, β and γ-HBCD respectively (EU, 2008). HBCD is classified, along with other brominated flame retardant as a priority hazardous substance in Norway. Emissions to the environment should be significantly reduced by 2010 and eliminated by 2020.

Polybrominated diphenyl ethers (PBDE)

Polybrominated diphenyl ether (PBDE) flame retardants are used in a wide variety of mainly plastic products. They are structurally similar to PCBs and as such, their environmental fate follows a similar pattern. PBDEs are priority hazardous substances in both Norway and the EU.

Table 1. Log K_{ow} values for polybrominated diphenyl ether flame retardants (de Wit, 2002; Langford and Lester, 2002; Tittlemier et al., 2002)

PBDE congener	Log K_{ow}
Deca-BDE	9.97
Octa-BDE	8.35-8.9
Hexa-BDE	6.86-7.92
Penta-BDE	6.64-6.67
Tetra-BDE	5.87-6.16
Tri-BDE	5.47-5.58

Perfluorinated compounds (PFCs)

Perfluorinated compounds (PFCs) are a large group of compounds with a high surface activity. The fluoroalkyl tails are both hydrophobic and oleophobic (oil repelling) and are therefore useful as stain resistant fabric protection, water proofing outdoor clothing and in non stick products, as well as reducing the surface tension in hydraulic fluids and fire fighting foams for example. N-EtFOSE is used as a paper coating in food products. In households PFCs are used in fabric coatings and cleaning products. The high surface activity of PFOS and PFOA make it difficult to determine some physico-chemical properties such as Log K_{ow} but it is this high surface activity that makes PFC compounds so useful. Their use in commercial products results in them entering wastewater treatment systems.

Fluorotelomer alcohols (FTOH) are volatile precursors to the formation of more persistent PFOA and PFOS, and have demonstrated estrogen-like properties (Maras *et al.*, 2006). As the use of the original PFC compounds, such as perfluorooctane which is a precursor of PFOS, decreases, the use of longer chain compounds such as the fluorotelomer compounds is increasing. High sorption rates have been observed for the longer chain compounds suggesting that sludge and sediment may be a sink for them. PFCs are classified as priority hazardous substances in Norway with a target of significantly reducing their emissions by 2010. The use of PFOS in fire-fighting foam and textiles is already banned in Norway and the use of PFOA is currently under review by the EFTA.

Organophosphate flame retardants

Organophosphate esters are high production chemicals with many applications, such as flame retardants and plasticizers. Chlorinated alkylphosphates, such as tris-(2-chloroisopropyl)-phosphate (TCPP), tris-(1,3-dichloroisopropyl)-phosphate (TDCP) and tris-(2-chloroethyl)-phosphate (TCEP) are mainly used as flame retardants in polyurethane foams. So called non-derivatised alkylphosphates such as triphenylphosphate (TPhP), tributylphosphates (both iso- and n- isomer; TIBP resp. TNBP) and tris-(butoxyethyl)-phosphate (TBEP) are used as plasticisers, lubricants and flame retardants. Chlorinated- and certain non-chlorinated-alkyl phosphates (TBEP) are a particular concern since they are carcinogens or suspected carcinogens.

Short and medium chain chlorinated paraffins
Chlorinated paraffin formulations are complex mixtures that contain a large number of structural isomers. These complex mixtures of chlorinated paraffins are classified into groups defined by their chain length and degree of chlorination. Short chain chlorinated paraffins (SCCP; [85535-84-8]) are defined as C_{10} to C_{13} paraffins with greater than 48% chlorination by weight. Medium chain chlorinated paraffins (MCCP; [85535-85-9]) are defined as having carbon chain lengths of between C_{14} and C_{17}. SCCP and MCCP have been in use for over thirty years primarily being used in metal working fluids, as flame retardants in rubber, as plasticizers in coatings and sealants and also in leather and textiles. In Norway this amounts to approximately 35 tonnes of SCCP per year, with a general decline in the amount being used. Releases into the environment are the result of production, processing and use of products containing SCCP and MCCP. SCCP and MCCP and are classified in Norway by Klif as priority hazardous substances. SCCP are classified as a group of substances ‘*whose emissions must be substantially reduced and eliminated before 2005*’ and MCCP as a group *‘whose emissions must be reduced substantially before 2010’*. SCCP are also classified as priority hazardous substances under the EUs Water Framework Directive (WFD).

Silver
Silver (CAS 7440-22-4) is a metallic chemical element commonly used in products that exploit its precious metal, conductive and bactericidal properties. Metallic silver is often used in currency, metallic decoration and mirrors as well as being commonly found in electrical and electronic products. Silver is increasingly being used as a bactericide, often in the form of nanoparticles, and is frequently found in household items, clothing and footwear.

Synthetic musk compounds
The synthetic musk compounds AHTN (7-acetyl-1,1,3,4,4,6-hexamethyltetralin; [1506-02-1 or 21145-77-7]; marketed as Tonalide, Fixolide) and HHCB (1,3,4,6,7,8-hexahydro-4,6,6,7,8,8-hexamethylcyclopenta- (g)-2-benzopyran; [1222-05-5] marketed as Galaxolide, Abbalide) are synthetic musk fragrances, which are used in almost all scented consumer products, such as perfumes, cosmetics and laundry detergents. They are known to be ubiquitous contaminants in domestic sewage effluent and sludge, as well as receiving environments. Both AHTN and HHCB are relatively hydrophobic chemicals with reported log K_{OW} of between 5.7 and 6.3. The risks of AHTN and HHCB to human health and the environment have been assessed by the EU. They concluded that according to their environmental risk assessment ‘*there is at present no need for further information and/or testing and no need for risk reduction measures beyond those which are being applied already*’ for both AHTN and HHCB. Musk xylenes are classified in Norway as hazardous substances ‘*whose emissions must be reduced substantially before 2010’*

The main detrimental effects of organic micropollutants are connected to their potential acute toxicity or sub-lethal effects on the biota (Eljarrat and Barceló, 2003). Toxic effects of effluent samples from WWTPs to algae, crustacean and fish have frequently been reported (Fischer et al., 1998; Schroder et al., 1991; Aguayo et al., 2004), and endocrine disruption of fish and freshwater mussels has been observed in rivers downstream biological WWTPs (Sumpter, 1995; Gagne et al., 2001; Tilton et al., 2002). The latter was primarily ascribed to the presence of estrogenic chemicals (estrogen mimics) in the WWTP effluents.

In addition to toxicity, organic compounds can also be classified by their persistence and potential to bioaccumulate. This classification approach is applied to compounds when their properties are such that ‘safe’ concentrations of these chemicals cannot be reliably estimated. Compounds identified to be persistent (P), bioaccumulative (B) and toxic (T) are deemed as hazardous to the environment. According to the EU, compounds are classified according to the following limits:

- P: half-life > 60 d in marine water or
- half-life > 40 d in freshwater or
- half-life > 180 d in marine sediment or
- half-life > 120 d in freshwater sediment
- vP: half-life > 60 d in marine- or freshwater or
- half-life > 180 in marine or freshwater sediment
- B: BCF > 2,000
- vB: BCF > 5,000
- T: chronic NOEC < 0.01 mg/l or CMR or endocrine disrupting effects

Compounds included in this study that are classified as persistent (P), bioaccumulative (B) and toxic (T) (PBT or vPvB) substances include; octa-PBDE, penta-BDE, HBCD, SCCP and PFOS. As PBT/vPvB substances their occurrence in the environment at concentrations below predicted no-effect concentrations (PNEC) does not necessarily safeguard the environment.

The objective of the present study was to investigate the occurrence of selected organic pollutants in effluent and dewatered sludge from eight Norwegian WWTPs. The concentrations found were compared to published predicted no-effect concentrations (PNECs) and Norwegian standards for the classification of surface waters and sediments. The comparison was used to indicate the risks posed by the concentrations of the various compounds in the effluents and sludge. The mass transport of the various pollutants from the WWTPs was estimated by multiplying the effluent concentrations with average flowrate.

2. Materials and methods

2.1 Wastewater treatment plants

Effluent water and sludge from eight WWTPs were sampled in this study. The WWTPs were located in the eastern and south-eastern parts of Norway (Figure 1) and had a relatively wide variation in capacity (0.8 – 110 mill m^3/year). Four of the plants were equipped with combined chemical and biological treatment (Bekkelaget, HIAS, RA-2 and VEAS), while the other four were only using chemical precipitation (Arendal RA, FREVAR, Sandefjord RA and TAU). Details from the WWTPs are shown in Table 2.

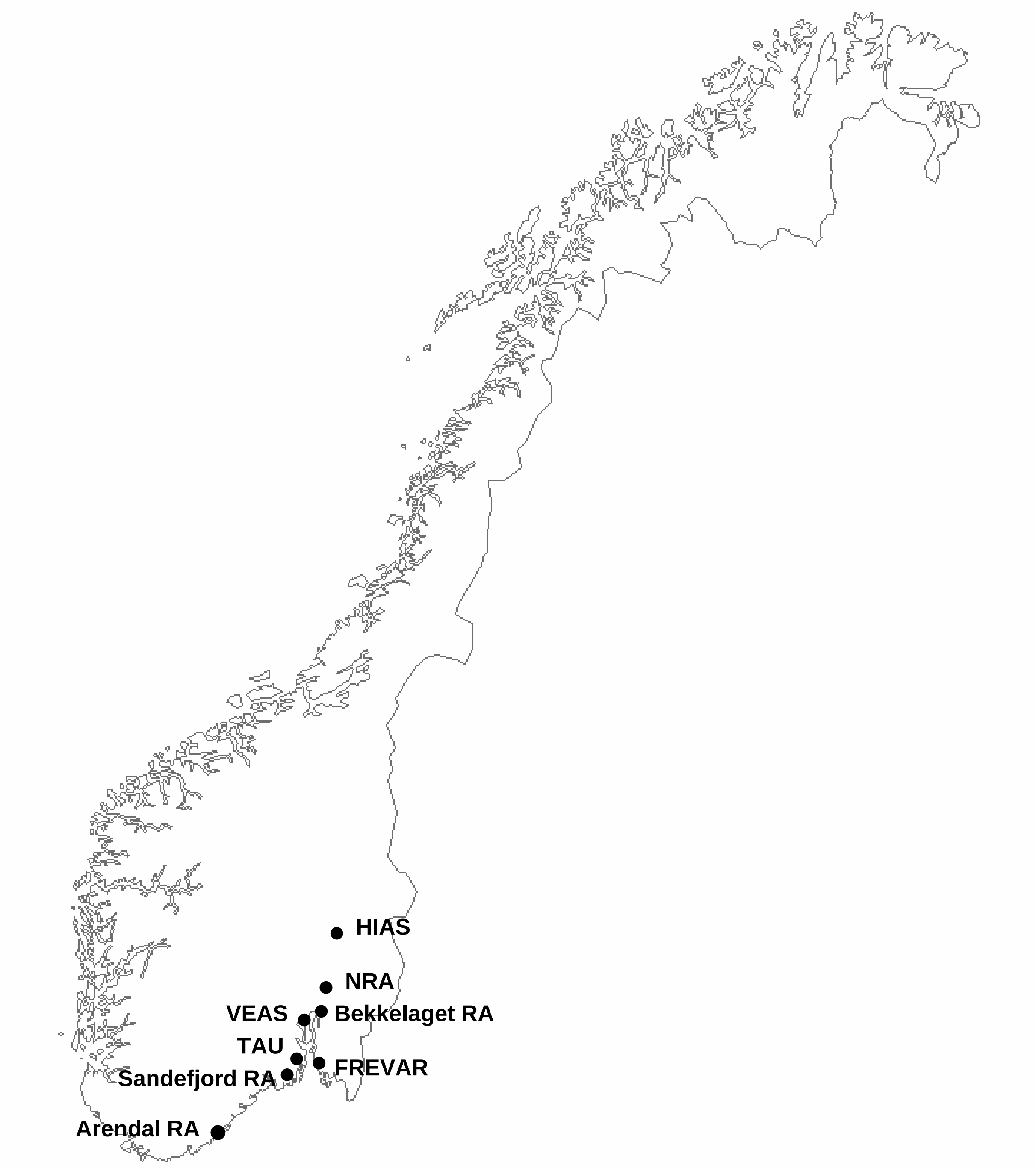

Figure 1. The names and locations of WWTPs involved in this screening project.

Table 2. Operational details of the wastewater treatment plants included in the study.

WWTPs	Municipalities	Wastewater treatment	Sludge treatment
Arendal RA	Arendal	Ca 4.75 mill. m^3/year (62,000 p.e.). 3 mm grid and 0.22 mm fibre strainer before coagulation with PAX18, separation by sedimentation. Cationic polyacrylamide (Pempure PW22) as dewatering agent.	Ca. 6,500 tonnes dewatered (ca 25-30% DS) final sludge per year. Anaerobic digestion and centrifuges.
Bekkelaget RA	Oslo, Oppegård, Nittedal	Ca. 40 mill. m^3/year (280,000 p.e.). 3 mm grid, sand trap and primary sedimentation before anoxic (~6 h) and aerobic (~6 h) activated sludge process with simultaneous precipitation with iron sulphate. Polishing by sand filtration. Anionic polyacrylamide (Cytec A120) as dewatering agent.	Ca. 14,000 tonnes final dewatered sludge per year. Two-stage thermophilic and anaerobic digestion. Centrifuges for dewatering to 30% DS.
FREVAR	Fredrikstad	Ca. 13.4 mill m^3/year (75,000 p.e.). Chemical primary sedimentation process for particle and partial BOD removal. The effluent is disinfected before discharge.	Ca. 8,000 tonnes final dewatered (29% DS) sludge per year. Anaerobic digestion is used, followed by centrifuges.
HIAS	Hamar	Ca. 8 mill m^3/year (55,000 p.e.). Screening, sand trap, primary sedimentation, activated sludge process with aeration for BOD removal, secondary sedimentation, and tertiary sedimentation with chemical addition for polishing before discharge to Mjøsa lake.	Ca. 4,000 TDS per year (or 13,500 tonnes per year at 30% DS). Cambi Thermal hydrolysis and mesophilic anaerobic digestion, followed by belt filter press for dewatering to 30% DS.
NRA (RA-2)	Lørenskog, Rælingen, Skedsmo	Ca. 15 mill. m^3/year (97,000 p.e.). Screening, primary sedimentation, Moving Bed Biofilm processes for BOD removal, nitrification and denitrification with external carbon source. Secondary sedimentation with addition of chemicals using DOSCON controlling system.	Ca. 20,000 tonnes final dewatered (ca 30% DS) sludge with lime stabilization.
Sandefjord RA	Sandefjord	0.78 mill m^3/year (44,000 p.e.) Screening, sand trap, chemical enhanced primary sedimentation for particle and partial BOD removal.	Ca. 3000-4000 tonnes dewatered sludge at 25-30% DS per year. Dewatering by centrifuges.
TAU	Tønsberg	Ca. 14 mill. m^3/year (75,000 p.e.). Chemical enhanced primary sedimentation for particle and partial BOD removal. A new biological treatment unit is under construction and will be in operation in 2014.	Ca. 15,000 tonnes final dewatered sludge per year. Centrifuges and lime stabilization, storage before agriculture use.
VEAS	Oslo, Asker, Bærum, Røyken, Nesodden	100-110 mill. m^3/year (700,000 p.e.), Screening, sand trap with sand washing, primary sedimentation with chemical (polymer) precipitation, aerated biofilter using Leca media for nitrogen removal with post denitrification and addtion of carbon source.	Ca. 25,000 tonnes final dried sludge (ca 50% DS) per year. Two-stage mesophilic anaerobic digestion and low temperature frame-vacuum drying.

2.2 Sampling

All samples from the effluent of the eight WWTPs were collected during autumn 2010. When possible, samples were collected during weeks of dry weather conditions.

The established automatic sampling systems of the various WWTPs were used for collecting samples. All the operators of the WWTPs involved in sampling were certified, or in the process of being certified, according to NS-EN ISO/IEC 17025 which include sampling of wastewater. The quality system specify requirements of sampling equipment, sampling point, procedures for preparation before sampling, and procedures for collecting secondary samples, including cleaning of equipment and sample vessel, calculation of frequency and volume of sub-samples, testing of sampler, control of refrigerator, required data to be collected and registered.

The sampling equipment for primary sampling at all WWTPs consisted of a flow-proportional composite water sampler with an input PLS-signal from the flow meter and a 25 litre polyethylene vessel placed in a refrigerator at 4°C. A volume of 20-25 litres was collected daily by calculating required frequency and volume of sub-samples based on expected flow rate per hour. Minimum frequency is 3 sub-samples per hour, and minimum volume is 50 ml.

From the primary sample, a 3 litre secondary sample was prepared by collecting a 430 ml daily sub-sample for 7 days. The 3 litre annealed glass vessel was stored in the dark at 4°C during the sampling period. Before subtracting the 430 ml sub sample from the primary sample, the primary sample was homogenised by manual stirring with a special stirring device according to procedures of the quality system. The 3 litre vessel was labelled and packed in an insulated box with ice and sent by overnight transport to Norwegian Institute for Air Research (NILU) in Oslo for analysis. Sample processing was started as soon as possible and usually within one week. The daily amount of treated water was recorded by the treatment plants.

A 500 ml grab sample was collected on the last day of water sampling from a batch of final dewatered sludge. The sample was collected using a clean glass spoon and transferred to pre-prepared plastic containers. The sludge samples were transported on ice to NILU in the same box as the water samples.

2.3 Chemical analyses

All chemical analyses were performed at the laboratories of the Norwegian Institute for Air Research (NILU) in Oslo and the University of Umeå, Sweden. The NILU laboratory is accredited by the Norwegian Accreditation as a testing laboratory, according to the requirements of NS-EN ISO/IEC 17025. Analytical standards are also certified by the participation in international calibration tests. The procedures are described below. All sample processing started with the bulk sample. All sludge concentrations are presented as dry weight.

2.3.1 Perfluorinated compounds

Sample preparation and Clean-up
Prior to extraction, samples were homogenised and an internal standard added to all samples. Sludge samples were treated with acid and neutralized again prior to extraction with acetonitrile in an ultrasonic bath. Effluent samples were filtered in order to collect the

particulates. Particulates were weighed and treated the same as the sludge samples. The water phase was extracted with an HLBplus cartridge which was eluted with methanol. Following centrifugation, the supernatant solution was concentrated and 25 mg ENVI-Carb added plus 50 µl glacial acetic acid. Following additional centrifugation, an aliquot of the solution was transferred in an autoinjector vial and a recovery standard (3,5-bis(trifluoromethyl)phenyl acetic acid (20 µl of a 0.25 ng/µl solution in methanol) and aqueous ammonium acetate added. For quantification ^{13}C labelled PFOS and PFOA were used as internal standards.

Instrumental analysis Ionic PFC
Aliquots (50 µl) were injected automatically onto a HPLC (Agilent 1100; Agilent Technologies, Palo Alto, CA) coupled to ESI time-of-flight-high-resolution MS in the negative ion mode (LCT, Micromass, Manchester, England). Compounds were separated on an ACE C18 column (150 x 2.1 mm, 3 µm particle size) (ACT, Aberdeen, U.K.) using a gradient of 200 µl/mins methanol and water (both with 2 mM NH_4OAc). The initial mobile phase was 50:50 methanol/water, followed by a 5 mins linear increase to 85:15, a 5 mins hold at 85:15, a 0.5 mins linear increase to 99:1, and isocratic hold until reverting to initial conditions after 15 mins. Full scan (m/z 165-720) high resolution mass spectra were monitored throughout the chromatograms.

Instrumental analysis Volatile- PFC
The extracts were analysed on a 6890 gas chromatograph (Agilent Technologies Inc., Santa Clara, CA, USA) coupled to 5975C-MSD (Agilent Technologies Inc., Santa Clara, CA,USA**)**. The injection was splitless at 200 °C. Separation was achieved on a SupelcoWax 10 column (60 m x 0.25 mm i.d. x 0.25 µm film thickness) held at 50 °C for 1 mins, ramped 5 °C/mins to 70 °C, 10 °C/mins to 130 °C, 10 °C/mins to 225 °C, then ramped at 20 °C/mins to 275 °C held for 10 mins. The transfer line was held at 250 °C. Helium was used as carrier gas at a flow rate of 1.5 ml/mins and methane as reagent gas in positive chemical ionisation (PCI) mode and ion source temperature of 250 °C and the quadrupole at 100 °C. Each compound was identified by using the base peaks (quantitative ions) and confirmed by additional ions. Quantification was performed using an internal standard.

2.3.2 PBDE/HBCD/CP

Sample preparation and Clean-up
Sludge samples were dried at 30 °C and soxhlet extracted with n-hexane/diethylether (9/1) for 8 hours. Effluent samples were extracted by liquid-liquid extraction with cyclohexane. Internal standards were added before extraction. The extract, dissolved in hexane, was first treated between one and four times with concentrated sulfuric acid to remove the majority of matrix interference followed by chromatography on a silica column using n-hexane/diethylether (9/1) as the mobile phase. Before instrumental analysis the extract was reduced in volume to between 100 and 1000 µl and recovery standards added.

Instrumental analysis

PBDE
The extracts were analyzed on a 7890A gas chromatograph (Agilent Technologies, Inc. Santa Clara, CA, USA) coupled to an AutoSpec high resolution mass spectrometer (Waters, Milford, MA, USA). The injection mode was pulsed splitless at 300 °C. The fused silica capillary column (DB-5HT 15 m x 0.25 mm i.d. x 0.10 µm film thickness, Agilent) was held at 100 °C for 2.5 mins, ramped 13.2 °C/mins to 220 °C, then 7 °C/mins to 280 °C, 50 °C/mins to 320 °C and isothermal for 10 mins. The transfer line was held at 290 °C. Helium was used

as the carrier gas. The detector was used in the SIM mode with an electron ionisation energy of 35 eV. The analytes were identified by their characteristic retention times and two daughter ions. The reported analyte concentrations were calculated according to the respective internal standard.

SCCP/MCCP
The extracts were analyzed on a 7890A gas chromatograph (Agilent Technologies, Inc. Santa Clara, CA, USA) coupled to an AutoSpec high resolution mass spectrometer (Waters, Milford, MA, USA). The GC was operated in constant flow mode at 1 ml/mins (He). The injection was performed using pulsed splitless at 260 °C. The fused silica capillary column (Restek Rxi5ms 15 m x 0.25 mm i.d. x 0.25 µm film thickness) was held at 90 °C, ramped to 245 °C by 20 °C/mins, then to 300 °C by 50 °C/mins, holding that temperature for 3 mins. The MS was operated in electron capture negative ion (ECNI) mode with argon at a pressure of $2x10^{-5}$ mbar as the reagent gas. To be able to monitor the $[M\text{-}Cl]^-$ ions for each formula group of interest, the MS experiment was divided into five time windows. The quantification were performed according to the method described by Tomy et al. (1997).

HBCD
The methanol extracts were analyzed for α-, β- and γ-HBCD using a Waters 2690 HPLC coupled to a single quadrupole Micromass z-spray mass detector (ZMD) in electro spray negative mode (ESI−) as previously described in detail. The HBCDD diastereomers were separated on a reversed-phase C18-column from Atlantis (150 mm, 2.1 mm i.d., 3.0 µm particle size) employing a trinary gradient of methanol (A), acetonitrile (B) and water (C) as eluent. The initial mobile-phase (time zero) composition of 30% A, 10% B and 60% C was changed to 83% A, 15% B and 2% C over 12 mins.

2.3.3 TBBPA/BPA

Sample preparation and Clean-up
Sludge samples were homogenised along with a drying agent, internal standards added and the samples extracted using methyl-tert-butyl ether (MTBE). The extracts were cleaned initially on a silica column and then on an Isolute M-M column. Internal standards were added to the effluent samples which were extracted on Oasis HLB cartridges. The effluent extracts were cleaned on Isolute M-M columns.

Instrumental analysis
TBBPA and BPA were analysed by UPLC-HRMS. Reversed phase separation was performed on a Waters Acquity UPLC using a C18 column and acetonitrile and purified water as the mobile phase. The analytes were detected in high resolution on a Waters LCT Premier Time-of-Flight mass spectrometer (TOF-MS) using atmospheric pressure chemical ionisation in negative mode. Quantification was performed using ^{13}C-labelled internal standards.

2.3.4 PFRs

Sample preparation and Clean-up
All samples were fortified with fully deuterated TBP internal standard and extracted in the same way as described in chapter 2.3.2 for PBDE. The effluent extracts were concentrated to approximately 1 ml by rotary evaporation, an excess of toluene was added, and the volume was reduced to 1 ml. This volume was quantitatively transferred to conical GC vials and the volume was further reduced to 100 µl under a gentle stream of nitrogen. The sludge extracts were evaporated to approximately 0.5 ml and 100 µl was then injected onto a 1.5×40 cm gel permeation column packed with Bio-beads SX-3 (Bio-Rad Laboratories, Hercules, CA, USA)

which was eluted with cyclohexane:ethyl acetate (3:1) at a flow rate of 2 ml/mins. The first fraction (0-20 mins) contained most of the interfering macromolecules (lipids, humic matter, etc) and was discarded. The second fraction (20-60 mins) was collected, dried over sodium sulfate, and evaporated to 1 ml. The solvent was then exchanged to toluene and transferred to GC vials, as described for the water samples.

Instrumental analysis
The samples were analyzed using a GC-HRMS high-resolution system consisting of an Agilent Technologies 6890 GC equipped with a CTC Analytics autosampler and coupled to a Micromass AutoSpec-Ultima mass spectrometer tuned to a resolution of 8000. 1 µl of each sample was injected into the GC, which was operated in splitless mode (2 mins splitless time). The injector temperature was set to 250 °C and gas chromatographic separation was carried out using a DB-5 fused silica capillary column (length 30 m, i.d. 0.25 mm, film thickness 0.25 µm) from J&W Scientific (Folsom, CA). The GC oven was initially held at 80 °C for 4 mins, increased to 190 °C at a rate of 15 °C/mins, and then at 10 °C/mins to the final temperature, 310 °C, which was maintained for 4 mins. Helium was used as a carrier gas at a flow rate of 1.3 ml/mins. The MS was operated in selected ion monitoring (SIM) mode with electron ionization at 36.5 eV. The OPs were identified by comparing mass fragment ratios and retention times of sample components and reference standards, and quantified using the internal standard technique, which automatically corrects the data for losses during sample work-up and analysis.

2.3.5 Musk compounds

Sample preparation and Clean-up
Sludge samples were dried and extracted by sonication with ethylacetate. Interfering matrix compounds were removed by gel permeation chromatography (GPC) followed by cleanup on silica. Effluent samples were extracted with diethylether/hexan and dichloromethane followed by cleanup on silica.

Instrumental analysis
The extracts were analysed on a 6890 N gas chromatograph (Agilent Technologies, Inc. Santa Clara, CA, USA) coupled to an AutoSpec high resolution mass spectrometer (Waters, Milford, MA, USA). The splitless injection was made at 260 °C. The fused silica capillary column (DB-5MS 30 m x 0.25 mm i.d. x 0.25 µm film thickness, Agilent) was held at 90 °C for 2 mins, ramped 6 °C/mins to 260 °C, then 30 °C/mins to 280 °C and isothermal for 10 mins. The transfer line was held at 280 °C. Helium was used as carrier gas. The detector was used in SIM mode with electron ionisation at energy of 35 eV. HHCB and AHTN were identified using the base peaks (quantification ions) and confirmed by additional ions. Quantifications were performed using the internal standard technique.

2.3.6 Ag

Sample preparation
Dried sludge samples were extracted in aqua regia using a microwave oven. Supra pure nitric acid was added to the effluent samples to give a final concentration of 1 (v/v)%. ^{185}Re was added to all standards, blanks and samples as internal standard.

Instrumental analysis
The concentration of silver was determined using a high resolution plasma mass spectrometer (ELEMENT2, Thermo Inc, Germany).The plasma generator was operated at 1250 W. The silver isotope ^{107}Ag was determined in low resolution mode (R~300).The flow rate of plasma

gas and auxiliary gas used were 15 l/mins and 1 l/mins, respectively. Nebulizer gas flow was ca. 0.9 l/mins and optimised daily. The spray chamber assembly consisted of a Meinhard concentric nebulizer and a glass spray chamber of type Scott. Nickel-tipped cones were used for both sampler and skimmer.

The data processing and instrument control were performed using the ELEMENT software.

2.3.7 Quality Control

The following quality criteria were used to ensure correct identification and quantification of the target compound: (a) the retention time should match those of the standard compounds within ± 0.05 mins, (b) the intensity ratios of the selected ions (target- and qualifier-ions) are within ± 15 % of expected/theoretical value (c) the signal-to-noise ratios are greater than 3:1.

An analytical method blank was included for each sample batch analysed to assess background interferences and possible contamination of the samples. Internal standard was added to the sample at the start of the working-up procedure of the sample. The internal standard has similar chemical and physical properties to the compounds to be analyzed. Where available, isotope labelled internal standards were used.

2.3.8 Uncertainties and variations

When performing environmental screening or monitoring, all steps in the study are generating some degree of uncertainty. These steps include design of the study, selection of sampling sites and sampling frequency, time of sampling, performing of sampling and collecting of sub-samples from sampling vessels, transport and storage of samples, chemical analysis and data handling. To quantitatively estimate the contribution of all these steps is an extremely difficult task, if not impossible. However, we will discuss the relevance of the different contributing factors in a qualitative way.

Study design

The concentrations of the different compounds of interest in effluent water from WWTPs vary considerably due to variations in factors outside the treatment plant, including water flow, precipitation, temperature and local emission patterns from households and industry. Internal factors like type of wastewater treatment process, chemicals used in the treatment process and performance of the treatment process during sampling will also influence the concentration level of compounds in the effluent. Many of the compounds in this study are rather lipophilic in character (octanol-water partition coefficient, Pow > 10^4), and will therefore tend to associate with the particulate matter, and be well removed in treatment plants applying chemical treatment only (Arendal RA, FREVAR, Sandefjord RA and TAU). These compounds will normally be found in the sludge samples at elevated concentrations. On the other side, the lack of a biological process step may have significant influence on the overall removal of some of the more biodegradable compounds.

To minimize the influence of storm water, samples were attempted collected during dry weather weeks. However, this was not possible to fulfil. In most periods, the flow exceeded the dry weather flow, indicating influence of storm water.

Sampling and sample handling

The factors which influence sampling uncertainty are analyte loss due to adsorption to sample containers, sampling procedures and particulate content of the water, contamination (for some compounds), and degradation during transport and storage. An important factor, especially for a WWTP study, is the collection of representative and homogenous sub-samples. Particles will settle during storage in sample containers. Proper stirring and re-suspension during withdrawal of sub-samples is a challenge and may represent a source of error, in particular for hydrophobic compounds attached to particles. Sampling procedures used in the current study to minimize uncertainties are described in Chapter 2.2.

The sludge samples analysed in this study were grab samples collected once a week from produced batches of dry sludge. Possible variations in sludge quality and characteristics due to external factors and internal process factors should be taken into account. Also the analytic procedures may have introduced uncertainties. Sludge, and to a lesser degree effluent, are very heterogeneous matrices. In theory it is possible to prepare a sludge sample in a way that identical subsamples can be taken. However, this would require several days of specialized treatment for each sample. A complete homogenisation of all sludge samples, aiming for identical subsamples, would be a very costly process. In addition extensive sample handling can have negative impact on the integrity of the true concentration of the analytes, either due to contamination or due to loss by adsorption to the homogenisation instruments in use. Therefore, normal practice is to find an acceptable balance between homogeneity and limited sample pre-treatment.

Chemical analysis

The uncertainty of the chemical analysis is governed by loss during extraction and clean-up, interference from other compounds, accuracy of analytical standards, instrumental parameters, and contamination. A normal approach to estimate and quantify these factors is the participation in a laboratory intercalibration. However, intercalibration studies are often not available for new and emerging contaminants. The uncertainty is expected to be larger for compounds which are analysed infrequently than for compounds which are analyzed routinely. That means that most compounds will probably have analytical uncertainties in the range of 20 to 40 %. For all analytes we consider the analytical uncertainty as fit-for-purpose (that means adequate for a screening study).

3. Results and discussion

3.1 Bisphenol A and tetrabromobisphenol A

Bisphenol A (BPA) was detected in all bar one of the WWTP effluent samples collected at concentrations of between < 35 and 3,400 ng/L with a median of 327 ng/L (Table 3). The levels detected appear to be higher in effluent samples collected from the WWTPs at Arendal and TAU than the other six WWTPs included in the study (Figure 2).

BPA concentrations in sludge ranged from 68 to 2,980 ng/g (median= 425 ng/g) with low concentrations being observed in the samples collected from the WWTPs located at Bekkelaget, HIAS, RA-2 and VEAS. Tetrabromobisphenol A (TBBPA) was not detected at a concentration higher than the LOD (ca. < 15 ng/L) in any of the effluent samples collected. The concentration of TBBPA in sludge ranged from below the LOD to 129 ng/g. TBBPA was only detected in 65% of the samples collected (Figure 3). Very little temporal differences were observed for the two compounds during the sampling campaign period.

Table 3. Summary occurrence data for bisphenol A (BPA) and tetrabromobisphenol A (TBBPA)

	Sludge (ng/g)			Effluent (ng/L)		
	Median	Min	Max	Median	Min	Max
BPA	425	68	2,980	327	< 35	3,400
TBBPA	5.6	<4.5	129	<LoD	<LoD	<LoD

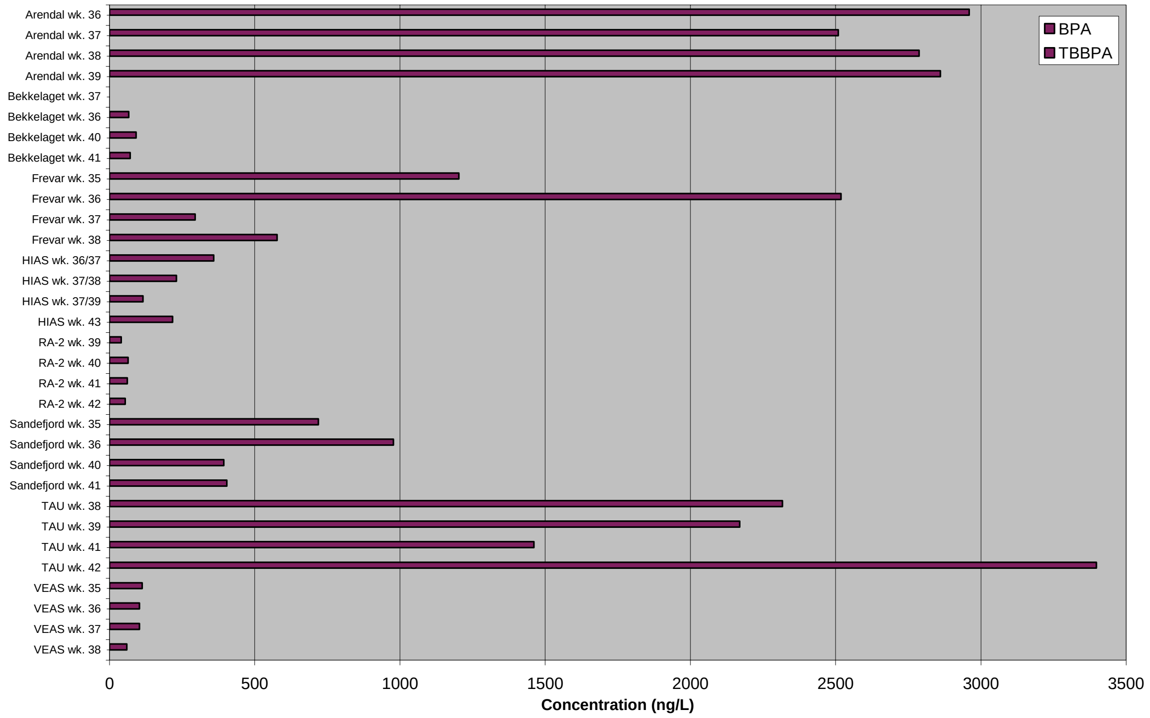

Figure 2. Occurrence of bisphenol A (BPA) and tetrabromobisphenol A (TBBPA) in WWTP effluent.

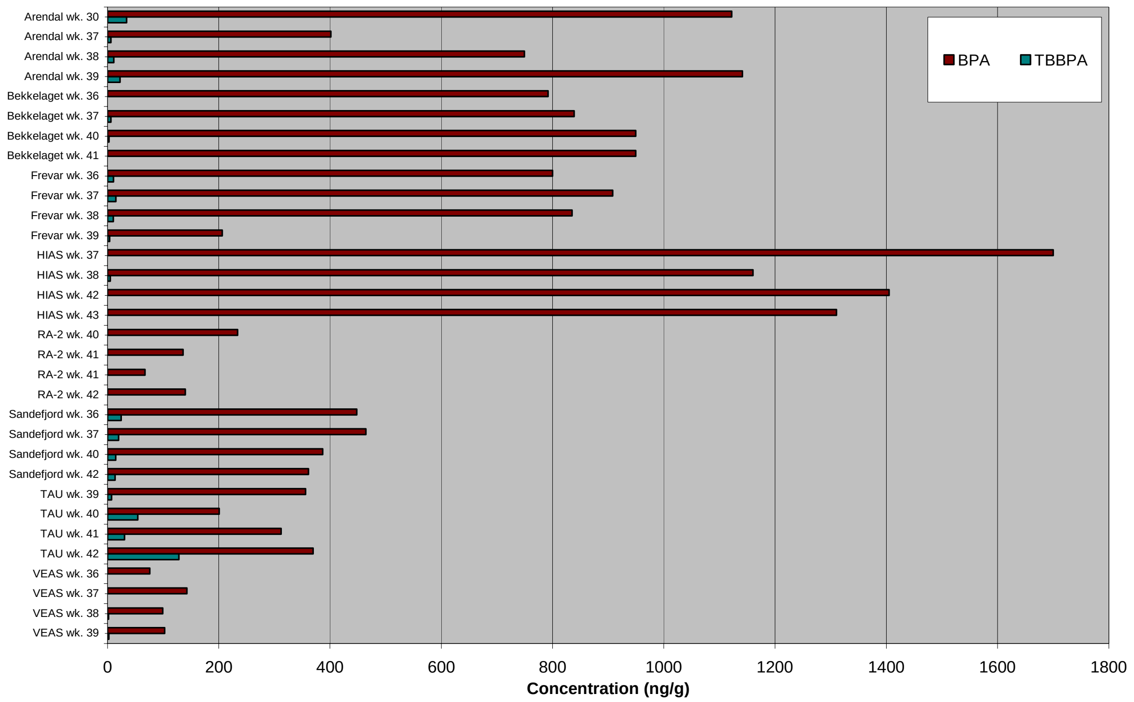

Figure 3. Occurrence of bisphenol A (BPA) and tetrabromobisphenol A (TBBPA) in sewage sludge.

The concentrations of BPA determined in effluents are very similar to those previously reported for BPA in Norwegian WWTP effluent (Fjeld, 2008; Blytt, 2010). For example, a Klif commissioned study in 2009 of four municipal WWTPs showed a median BPA concentration of 996 ng/L with a range of 33 to 5,360 ng/L (Blytt, 2010). BPA has previously been detected in Norwegian sewage sludge of up to 3,200 ng/g with a mean of 1,400 ng/g (Nedland and Paulsrud, 2006). A number of other studies in Norway have reported concentrations between 80 and 2,100 ng/g in samples collected between 2003 and 2007 (Blytt, 2010).

TBBPA is hydrophobic and with previously reported effluent levels being associated with the particulate load. It is therefore not surprising that the concentrations determined in the WWTP effluents collected were below the LOD. This is similar to previously reported studies from Norway and the rest of Europe where very low concentrations have been reported in WWTP effluents (Fjeld, 2008). The concentrations of TBBPA in the sludge were similar to those previously reported in Norway and Sweden (Sellström and Jansson, 1995; Sellström *et al.*, 1999; Oberg *et al.*, 2002; Fjeld, 2008).

The risks posed by the concentrations determined in this study are best described by comparing the results with published PNEC (Table 4) and Norwegian standards for the classification of surface waters and sediments.

Table 4. Predicted no-effect concentration (PNEC) for BPA and TBBPA.

Chemical	Compartment	PNEC	Source
BPA	Fresh water	1,500 ng/L	(EU, 2008)
	Marine water	150 ng/L	
	WWTP (effluent)	320 mg/L	
	Soil	6.3 ng/g (dry weight)	
TBBPA	Fresh water	1200 ng/L	(Swedish Chemical Inspectorate, 2006)
	Marine water	250 ng/L	
	Freshwater sediment	540 ng/g (wet weight)	
	Marine sediment	54 ng/g (wet weight)	
	WWTP (sludge)	$\geq 1.5 \times 10^6$ ng/L	
	Soil	300 ng/g	(Sverdrup *et al.*, 2006)

It is clear from comparing the measured data with the $\text{PNEC}_{\text{WWTP}}$ that there is little risk to the WWTP microorganisms from the levels of BPA measured in the final effluent. The effluent concentrations in samples collected from a number of the WWTPs are greater than the $\text{PNEC}_{\text{Water}}$ and many are greater than the $\text{PNEC}_{\text{Marine water}}$. For example the effluents of both Arendal WWTP and TAU typically contained concentrations ten times higher than the $\text{PNEC}_{\text{Marine water,}}$ indicating that rapid dispersion and dilution of the effluent is required in these two areas in order to avoid risks associated with the occurrence of BPA in the effluent. Should these levels be found in receiving waters then they would be classified as ‘moderate’ according to the Norwegian classification guideline (1,600-11,000 ng/L) meaning that chronic effects would be expected with long term exposure (Bakke *et al.*, 2007). The risk to the aquatic environment from releases of TBBPA from WWTP appears to be low since the levels detected were all below the LOD and the concentrations measured in sludge were all below the $\text{PNEC}_{\text{WWTP (sludge)}}$ value. The levels of TBBPA in sludge were all below available sediment PNEC indicating that there is very little risk to sediment dwelling organisms. With no regulatory PNEC available for TBBPA comparison with a published $\text{PNEC}_{\text{Soil}}$ of 300 ng/g suggests that there is very little risk to soil dwelling organisms from the levels determined in sludge (Sverdrup *et al.*, 2006).

The mass transport of BPA in the effluents from the various treatment plants is given in table 5. It should be noted that smaller plants, such as TAU and Arendal, have substantially higher effluent loads than larger plants like VEAS and Bekkelaget.

Table 5. Mass transport of bisphenol A (BPA) (mg/day) from the WWTPs included in study. Tetrabromobisphenol A (TBBPA) was not detected in any of the the effluents.

	BPA (mg/day)							
	Bekkelaget	VEAS	Sandefjord	TAU	Arendal	HIAS	FREVAR	RA2
Min	6617	21105	10649	66012	30264	2427	16887	1659
Median	10656	25584	16974	107427	43100	5473	34187	2120
Max	14695	30306	21398	125855	61184	7894	115128	3734

3.2 Polybrominated diphenyl ethers (PBDE)

Deca-BDE (BDE-209) was the dominant congener in both sludge and effluent samples with significantly higher concentrations detected in sludge samples (Figure 4). The median concentrations for BDE-209 were 335 ng/g and 2.12 ng/L in sludge and effluent respectively (Table 6) with a maximum concentration of 1.71 µg/g in sludge from Arendal. BDE-209 concentrations were higher in all Arendal sludge samples than any of the other wastewater treatment plants. The lowest sludge concentrations were measured at RA-2 with a concentrations range of 43.9 – 81.8 ng/g. In effluent from RA-2 however, the median concentration was 3.25 ng/L which is higher than all other WWTP with the exception of Arendal. The sum-PBDE concentration measured in sludge varies for different WWTP. Arendal had the highest concentrations measured with a mean of 1494 ng/kg and Bekkelaget, FREVAR, HIAS, Sanderfjord, TAU and VEAS have median sum-BDE concentrations in the range 172-427 ng/g. The median sum-BDE concentration measured at RA-2 was 68 ng/g.

Table 6. Summary occurrence data for polybrominated diphenyl ethers (PBDEs).

	Sludge (ng/g)			Effluent (ng/L)		
	Median	Min	Max	Median	Min	Max
2,4,4'-TriBDE (BDE-28)	0.21	0.05	0.38	0.02	<LoD	0.08
2,2',4,4'-TetBDE (BDE-47)	11.4	2.30	18.5	0.49	0.29	1.21
2,3',4,4'-TetBDE (BDE-66)	<LoD	<LoD	0.32	<LoD	<LoD	0.04
2,2',4,5' + 2,3',4',6-TetBDE (BDE-49+71)	0.65	0.08	4.72	<LoD	<LoD	0.43
3,3',4,4'-TetBDE (BDE-77)	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
2,2',3,4,4'-PenBDE (BDE-85)	0.25	<LoD	1.41	<LoD	<LoD	0.02
2,2',4,4',5-PenBDE (BDE-99)	12.7	2.50	20.6	0.20	0.11	0.91
2,2',4,4',6-PenBDE (BDE-100)	1.74	0.34	2.94	0.06	0.02	0.23
2,3',4,4',6-PenBDE (BDE-119)	<LoD	<LoD	0.42	<LoD	<LoD	<LoD
2,2',3,4,4',5'-HexBDE (BDE-138)	0.21	<LoD	0.39	<LoD	<LoD	0.03
2,2',4,4',5,5'-HexBDE (BDE-153)	1.75	0.27	2.62	<LoD	<LoD	0.20
2,2',4,4',5,6'-HexBDE (BDE-154)	0.70	0.15	1.11	<LoD	<LoD	0.10
2,2',3,4,4',5',6-HepBDE (BDE-183)	1.18	0.11	2.81	<LoD	<LoD	0.19
2,2',3,3',4,4',5,6'-OctBDE (BDE-196)	3.61	<LoD	21.9	<LoD	<LoD	<LoD
2,2',3,3',4,4',5,5',6-NonBDE (BDE-206)	7.60	1.63	25.5	<LoD	<LoD	0.37
DecaBDE (BDE-209)	277	43.9	1 710	1.35	<LoD	11.2
Sum	335	51.7	1 772	2.12	0.43	12.3

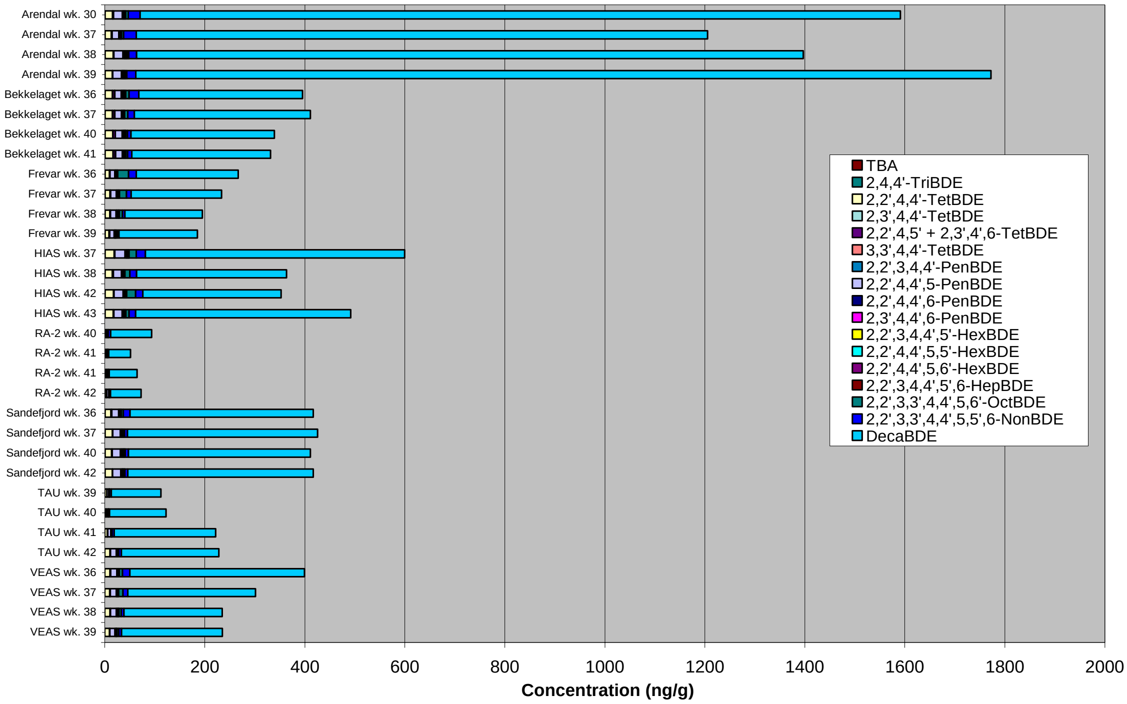

Figure 4. Occurrence data for polybrominated diphenyl ethers (PBDEs) in sludge.

The congener distribution in sludge shows the same distribution pattern as has been frequently reported elsewhere in the literature with the highest concentrations measured relating to BDE-209, 154, 153, 100, 99 and 47.

In the effluent samples, the 2 penta congeners, BDE-99 and BDE-100 and the tetra congener, BDE-47, were the most frequently detected compounds, with the exception of BDE-209. The median concentration for BDE-47 in effluent was 0.49 ng/L and the maximum was 1.21 ng/L. BDE-49 and BDE-71 were also detected in effluent but could not be separated chromatographically therefore the results are summed, although it is likely that the majority is BDE-49 as the BDE-71 congener is less frequently detected in environmental samples.

The congener distribution in effluent shows a slightly different distribution pattern to that reported in the literature (Figure 5). The prevalence of BDE-209 is unpredicted as due to its high Log K_{ow} value it is more likely to be associated with the solid phase. Its high occurrence in this sampling campaign is most likely to be related to the suspended matter in the effluent where a high binding rate would be expected. The occurrence of the tetra congener, BDE-47, and the penta congeners, BDE-99 and 100 follows results previously reported in the literature and is as expected based on Log K_{ow} values and usage patterns.

The results for effluent and sludge in this study are comparable to those from a previous Norwegain study focussing on 3 WWTPs around Lake Mjøsa (Fjeld, 2008) (Table 7).

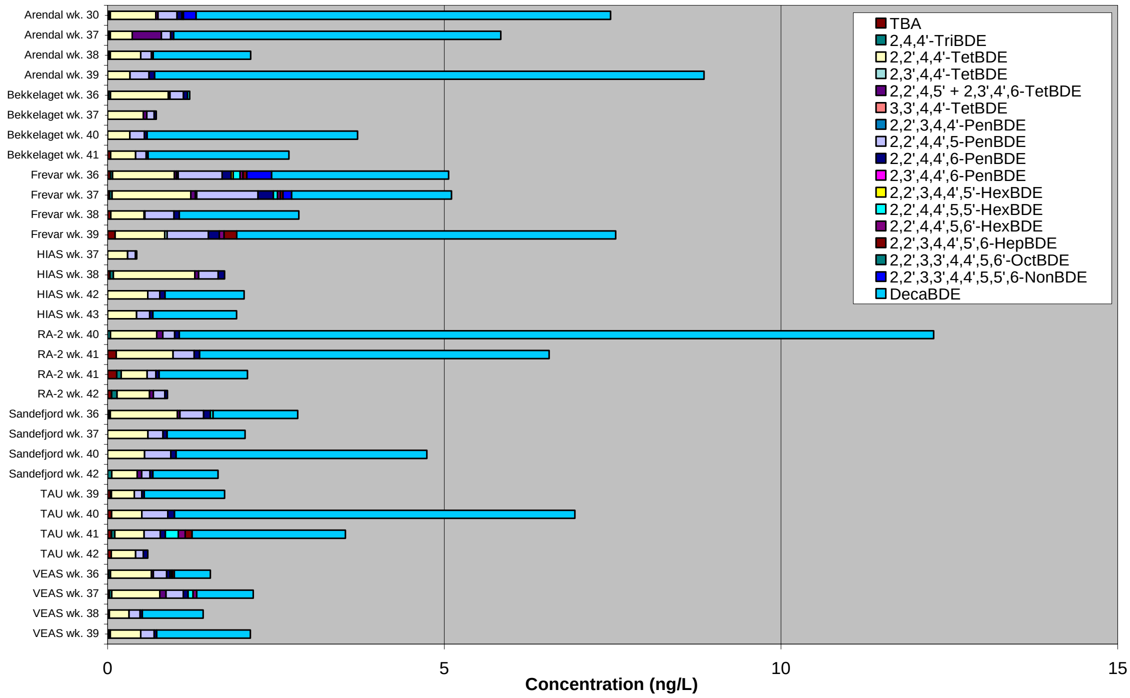

Figure 5. Occurrence data for polybrominated diphenylethers (PBDEs) in WWTP effluent.

Table 7. Previously reported occurrence data for PBDEs in Norwegian WWTP effluent and sludge (Fjeld, 2008).

	Penta-BDE	Octa-BDE	Deca-BDE
Effluent concentrations (ng/L)	0.58-2.69	0.18-0.37	0.91-1.89
Sludge concentrations (ng/g)	76.3-489.4	6.4-23	217-380

There are insufficient data available to calculate a $\mathrm{PNEC}_{\mathrm{WWTP}}$ and it is not known what effect the commercial penta-BDE formulation has on the microorganisms involved in wastewater treatment processes. None of the maximum penta-BDE concentrations exceed the $\mathrm{PNEC}_{\mathrm{water}}$ values (Table 8) and when the dilution factor in the receiving waters is taken into account, it is unlikely that values will be exceeded in receiving waters either.

Table 8. Predicted no-effect concentrations (PNEC) for selected polybrominated diphenylethers (PBDEs).

Chemical	Compartment	PNEC	Source
PentaBDE	Water	530 ng/L	(EU, 2001)
	Sediment	1,550 ng/g	
	Soil	5,910 ng/g (wet weight)	
HexaBDE	Water	530 ng/L	(Møskeland, 2010)
	Sediment	≥ 7,000 ng/g	
	Soil	> 1,200 ng/g (wet weight)	(EU, 2003)
Commercial OctaBDE	Water	≥ 200 ng/L	(Møskeland, 2010)
	Sediment	≥ 127 000 ng/g	
	Soil	> 20,900 ng/g (wet weight)	(EU, 2003)
DecaBDE	Water	≥ 200 ng/L	(EU, 2002)
	Sediment	≥ 127,000 ng/g	
	Soil	> 5,600 ng/g wet wt	

The PNEC values for commercial deca-BDE are difficult to calculate due to a paucity of data. Deca-BDE has a very low water solubility (<100 ng/L) and low toxicity was observed in fish and algae up to the maximum water solubility and this lack of effects data make it difficult to calculate PNEC values. A minimum $PNEC_{water}$ >100 ng/L based on EC_{50} values for algae tests or a minsimum $PNEC_{water}$ >200 ng/L based on octa-BDE toxicity data have been calculated (EU, 2002). Effluent samples were lower than $PNEC_{WWTP}$ values for all samples. Adverse effects are a possibility when PNEC values are exceeded.

Median values for the most important PBDE congeners discharged from 3 WWTP on Lake Mjøsa, Norway (Fjeld, 2008) are shown in table 8. In other studies, the PBDE congeners with highest abundance in sludge were BDE-47, BDE-99 and BDE-209 as has been seen in this study. In effluent BDE-209 represented just 6% of the load while the tetra, penta and hexa congeners, BDE-47, 99, 100, 153 and 154 make up 88% of the effluent load. The total load of PBDEs discharged to the San Francisco Estuary was 0.9 kg/year (North, 2004). Another WWTP on a US river had a comparable mass loading of approximately 0.7 kg/year (Song *et al.*, 2006). Another WWTP on a US river had a comparable mass loading of approximately 0.7 kg/year (Shin *et al.*, 2007) and sludge concentrations of 400-2 000 ng/g (Anderson and Macrae, 2006) and 400-2 000 ng/g (Kupper, 2008). Concentration of 310, 95 and 17 µg/kg for deca, penta and octa congeners respectively were measured in a Swiss study (Langford *et al.*, 2005). .

Fish samples collected downstream of WWTP demonstrated higher PBDE concentrations than those collected upstream (La Guardia *et al.*, 2007).

The mass transport in mg/day of PBDEs in the effluents from the various treatment plants is given in table 9. If it is assumed that the load calculated from our measurements is representative for the total year, the total load of PBDEs discharged from VEAS is approximately 0.1 kg/year. FREVAR and Sandefjord have the highest effluent loads relative to population equivalents. Neither of these WWTP use biological treatment processes. WWTP with biological treatment showed lower relative PBDE loadings. Debromination of BDE-209 to produce less brominated congeners has been observed under aerobic (Langford *et al.*, 2007) and anaerobic conditions (Andreas C. Gerecke, 2006)

Detailed terrestrial risk assessment of the different application and soil scenarios are outside the scope of this report but none of the median concentrations of any of the PBDE congeners are sufficiently low to be above the $PNEC_{Soil}$ following moderate dilution upon application to land. Comparing the median sumPBDE measured data to the lowest $PNEC_{Soil}$ value indicates there is little risk to soil dwelling organisms.

Table 9. Mass transport of polybrominated diphenyl ethers PBDE (mg/day) from the WWTPs included in the study. Absent or single value indicates no or only one detected measurement.

		Brominated compounds PBDE (mg/day)							
		Bekke-laget	VEAS	Sandefjord	TAU	Arendal	HIAS	FREVAR	RA2
TBA	Min		4.0		1.8	0.3		1.3	2.2
	Median		6.3		2.1	0.3		2.2	5.2
	Max	1.6	7.5	0.4	3.9	0.4	0.9	4.0	7.7
2,4,4'-TriBDE	Min		7.0	0.5	0.7	0.2		1.4	1.8
	Median		7.7	1.1	2.1	0.2		1.6	2.5
	Max	3.0	8.3	1.7	3.4	0.4	1.2	1.7	2.8
2,2',4,4'-TetBDE	Min	53.0	87.0	9.9	13.5	3.9	6.6	26.2	14.7
	Median	53.2	155.8	17.1	15.3	7.0	12.6	32.3	22.6
	Max	91.2	162.2	26.2	29.6	10.0	28.8	53.6	49.4
2,2',4,5' + 2,3',4',6-TetBDE	Min	2.8	6.7	0.8		0.4		0.8	2.1
	Median	4.1	13.7	1.3		2.8		1.3	2.9
	Max	5.4	20.8	1.8		5.2	1.4	3.0	3.8
2,2',3,4,4'-PenBDE	Min							0.9	
	Median							1.0	
	Max							1.0	
2,2',4,4',5-PenBDE	Min	11.2	48.7	3.3	4.2	1.6	2.5	21.8	4.9
	Median	21.2	53.4	6.2	8.7	3.4	3.8	25.4	6.7
	Max	35.0	70.7	18.5	16.3	5.0	6.9	41.6	18.3
2,2',4,4',6-PenBDE	Min	3.0	9.6	1.1	1.8	0.6	0.4	4.6	1.4
	Median	6.3	13.5	1.7	2.7	0.8	1.7	5.6	2.3
	Max	6.6	15.9	3.7	5.1	1.4	2.2	10.4	4.7
2,2',4,4',5,5'-HexBDE	Min		5.9					2.8	
	Median		11.0					3.4	
	Max	3.2	16.2	0.8	13.4			4.1	
2,2',4,4',5,6'-HexBDE	Min		2.6					1.8	
	Median		7.7					1.9	
	Max		12.7		6.7	0.2		2.4	
2,2',3,4,4',5',6-HepBDE	Min							1.8	
	Median							2.3	
	Max		7.9		7.0			6.8	
2,2',3,3',4,4',5,5',6-NonBDE	Min							5.9	
	Median							10.3	
	Max					2.3		14.7	
DecaBDE	Min		133.7	25.4	59.6	31.7		101.6	50.1
	Median		227.0	26.0	155.0	66.6		106.4	304.1
	Max	502.8	497.5	177.4	180.9	143.6	24.7	200.4	462.2
Sum of PBDE	**Median**	**84.8**	**356.1**	**52.4**	**185.9**	**81.1**	**18.1**	**193.7**	**346.3**

2,3',4,4'-TetBDE and 2,2',3,4,4',5'-HexBDE were only detected in one sample at FREVAR, resulting in a mass transport of 1.3 mg/day for both compounds. Sum PBDE is calculated from median values.

3.3 Hexabromocyclododecane (HBCD)

All 3 stereoisomers were detected in sludge samples (Table 10; Figure 6). The highest median value was measured for γ-HBCD (4.51 ng/g) but the highest maximum value was 70.6 ng/g α-HBCD measured at Sandefjord. This sample collected in week 40 at Sandefjord also had the highest β- and γ-HBCD measurements (6.27 and 56.9 ng/g respectively).

None of the HBCD stereoisomers were detected in sludge from HIAS WWTP and low concentrations were measured at FREVAR.. The PNEC values calculated in the EU risk assessment (EU, 2008) use the sum of all 3 stereoisomers. Using the sum-HBCD data from this study, the $PNEC_{WWTP}$ 150,000 ng/L was not reached in any of the samples. The application of sludge to land poses little risk to the terrestrial environment with respect to HBCD as the median and maximum sludge concentrations are below the $PNEC_{soil}$.

Table 10. Summary occurrence data for hexabromocyclododecane (HBCD).

	Sludge (ng/g)			Effluent (ng/L)		
	Median	Min	Max	Median	Min	Max
α-HBCD	1.11	<LoD	70.6	<LoD	<LoD	1.01
β-HBCD	<LoD	<LoD	6.27	<LoD	<LoD	<LoD
γ-HBCD	4.51	<LoD	56.9	<LoD	<LoD	<LoD
Sum	7.70	<LoD	134	<LoD	<LoD	1.01

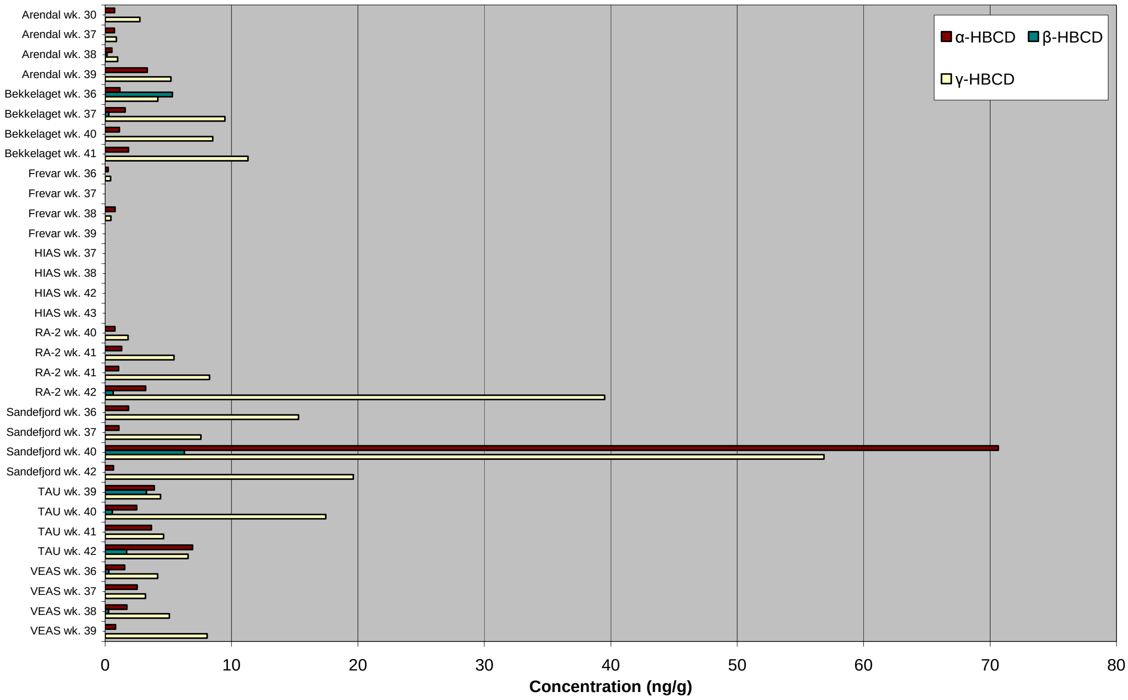

Figure 6. Occurrence of hexabromocyclododecane (HBCD) in sludge.

One effluent sample collected from TAU WWTP contained 1.01 ng/L α-HBCD which is less than the PNEC values for water or WWTP (Table 11) and all other effluent samples were <LoD meaning there is currently no risk posed by HBCD in WWTP.

Table 11. Predicted no-effect concentrations for hexabromocyclododecane (HBCD)

Chemical	Compartment	PNEC	Source
Total-HBCD	Water	310 ng/L for long term exposure 520 ng/L for short term intermittent exposure	(EU, 2008)
	Sediment	860 ng/g in a chronic test	
	Effluent	150,000 ng/L	
	Soil	5,900 ng/ (dry weight)	

In a previous Norwegian study (Fjeld, 2008) the median effluent concentrations at the 3 WWTPs studied were 0.57-1.03 ng/L and the maximum concentrations were 2.32-15.1 ng/L which is higher than the current study. In the 2005-2007 study the median concentration at HIAS was 0.73 ng/L compared to >LoD in this 2010 study. Sludge results were comparable between the 2 studies and the median range of sum-HBCD was 0.9-19.44 ng/g in 2005-2007, although in the current 2010 study none on the HBCD congeners were detected in HIAS compared to 0.47-2.53 ng/g in 2005-2007 (Fjeld, 2008).

Substantial degradation of HBCD has been observed in sewage sludge and in anaerobic conditions debromination was observed (John W. Davis, 2006). Elsewhere, HBCD was observed to enrich in sludge and a maximum total HBCD concentration of 9.1 mg/kg was measured (Steve morris, 2004). Another anaerobic sludge study showed that β and γ biodegraded more rapidly than α (Andreas C. Gerecke, 2006).

The mean concentration of the sum of HBCD congeners in a Swiss sludge survey was 149 µg/kg and a loading of 3.3 mg/cap/yr (Kupper, 2008). In a Swedish study, HBCD was below detection limits in digested sludge but was detected was detected in primary sludge (Remberger *et al.*, 2004) however this contradicts an earlier Swedish study where HBCD was detected in digested but not primary sludge (Nylund, 2002). HBCD was detected in one sludge sample collected from an activated sludge treatment plant. The concentrations were 175, 6 and 375 ng/g (d/w) for α, β and γ respectively. In another 6 samples collected, no HBCD was detected with the exception of 1 sample where 2 ng/g γ-HBCD was measured (Guerra, 2010).

No table for mass transport calculations were made for HBCD, since only one sample from one WWTP contained HBCD, the TAU WWTP. The measured concentration resulted in a mass transport of 30.7 mg/day.

3.4 Fluorinated Compounds

In all WWTPs, the main removal pathway for 6:2 FTS was through effluent with very little sorbed to sludge. It was measured in all but one of the effluent samples collected at a maxiumum concentration of 31.9 ng/L and median of 5.8 ng/L. 8:2 FTS was not detected in any sample of sludge or effluent (Table 12, Figure 7 and Figure 8).

As expected, PFOS was detected in all samples with effluent concentrations in the range 1.55 -42.1 ng/L and sludge concentrations in the range 2.61-9.6 ng/g. RA-2 had the highest effluent concentration with a median value of 29.8 ng/L compared to HIAS, where the lowest concentrations were measured with a median of 1.79 ng/L. PFOS concentrations in sludge were all comparable at each location, although Bekkelaget concentrations were higher than other locations with a median of 9.4 ng/g. None of the effluent concentrations were above the

$\text{PNEC}_{\text{WWTP}}$ value reported and are therefore deemed to pose no risk to the WWTPs microorganisms. The measured effluent concentrations were also all below the $\text{PNEC}_{\text{water}}$ concentrations (table 13) and are not expected to pose a risk through discharge to marine or freshwater systems. Comparing the measured data from this study with $\text{PNEC}_{\text{soil}}$ values, there is very little risk to soil dwelling organisms as measured values are all below the $\text{PNEC}_{\text{soil}}$ values reported (table 13). With no regulatory PNEC available for the fluorotelomer alcohol compounds, assessing the risk to the aquatic or terrestrial environment is difficult.

Table 12. Summary of occurrence data for selected fluorinated substances in WWTP effluent and sludge.

	Sludge (ng/g)			Effluent (ng/L)		
	Median	Min	Max	Median	Min	Max
6:2 FTS	<LoD	<LoD	1.10	22.7	<LoD	800
8:2 FTS	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
PFOS	4.53	2.61	9.60	6.15	1.55	42.1
PFOA	<LoD	<LoD	4.75	8.75	<LoD	20.2
4:2 FTOH	<LoD	<LoD	7.66	<LoD	<LoD	12.1
6:2 FTOH	1.38	<LoD	12.3	<LoD	<LoD	11.8
8:2 FTOH	8.49	2.25	147	1.68	<LoD	20.0
10:2 FTOH	6.88	<LoD	60.2	<LoD	<LoD	0.03
N-Et FOSA	<LoD	<LoD	7.69	<LoD	<LoD	8.97
N-Me FOSA	<LoD	<LoD	7.32	<LoD	<LoD	8.75
N-Me FOSE	4.68	<LoD	20.3	<LoD	<LoD	11.8
N-Et FOSE	2.23	<LoD	10.4	<LoD	<LoD	8.69

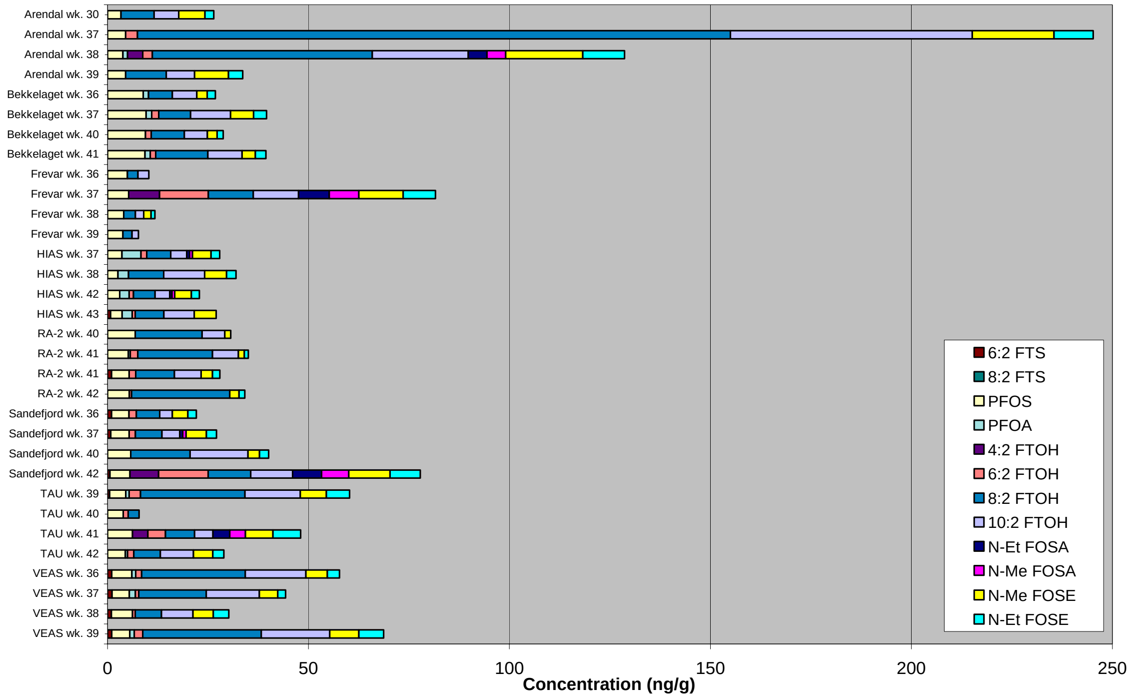

Figure 7. Occurrence of selected fluorinated substances in sludge

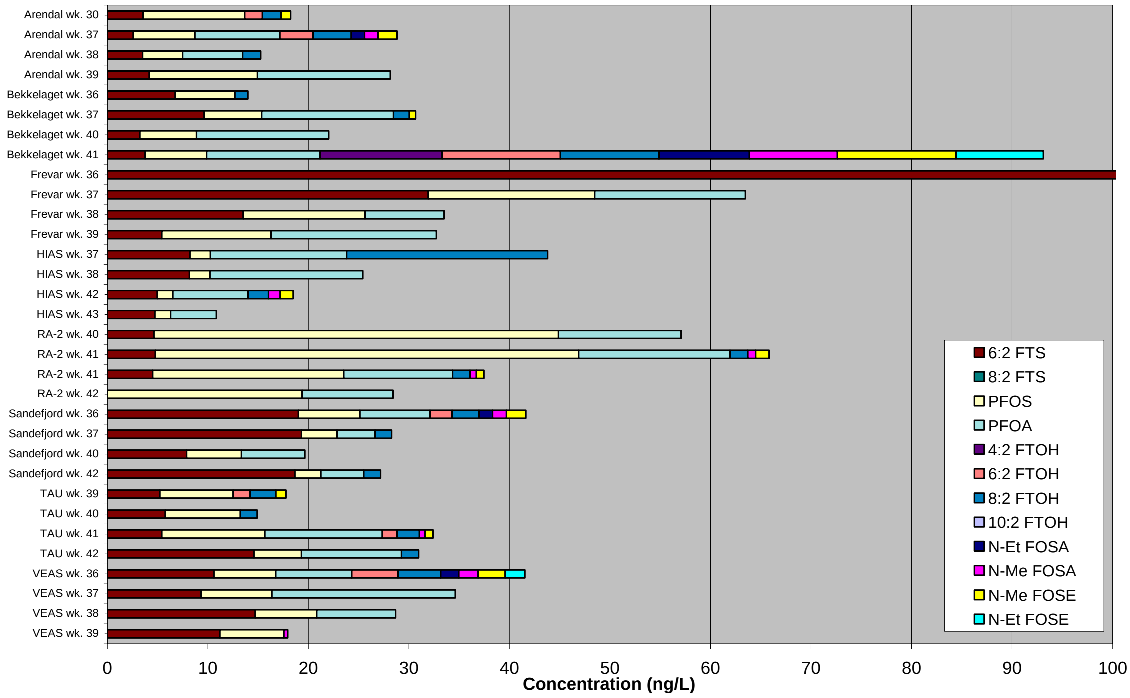

Figure 8. Occurrence of selected fluorinated substances in WWTP effluent

8:2 FTOH was detected in all sludge samples and had the highest median value of all of the perfluorinated compounds studied. Arendal had the highest median value for 8:2 FTOH (32.4 ng/g) and also the highest median for 10:2 FTOH (15.5 ng/g). Both of the fluorotelomer alcohols were less predominant in the effluent samples with median values for 8:2 FTOH and 10:2 FTOH of 1.68 ng/L and <LoD respectively. Data for 2 additional fluorotelomer compounds, 4:2 and 6:2 FTOH, are also included for comparison and were less frequently detected.

Data on the occurrence of the PFOS precursors, ethylated and methylated sulfonamides, N-Et FOSA and N-Me FOSA, and ethylated and methylated sulfonamidoethanols, N-Ee FOSE and N-Mt FOSE, are also presented.

Table 13. Predicted no-effect concentrations (PNEC) for selected fluorinated substances.

Chemical	Compartment	PNEC	Source
PFOS	Freshwater	25,000 ng/L	(Brooke *et al.*, 2004)
	Marine water	2,500 ng/L in chronic tests	
	Marine water	36,000 ng/L in acute tests	
	Sediment (freshwater)	67 ng/g (wet weight)	
	Sediment (marine)	6.7 ng/g (wet weight)	
	Effluent	>9.05 x 10^6 ng/L	
	Soil	100 ng/g (dry weight)	(Tremoen, 2008)
PFOA	Water	1.25 x 10^6 ng/L	(Colombo *et al.*, 2008)
	Soil	160 ng/g (dry weight)	(Tremoen, 2008)
6:2 FTS	Soil	210 ng/g (dry weight)	(Tremoen, 2008)

A study including 2 sewage sludge samples in Norway revealed concentrations of sum-PFC of 1048-1654 pg/g (wet weight) (Kallenborn *et al.*, 2004). These concentrations were higher than those measured in Finland but less than in Sweden. The sum-PFC included PFOSA + PFHxS + PFOS + PFHxA + PFOA + PFNA but excluded the fluorotelomers measured in the present study. In a Danish study of sludge and effluent at 6 WWTPs the effluent range of PFOS was <1.5-18.1 ng/L and <2-24.4 ng/L for PFOA, with higher concentrations generally detected in the effluent than the influent (Bossi *et al.*, 2008). The concentrations are comparable with the present study. The sludge concentrations were 4.8-74.1 and 3.4-19.7 ng/g (dry weight) for PFOS and PFOA respectively which are higher than those in this study. Another recent sludge study reported comparable PFOA concentrations in Spanish WWTP sludge but higher concentrations for PFOS (Navarro *et al.*, 2011) and a similar pattern was observed in Taiwan (Lin *et al.*, 2010) and Switzerland (Sun *et al.*, 2011). (Ma and Shih, 2010) noted that sludge was the major sink for PFOS and PFOA during activated sludge treatment.

Biodegradation of fluorotelomer alcohols during biological wastewater treatment results in the formation of PFOA and fluorotelomer acids. 6:2 FTS was determined in a US study (Schultz, 2006). It was not measured in sludge, but influent and effluent measurements showed no change in mass flow during the treatment process, in the same study, and increase in PFOS mass flow was observed. No PFOS degradation has been observed in aerobic or anaerobic wastewater treatment processes (Dinglasan *et al.*, 2004; Wang *et al.*, 2005a; Wang *et al.*, 2005b).

The occurrence of N-EtFOSA results from commercial products as well as transformation from N-EtFOSE during actived sludge treatment, where as the presence of M-MeFOSA is during from the transformation of N-MeFOSE (Schultz, 2006). A sediment study showed greater sorption of N-EtFOSA and N-MeFOSA than PFOS (Higgins, 2006) to sediment so the most likely removal pathway for these compounds is through sludge sorption as with PFOS.

The mass transport in mg/day of fluorinated compounds in the effluents from the various treatment plants is given in table 14. Based on the concentrations and flow rates during sampling, the total load of fluorinated compounds from VEAS is approximately 3 kg/year. In general, relative to population equivalents, the effluent load of perfluorinated compounds is less for the WWTP with biological treatment processes, compared to those without. FREVAR for example, uses a chemical sedimentation process and has the highest loads of 6:2 FTS, PFOS and PFOA compared to the other WWTP, although inputs sources have not been accounted for. Sandefjord and TAU effluent loadings are also higher than the biological treatment plants relative to population equivalents.

Table 14. Mass transport of fluorinated compounds (mg/day) from the WWTPs included in the study. Absent or single value indicates no or only one detected measurement.

		Fluorinated compounds (mg/day)							
		Bekkelaget	VEAS	Sandefjord	TAU	Arendal	HIAS	FREVAR	RA2
6:2 FTS	Min	523	2088	375	175	31	105	193	172
	Median	718	3331	412	314	58	181	1117	192
	Max	968	4349	493	540	77	195	32048	280
PFOS	Min	577	1541	68	175	74	33	388	685
	Median	631	1690	103	296	105	45	563	1192
	Max	905	2274	261	698	189	49	756	2468
PFOA	Min	1318	1889	83	368	102	158	451	320
	Median	1715	2316	130	581	131	298	636	458
	Max	2113	4087	301	794	233	362	804	882
6:2 FTOH	Min				85	21			
	Median				92	31			
	Max		1150	47	99	40			
8:2 FTOH	Min	137		36	51	22	43		66
	Median	148		44	96	40	241		85
	Max	158	1065	57	152	46	440		104
N-Et FOSA	Min								
	Median								
	Max		452	29		16			
N-Me FOSA	Min		133						
	Median		308						
	Max		482	29	38	16	25		35
N-Me FOSE	Min				49	11			45
	Median				51	17			29
	Max	63	670	40	54	23	27		54
N-Et FOSE	Min								
	Median								
	Max		486						78

3.5 Organophosphate flame retardants

All bar two of the organophosphate flame retardants (OPFR) were detected in all sludge samples collected at concentrations ranging from 0.8 to 23,600 ng/g (Table 15). DPhBP and TEHP were the two compounds that were not detected at quantifiable concentrations. The pattern of compounds was generally similar between all of the WWTPs with TCPP, TBEP and EHDPP dominating (Figure 9). The exceptions were FREVAR WWTP where TCP dominated the OPFR signal in three of the four samples collected and a single sample from Sandefjord (week 37) which contained relatively high levels of TPhP compared to the other samples. TBEP was present at the highest concentration in the majority of samples with a median concentration of 3,660 ng/g and a maxium concentration of 23,600 ng/g. This was followed by TCPP (median=2,580 ng/g; max.=7,200 ng/g) EHDPP (median=2,300 ng/g; max.=12,800 ng/g) and TCP (median=460 ng/g; max.=12,000 ng/g). Median concentrations of the other OPs were similar and typically below 200 ng/g.

Table 15. Summary of the occurrence of organophosphate flame retardants in WWTP effluent and sludge.

	Sludge (ng/g)			Effluent (ng/L)		
	Median	Min	Max	Median	Min	Max
TIBP	102	38.8	356	168	104	308
TBP	94	26.4	304	41	-2	316
TCEP	128	30.4	276	62	3.64	272
TCPP	2,580	560	7,200	600	14.4	1,160
DBPhP	3.9	0.8	20.8	0.36	<LoD	4
DPhBP	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
TDCP	246	8.8	1040	104	<LoD	276
TBEP	3,660	236	23,600	158	<LoD	60,000
TPhP	150	26.8	6,000	16.4	0.68	2,000
EHDPP	2,300	21.2	12,800	7.6	1.04	48
TEHP	<LoD	<LoD	<LoD	<LoD	<LoD	5.6
ToCP	6	<LoD	20.8	<LoD	<LoD	14.4
TCP	460	10.8	12,000	1.88	0.48	640

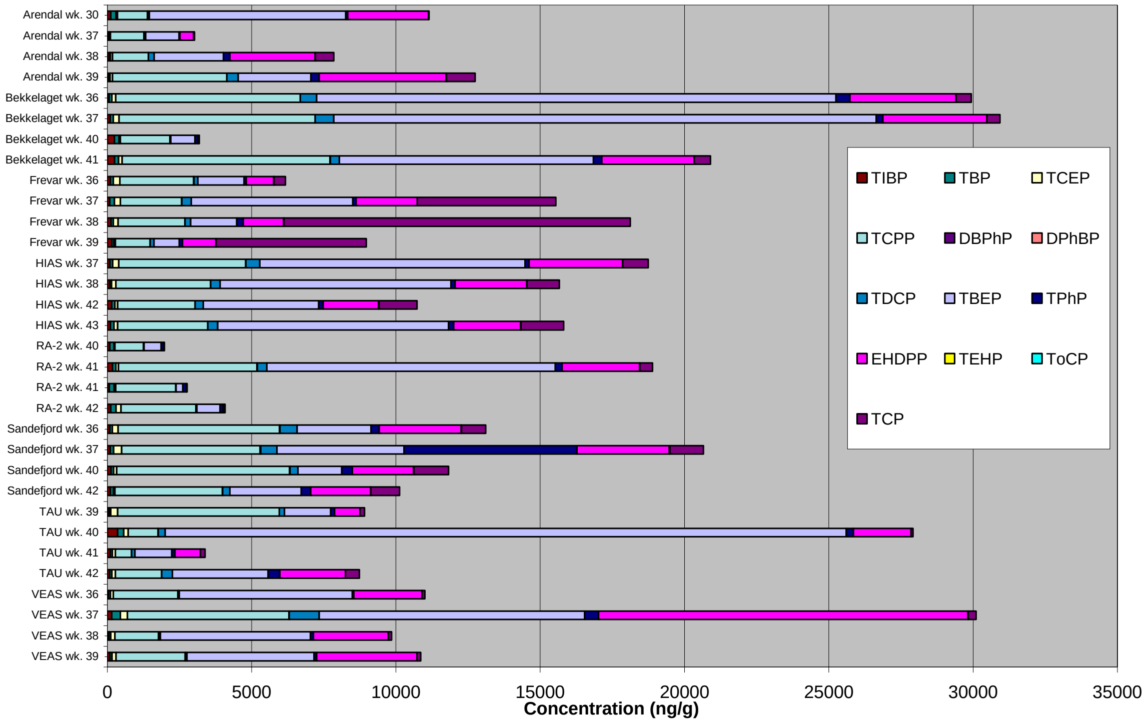

Figure 9. Occurrence of organophosphate flame retardants in sewage sludge.

Previous studies in Norway have reported the occurrence OPFR in sludge from Bekkelaget and Solumstrand WWTPs with a total OPFR concentration of around 5,000 ng/g (Green *et al.*, 2008). These data are similar to those reported here. In the previous study on sludge TBEP, EHDPP, TCPP and TPhP dominated the OPFR pattern. Again this is generally similar to the pattern observed in this study except for the occurrence of TPhP, which only predominated in the single sample from Sandefjord. A study from Sweden also reported similar levels of total OPFR in sewage sludge with EHDPP and TCPP dominating (Marklund *et al.*, 2005). Where

TBEP was detected, it dominated the OPFR occurrence pattern. TBEP was also not detected in a number of samples.

A different pattern of OPFR occurrence was detected in the WWTP effluent samples (Figure 10). TCPP was present at the highest median concentration (600 ng/L), however in four of the samples TBEP was present at relatively high concentrations with a maximum of 60,000 ng/L in a sample from Bekkelaget. TBEP dominated the OPFR occurrence pattern in a number of the samples but was also present in lower concentrations in a number of samples as is reflected by the median concentration of 158 ng/L. Overall, the OPFR pattern was dominated by TCPP, TIBP and TBEP (Figure 10). The high concentration of TPhP in the sludge sample from Sandefjord (week 37) was also repeated in the composite effluent sample. This was also true to TCP which predominated the OPFR pattern in both sludge and effluent from FREVAR WWTP.

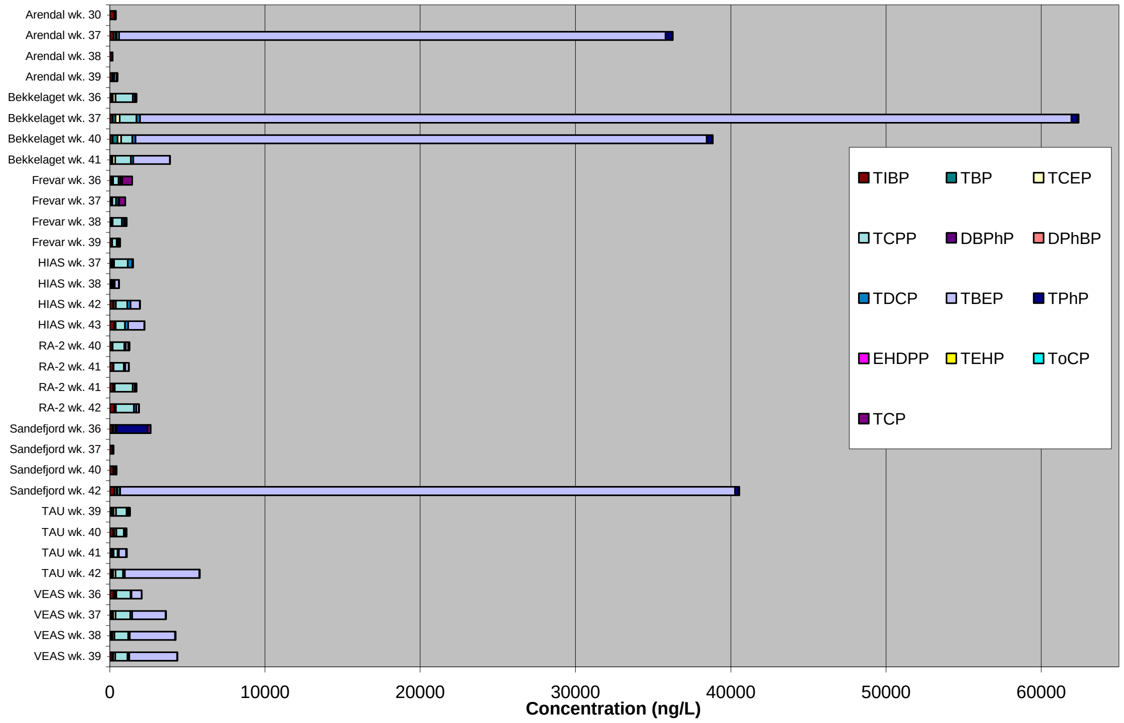

Figure 10. Occurrence of organophosphate flame retardants in WWTP effluent.

This pattern was generally reflected in the previously reported occurrence of OPFR in Norwegian WWTPs where TCEP, TCPP, TPhP and TBEP dominated (Green *et al.*, 2008). TCEP has previously reported as present between 1,600 and 2,200 ng/L in Norwegian WWTP effluents, between 190 and 1,800 ng/L in Swedish WWTP effluent and between 310 and 560 ng/L in German effluent (Meyer and Bester, 2004; Marklund *et al.*, 2005; Green *et al.*, 2008; Lilja *et al.*, 2010) suggesting the median concentration of 16 ng/L reported here to be very low. This is true for all of the OPFR concentrations reported here, with the exception of TBEP in four samples, in that they are an order of magnitude below those previously reported for WWTP effluents. For example, in Sweden, TBEP has previously been reported to occur at very high levels in WWTP effluents at concentrations of between 240 and 30,000 ng/L (Marklund *et al.*, 2005; Lilja *et al.*, 2010) with levels of 1,600 and 3,200 ng/L previously reported in effluents from Norway (Green *et al.*, 2008).

These occurrence data can be compared with the PNECs provided from different studies (Table 16). When comparing the available $PNEC_{WWTP}$ data with the effluent occurrence data there is very little risk to WWTP microorganisms from TCEP, TCPP and TDCP. No $PNEC_{WWTP}$ are available at present for the other OPFR. When comparing the effluent occurrence data with the available $PNEC_{Water}$ and $PNEC_{Marine\ water}$ only TBEP and TCP merit discussion. Four of the effluent samples collected (Arendal week 37, Bekkelaget weeks 37 and 40 and Sandefjord week 42) contained concentrations of TBEP greater than the $PNEC_{Marine\ water}$ of 1,300 ng/L. TCP has a particularly low $PNEC_{Marine\ water}$ of 3.2 ng/L and twelve samples contained effluent concentrations above the $PNEC_{Marine\ water}$. These concentrations only pose a risk to the environment if the dilution at the point of discharge is not sufficient to reduce the concentration to below the $PNEC_{Marine\ water}$. The median sludge concentration for TCPP, TBEP, TCP and TPhP were greater than the $PNEC_{Soil}$ (Table 16). Detailed terrestrial risk assessment of the different application and soil scenarios are outside the scope of this report but none of the median concentrations are sufficiently high to be above the $PNEC_{Soil}$ following moderate dilution upon application to land.

Table 16. Available predicted no-effect concentrations (PNEC) for selected OPFR

Chemical	Compartment	PNEC	Source
TCEP	Water	6,500 ng/L	(EU, 2009)
	WWTP (effluent)	32 x 10^6 ng/L	
	Sediment	200 ng/g (wet weight)	
	Soil	0.341 ng/g (wet weight)	
TCPP	Water	640,000 ng/L	(EU, 2008)
	WWTP (effluent)	7.84 x 10^6 ng/L	
	Sediment	1,700 ng/g (dry weight)	
	Soil	0.341 ng/g (wet weight)	
TBEP	Water	13,000 ng/L	(Verbruggen *et al.*, 2005)
	Marine water	1,300 ng/L	
	Soil	810 ng/g (dry weight)	
TDCP	Water	10,000 ng/L	(EU, 2008)
	Marine water	1,300 ng/L	
	WWTP (effluent)	≥100 mg/L	
	Soil	330 ng/g (dry weight)	
	Sediment	80 ng/g (wet weight)	
TIBP	Water	11,000 ng/L	(Verbruggen *et al.*, 2005)
	Marine water	1,100 ng/L	
	Soil	640 ng/g (dry weight)	
TPhP	Water	160 ng/L	(Verbruggen *et al.*, 2005)
	Marine water	16 ng/L	
	Soil	95 ng/g dw	
TBP	Water	66,000 ng/L	(Verbruggen *et al.*, 2005)
	Marine water	6,600 ng/L	
	Soil	5300 ng/g (dry weight)	
TCP	Water	32 ng/L	(Verbruggen *et al.*, 2005)
	Marine water	3.2 ng/L	
	Soil	8.9 ng/g (dry weight)	

The mass transport in mg/day of organophosphate flame retardants in the effluents from the various treatment plants is given in table 17. Based on the current concentration measurements, the total load of organophosphate flame retardants from Bekkelaget WWTP is approximately 560 kg/year, and from VEAS approximately 100 kg/year. The larger difference in mass transport is reflected in the high concentrations measured in two of the four samples from Bekkelaget WWTP.

Table 17. Mass transport of organophosphate flame retardants (mg/day) from the WWTPs included in the study. Absent or single value indicates no or only one detected measurement.

		Organophosphate flame retardants (mg/day)							
		Bekkelaget	VEAS	Sandefjord	Tau	Arendal	HIAS	FREVAR	RA2
TIBP	Min	15304	29548	3764	5036	3040	2476	5276	6768
	Median	19712	43136	5968	6960	3216	2816	5536	9348
	Max	30868	64032	10668	9248	3684	4728	6420	13132
TBP	Min	6376	19008	356	2432	100	364	68	
	Median	16896	20484	592	3732	944	1012	1176	
	Max	50808	27184	3696	5984	1784	1056	1948	536
TCEP	Min	17428	22012	480	3408	112	200	128	992
	Median	27352	34876	1164	4856	132	2376	1264	2608
	Max	36660	44356	2476	7180	772	2616	2752	3656
TCP	Min	108608	214884	316	14604	404	2288	10548	31368
	Median	115764	245024	3264	17456	1016	15196	12652	39252
	Max	119020	286168	4572	33908	1688	19356	34392	44188
DBPhP	Min	68	868	20	12		8	12	
	Median	136	1028	20	20		8	16	
	Max	308	1304	20	24	16	8	28	132
TDCP	Min	14452	16008	476	2800	64	2380	3564	3632
	Median	22928	27000	584	4444	192	4812	5584	5504
	Max	31512	37200	1392	4624	3096	6072	7336	6096
TBEP	Min	9352	160080	76	864	224	1232	284	3200
	Median	5916732	672996	900	16800	288	6192	548	6036
	Max	6033884	1101740	1045384	177792	424664	12664	1400	12896
TPhP	Min	596	2800	24	1156	16	44	840	140
	Median	39420	4768	3080	2104	48	140	1180	224
	Max	54024	6500	42312	4588	4824	208	1628	2972
EHDPP	Min	110.4	950.4	51.6	237.2	38.4	56.4	684.4	127.2
	Median	3057.2	2239.6	198.8	391.6	112.8	62.4	1506.8	182.8
	Max	3344.4	2861.6	506.8	680	330.8	152.4	1908.4	281.2
TEHP	Min			105.2		13.6	22	64	
	Median			186		17.6	57.6	68.4	
	Max		220	266.8	28	22	92.8	73.2	
ToCP	Min	112.8		16	21.6		10.4		14.8
	Median	265.2		160.4	52.8		10.4		27.6
	Max	418	150	304.8	84	110.8	10.4	153.6	49.6
TCP	Min	238	257.6	21.2	48.8	9.2	12	2851.2	28.8
	Median	3902	415.2	384	88.8	17.6	33.6	12285.2	78
	Max	5788	840.4	4316	121.6	579.2	95.2	25443.2	264

3.6 Short and medium chain chlorinated paraffins

Short chain chlorinated paraffins (SCCP) were detected in 65% of the effluent samples collected over a concentration range of <LOD to 560 ng/L with at a median determined concentration of 102 ng/L (Table 18; Figure 11). SCCP were detected in all sludge samples at concentrations of between 74 and 12,258 ng/g with a median of 416 ng/g (Figure 12). Medium chain chlorinated paraffins (MCCP) were only detected at quantifiable levels in 13% of the effluent samples analysed. These few samples contained concentrations of between 170 and 942 ng/L with three of the five samples that contained measurable concentrations collected from FREVAR WWTP. MCCP were detected in all of the sludge samples collected at levels of between 14 and 7,000 ng/g.

Table 18. Summary of the occurrence of short- and medium-chain chlorinated paraffins in WWTP effluent and sludge.

	Sludge (ng/g)			Effluent (ng/L)		
	Median	Min	Max	Median	Min	Max
SCCP	416	74	12,258	<LOD	<LOD	560
MCCP	385	14	7,000	<LOD	<LOD	942

The concentrations of SCCP in effluent and sludge are similar to those previously reported in Norway (Fjeld, 2008). The concentration of MCCP in the samples collected is also similar to those previously reported in Norway, with a similar level of occurrence (Blytt and Storhaug, 2007; Fjeld, 2008). Occurrence data for MCCP in sewage sludge has been reported for a number of decades with concentrations of between 3,000-93,000 ng/g being reported in the UK (CEFAS, 1999). Previously reported median levels in Norwegian sewage sludge range from between 473 and 5,700 ng/g with a maximum of 11,800 ng/g (Fjeld, 2008).

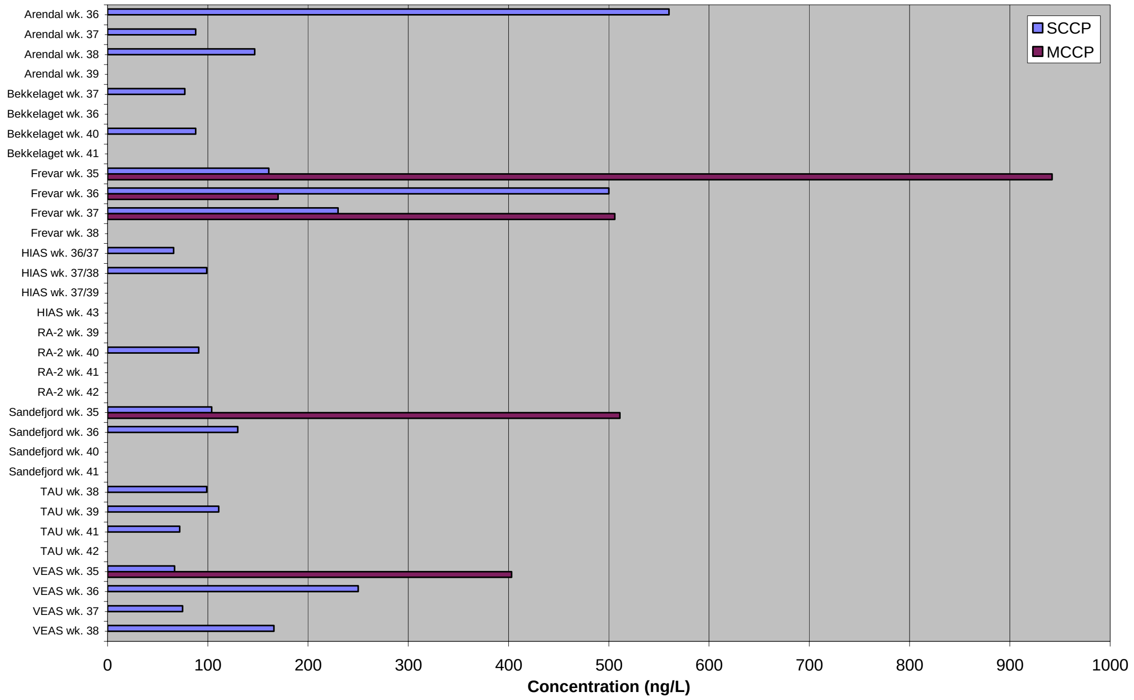

Figure 11. Occurrence of short- and medium chain chlorinated paraffins in WWTP effluent.

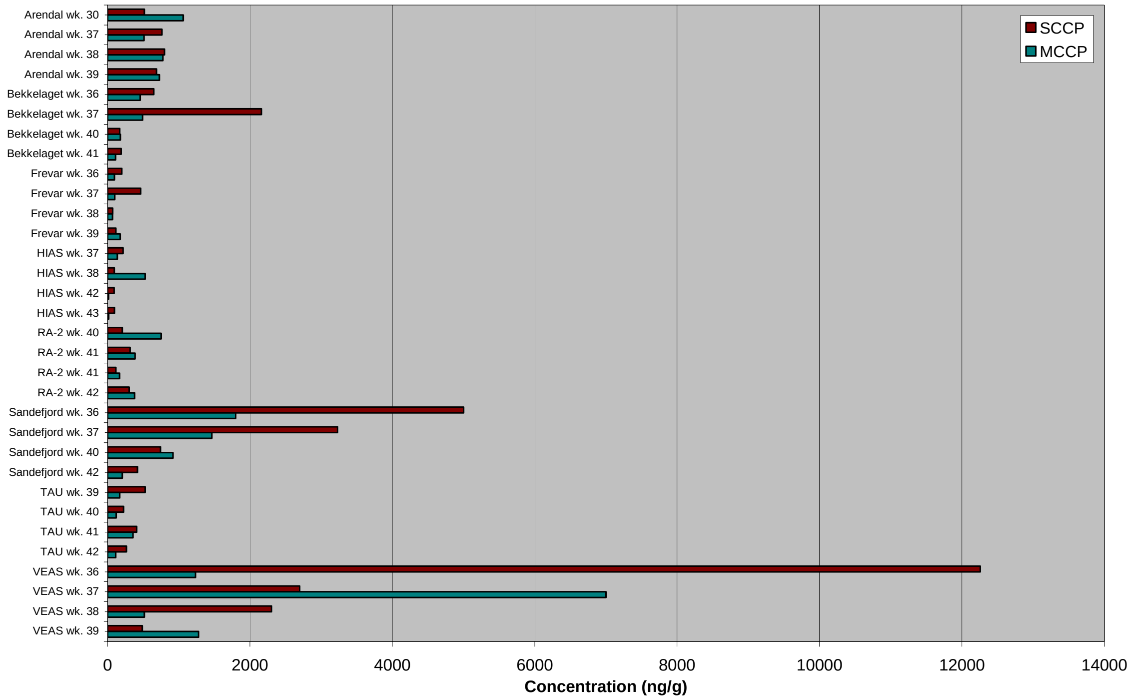

Figure 12. Occurrence of short- and medium chain chlorinated paraffins in sludge.

SCCP poses little risk to WWTP microorganisms when the maximum SCCP concentration is compared to the PNEC$_{\mathrm{WWTP}}$ (Table 19). Provided there is sufficient dilution at the point of effluent discharge there is also little risk to the aquatic environment from the levels of SCCP being released in WWTP effluent when using the PNEC$_{\mathrm{Water}}$ of 500 ng/L. The concentration of SCCP in sludge is also below the PNEC$_{\mathrm{Soil}}$ suggesting little risk to the terrestrial environment should the sludge be applied to land. Comparison of the MCCP occurrence data obtained within this study and available PNEC suggests that there is little risk to WWTP microorganisms and receiving waters from MCCP since the concentrations are numerous orders of magnitude below the PNEC$_{\mathrm{Water}}$ and PNEC$_{\mathrm{WWTP}}$. The median sludge concentration is below the PNEC$_{\mathrm{Soil}}$ indicating little risk should the sludge be applied to land.

Table 19. Predicted no-effect concentrations (PNEC) for SCCP and MCCP.

Chemical	Compartment	PNEC	Source
SCCP	Water	500 ng/L	(EU, 2000)
	WWTP (effluent)	6 x 10^6 ng/L	
	Soil	800 ng/g (wet weight)	
	Marine sediment	880 ng/g (wet weight)	
MCCP	Water	1,000 ng/L	(EU, 2005)
	WWTP (effluent)	80 x 10^6 ng/L	
	Soil	10,400 ng/g (wet weight)	
	Marine sediment	5,000 ng/g (wet weight)	

3.7 Silver

Silver was measured in concentrations above the limit of detection in all of the effluents samples and in all bar one of the sludge samples (Figures 13 and 14). Concentrations in effluent ranged from 0.01 to 0.49 µg/L with a median concentration of 0.05 µg/L (Table 20). Concentrations in sludge ranged from <0.01 to 9.55 µg/g with a median concentration of 2.52 µg/g.

Table 20. Summary data for the occurrence of silver in WWTP effluent and sludge.

	Sludge (µg/g)			Effluent (µg/L)		
	Median	Min	Max	Median	Min	Max
Silver	2.52	0.4	10.9	0.05	0.01	0.49

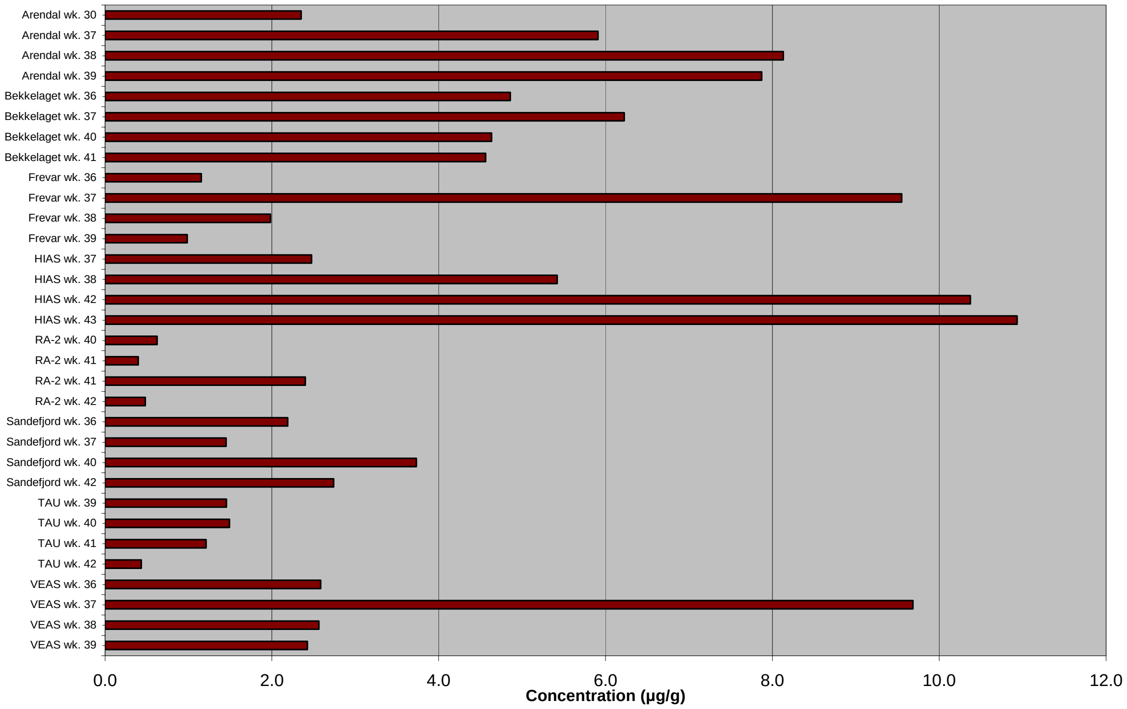

Figure 13. Concentration of total silver in sludge

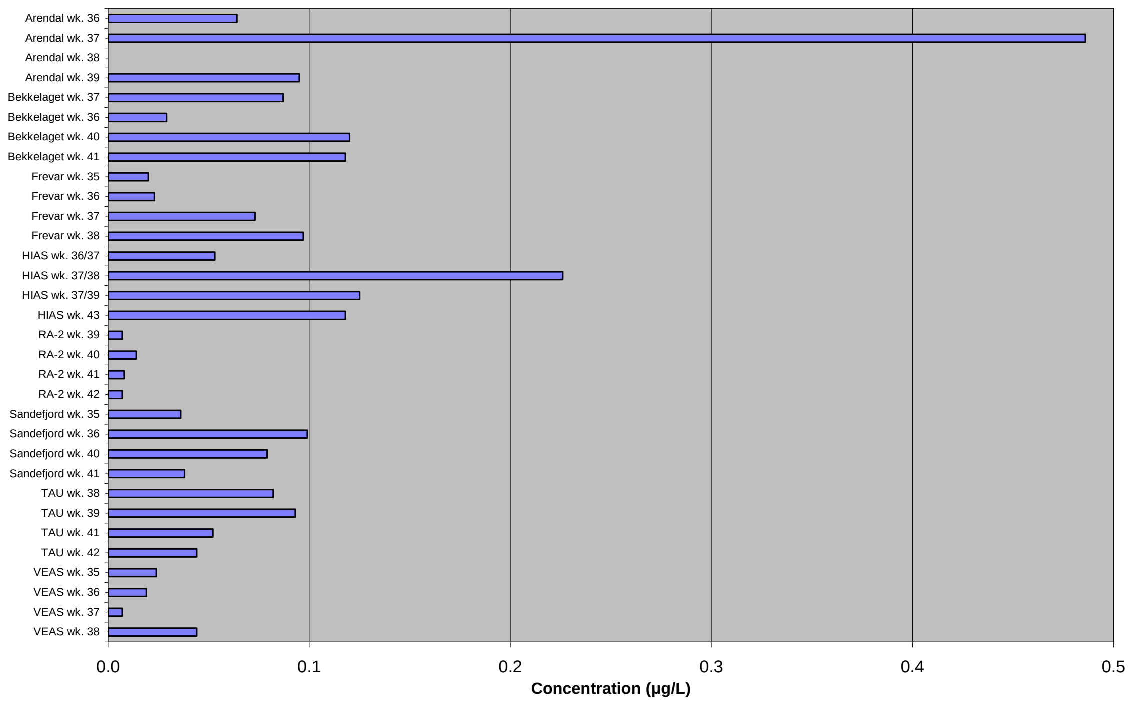

Figure 14. Concentration of total silver in treated effluent.

The discharged WWTP effluent concentrations are slightly higher than those reported in a previous screening of silver in the Norwegian environment in 2007 which were all below the limit of detection of 0.01 µg/L (Green *et al.*, 2008). These levels are within the range of silver effluent concentrations reported in the US (Shafer *et al.*, 1998). WWTP discharges along with disused metal mines are considered a major and important source of silver to the aquatic environment (Tappin *et al.*, 2010) with the incorporation of silver nanoparticles into consumer products being of clear concern in terms of inputs to WWTPs (Nowack, 2010). Silver has very low toxicity to humans, however this is not the case for microbe and invertebrate communities.

The concentrations of silver in the sludge samples collected were the same range as those reported in 2008 for Norwegian WWTPs (Green *et al.*, 2008) which in turn was similar to those being reported to occur in Swedish sludge. There exists no Norwegian classification limit for silver in sewage sludge, however the Swedish Environmental Protection Agency recommends a limit value of 15 µg/g in sludge that will be used in agriculture. All of the samples analysed in this study were below this limit value.

There is increasing focus on the occurrence of silver in both WWTP effluent and sludge due to its increasing use in nanoparticle form in consumer products. Recent studies have shown that much of the silver entering WWTPs 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). This report has important ramifications on understanding the risks associated with silver in both sludge and the aquatic environment receiving WWTP effluent since the toxicity of Ag_2S nanoparticles cannot be evaluated from ionic or nanoparticulate silver. It is clear that the speciation of silver in WWTPs and the toxicity of these species need to be understood before the risks associated with silver in WWTP sludge and effluent can be evaluated.

3.8 Synthetic musk compounds

The two synthetic musk compounds AHTN and HHCB were detected at quantifiable concentrations in all of the samples analysed. WWTP effluent concentrations of HHCB were much higher than those for AHTN with HHCB concentrations ranging from 267 to 1,259 ng/L with a median concentration of 267 ng/L (Table 21, Figure 15). AHTN was present at concentrations between 33 and 87 ng/L with a median concentration of 59 ng/L.

A similar pattern was observed in sewage sludge with both compounds being detected at quantifiable concentrations in all samples (Figure 16). Again HHCB was present at much higher concentrations than AHTN. HHCB was present at concentrations of between 423 and 26,300 ng/g with a median concentration of 9,550 ng/g whilst AHTN occurred at concentration of between 63 and 1334 ng/g at a median concentration of 800 ng/g.

Table 21. Summary data for the occurrence of AHTN and HHCB in WWTP effluent and sludge

	Sludge (ng/g)			Effluent (ng/L)		
	Median	Min	Max	Median	Min	Max
HHCB	9,550	63	1,334	709	267	1,259
AHTN	800	423	26,203	58.6	33	87

The occurrence of high concentrations of both compounds in sewage sludge is because they have a high affinity for sludge particles. Sorption to particles is the main mechanism by which these two synthetic musks are removed in WWTPs (Simonich *et al.*, 2002; Bester, 2004; Mogensen *et al.*, 2004; Zeng *et al.*, 2007; Zhang *et al.*, 2008). HHCB has a reported log K_d of 3.92 suggesting that the majority of the HHCB will be found adsorbed to solids.

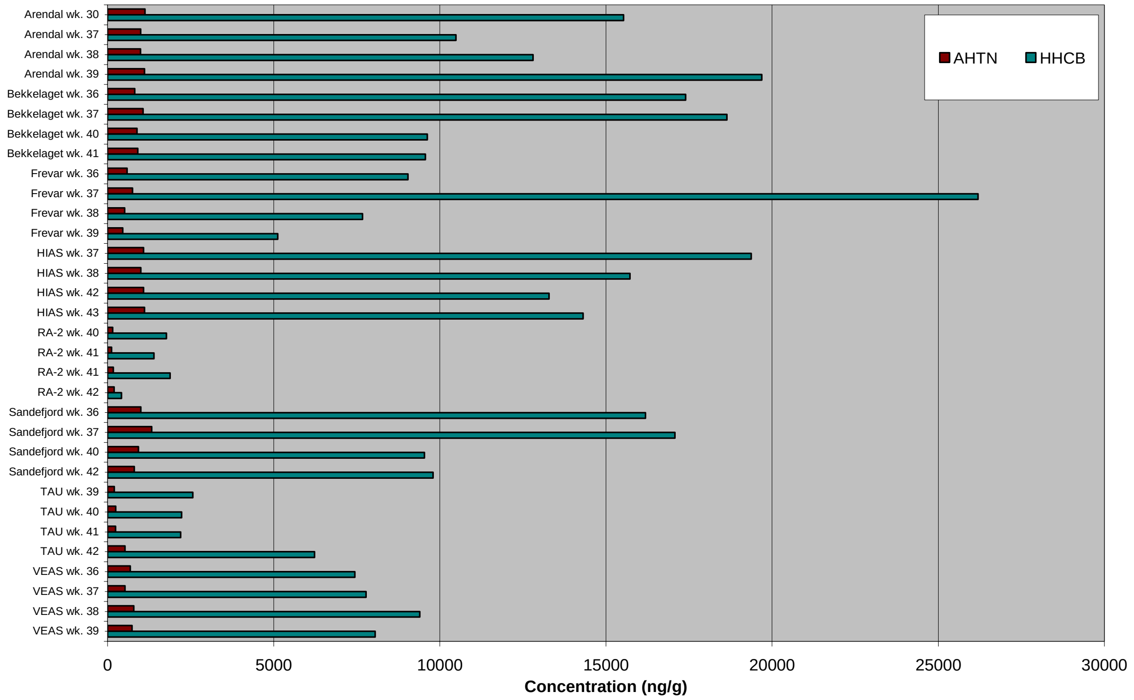

Figure 15. Occurrence of the synthetic musk compounds AHTN and HHCB in WWTP effluent.

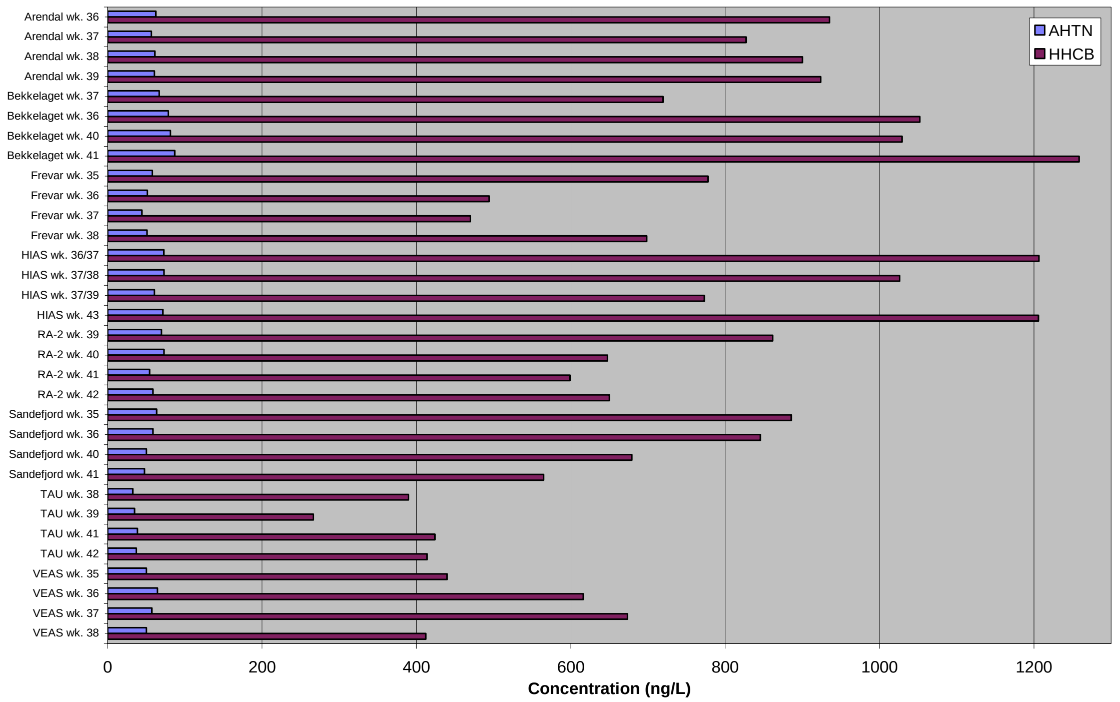

Figure 16. Occurrence of the synthetic musk compounds AHTN and HHCB in sludge.

Previously concentrations of between 254 and 22,400 ng/g for HHCB, and between 68 and 3,500 ng/g for AHTN, have been reported in the Norwegian sludge samples collected as part of the Nordic Screening programme in 2003 (Mogensen *et al.*, 2004). Higher concentrations of HHCB of up to 26,500 ng/g and AHTN of up to 3,610 ng/g were reported in sewage sludge from Denmark, reported as part of the same screening study. A 100% occurrence was reported for both compounds. The results of this study are therefore very similar to those reported in 2004. The EU in their risk assessment of these two synthetic musks report effluent occurrence data from across Europe with median concentrations of between 100 and 4,000 ng/L being typical for HHCB and between 100 and 500 ng/L for AHTN being common for Northern European countries. Similarly for sludge they report typical median concentrations of up to 20,000 ng/g for HHCB and 5,000 ng/g for AHTN. The results of this survey therefore appear to be in line with previously reported occurrence data for Northern Europe.

It is commonly reported that HHCB occurs at higher concentrations than AHTN in environmental samples with (Heberer, 2002) suggesting a ratio of 2.5:1 HHCB/AHTN for sewage sludge and effluent. The 2004 Nordic screening study reported a mean HHCB/AHTN ratio of 5:1 in sewage sludge whilst ratios of up to 15:1 (Mean 9:1) were reported for sewage sludge samples collected from a WWTP in Sweden (Mogensen *et al.*, 2004). This is clearly linked to the quantities of compounds used and compares to a median ratio of 12:1 HHCB/AHTN for the data reported in this study suggesting very different consumption pattern to that seen previously. It has recently been reported that the estimated Northern European per capita daily use of HHCB and AHTN is 6 and 1.5 mg/day respectively (EU, 2008; EU, 2008). The results when compared to previous studies suggest that there is very little reduction in the amounts being used in consumer products.

The risks posed by the concentrations determined in this study are best described by comparing the results with published PNEC (Table 22).

Table 22. Predicted no-effect concentrations (PNEC) for HHCB and AHTN

Chemical	Compartment	PNEC	Source
HHCB	Water	4,400 ng/L	(EU, 2008)
	WWTP (effluent)	> 2 x 10^6 ng/L	
	Soil	154,000 ng/g (dry weight)	
	Marine sediment	394 ng/g (dry weight)	
AHTN	Water	280 ng/L	(EU, 2008)
	WWTP (effluent)	> 3 x 10^6 ng/L	
	Soil	310 ng/g (dry weight)	
	Marine sediment	345 ng/g (dry weight)	

The effluent concentrations determined in this study are many orders of magnitude lower than $PNEC_{WWTP}$ for both HHCB and AHTN. The median effluent concentrations are also below the $PNEC_{Water}$ of 4,400 and 280 ng/L respectively, indicating that the levels determined in effluent pose little risk to the receiving environment since undiluted effluent contains concentrations below the PNEC level for surface waters meaning that the concentrations present in undiluted effluent are below that required to pose a risk to the environment.

The concentrations determined in sludge for HHCB are much lower than the $PNEC_{Soil}$ indicating that there is very little risk to the terrestrial environment from the application of sludge containing these levels. The median concentration of AHTN was above the $PNEC_{Soil}$ ($MEC_{Sludge}/PNEC_{Soil} = 2.6$) suggesting that with there is little risk to the terrestrial environment once the sludge has been dispersed on land.

The mass transport in mg/day of synthetic musk compounds in the effluents from the various treatment plants is given in table 23. The total load of synthetic musk compounds from Bekkelaget WWTP was estimated to approximately 40 kg/year, and from VEAS approximately 60 kg/year.

Table 23. Mass transport of synthetic musk compounds (mg/day) from the WWTPs included in the study.

		Synthetic musk compounds (mg/day)							
		Bekke-laget	VEAS	Sandefjord	Tau	Arendal	HIAS	FREVAR	RA2
HHCB	Min	76457	109960	14909	8114	9979	16320	22597	22815
	Median	105796	142661	18623	17381	13793	24435	25915	29270
	Max	165448	198905	32340	28824	19762	26536	30927	37967
AHTN	Min	7087	12572	1262	1065	683	1282	1825	2069
	Median	7907	15700	1312	1503	913	1605	2329	2479
	Max	13108	18048	2400	2630	1346	1743	2552	4281

3.9 General

All of the compounds selected for screening were detected in the dewatered sludge. Should sludge from these WWTPs be applied to land there will be a direct release into the Norwegian terrestrial environment. With the exception of HBCD and TBBPA, all of the compounds selected for screening were detected in the WWTPs effluents at measurable concentrations, and are therefore being released into the Norwegian aquatic environment.

The simple environmental risk assessment applied in this report has used a conventional approach (as recommended by the EU TGD) whereby the measured median concentration for each individual chemical was compared with published PNEC for that particular chemical. This method of risk assessment will provide an indication of whether any risk is posed to the environment from levels of an individual chemical. This evaluation does not rule out risks associated with the combined effect of mixtures of chemicals that may have a combined effect even when present at concentrations below the PNEC.

When the median concentration of each individual compound is compared to available $PNEC_{WWTP,}$ there appears to be very little risk to WWTP microorganisms from the effluent concentrations of the selected contaminants. For many of the compounds included in the present screening, there also appears to be very little risk to the receiving aquatic environments since very few effluent samples contained levels greater than the $PNEC_{Water.}$ It is expected that dilution following discharge would bring these levels to below the $PNEC_{Water}$.

It is possible that the OPFRs, TBEP and TCP, pose a risk to aquatic organisms at the point of discharge at certain locations since the effluent concentrations detected were significantly above the $PNEC_{Marine\ water}$. A paucity of ecotoxicity data for deca-BDE make any assessment difficult, but measured effluent concentrations in excess of the proposed minimum $PNEC_{Water}$ suggest a potential risk to aquatic organisms at the point of discharge. Assessment of the environmental risk associated with silver is not possible at present due to a lack of understanding of silver speciation in WWTPs.

It is difficult to generalise with regard to the environmental risks associated with the levels detected in sludge since it is very much dependent on how the sludge will be used or disposed of. A detailed risk assessment based on different scenarios would be required. This is beyond the scope of the current study. None of the samples analysed contained levels of the selected contaminants significantly above sediment or soil PNEC values, suggesting little risk to soil dwelling organisms once the sludge is dispersed on land.

Octa-PBDE, penta-BDE, HBCD, SCCP and PFOS are classified as persistent (P), bioaccumulative (B) and toxic (T) (PBT or vPvB) substances, whilst the classification of MCCP is being evaluated. As PBT/vPvB substances, their occurrence in the environment at concentrations below predicted no-effect concentrations (PNEC) does not necessarily mean that there is no risk to the environment from these compounds. This also applies to compounds that have not yet been classified. The risk assessment used for hydrophobic chemicals into the environment do not tell us about the risks associated with their accumulation in biota, sediments and soils, even though they appear not to represent a direct risk to the aquatic and soil environments.

4. Conclusions

- The majority of the chemicals selected for screening, except for HBCD and TBBPA, are directly entering the Norwegian aquatic environment via effluent from WWTPs.

- All of the compounds selected for screening were detected in dewatered sludge and application to land will result in direct input to the terrestrial environment.

- There appears to be very little risk to WWTP microorganisms from the effluent concentrations of the selected contaminants based on an assessment of individual chemicals. Risk associated with the combined effect of mixtures of the present contaminants, and also mixtures including other chemicals, has not been estimated in the present study.

- Few of the chemicals selected for screening pose a direct risk to the aquatic environment at the sites investigated. It is possible that the OPFRs, TBEP and TCP, pose a risk to aquatic organisms at the point of discharge at certain locations. There is uncertainty associated with the risks posed to the aquatic environment by deca-BDE and silver due to insufficient data. Again, the risk associated with the occurrence of mixtures of these and other chemicals having a combined effect has not been evaluated and can not be estimated.

- It is difficult to generalise with regards to the environmental risks associated with the levels detected in sludge since it is very much dependent on how it will be used or disposed of. None of the samples contained levels of the selected contaminants significantly above sediment or soil PNECs, suggesting little risk.

- For compounds such as octa-PBDE, penta-BDE, HBCD, SCCP and PFOS classified as persistent (P), bioaccumulative (B) and toxic (T) (PBT or vPvB), a measured environmental concentration below the PNEC does not guarantee that there is no risk to the environmental. This applies also to other compounds that are not yet classified.

- With the release of hydrophobic chemicals into the environment, there can be risks associated with their accumulation in biota, sediments and soils, even though data indicate no major direct risk to the aquatic environment.

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6. Appendix A

Table A1. Concentrations of *silver* in sludge and effluent samples from the 8 WWTPs

Silver	Sludge ug/g	Effluent ug/L
Arendal wk. 30	2.35	0.06
Arendal wk. 37	5.91	0.49
Arendal wk. 38	8.13	<LoD
Arendal wk. 39	7.87	0.10
Bekkelaget wk. 36	4.86	0.09
Bekkelaget wk. 37	6.23	0.03
Bekkelaget wk. 40	4.63	0.12
Bekkelaget wk. 41	4.56	0.12
Frevar wk. 36	1.15	0.02
Frevar wk. 37	9.55	0.02
Frevar wk. 38	1.98	0.07
Frevar wk. 39	0.98	0.10
HIAS wk. 37	2.48	0.05
HIAS wk. 38	5.42	0.23
HIAS wk. 42	10.4	0.13
HIAS wk. 43	10.9	0.12
RA-2 wk. 40	0.63	0.01
RA-2 wk. 41	0.40	0.01
RA-2 wk. 41	2.40	0.01
RA-2 wk. 42	0.48	0.01
Sandefjord wk. 36	2.19	0.04
Sandefjord wk. 37	1.45	0.10
Sandefjord wk. 40	3.73	0.08
Sandefjord wk. 42	2.74	0.04
TAU wk. 39	1.46	0.08
TAU wk. 40	1.49	0.09
TAU wk. 41	1.21	0.05
TAU wk. 42	0.44	0.04
VEAS wk. 36	2.59	0.02
VEAS wk. 37	9.69	0.02
VEAS wk. 38	2.56	0.01
VEAS wk. 39	2.43	0.04

Table A2. Concentrations of s*hort and medium chained paraffins (SCCP and MCCP)* in sludge and effluent samples from the 8 WWTPs

	Sludge (ng/g)		Effluent (ng/L)	
	SCCP	MCCP	SCCP	MCCP
Arendal wk. 30	517	1 064	560	<LoD
Arendal wk. 37	765	512	88.0	<LoD
Arendal wk. 38	800	778	147	<LoD
Arendal wk. 39	687	730	<LoD	<LoD
Bekkelaget wk. 36	650	460	77.0	<LoD
Bekkelaget wk. 37	2 161	492	<LoD	<LoD
Bekkelaget wk. 40	173	182	88	<LoD
Bekkelaget wk. 41	192	111	<LoD	<LoD
Frevar wk. 36	200	100	161	942
Frevar wk. 37	465	101	500	170
Frevar wk. 38	74.0	71.0	230	506
Frevar wk. 39	118	178	<LoD	<LoD
HIAS wk. 37	220	136	66.0	<LoD
HIAS wk. 38	96.0	528	99.0	<LoD
HIAS wk. 42	92.0	14.0	<LoD	<LoD
HIAS wk. 43	100	16.0	<LoD	<LoD
RA-2 wk. 40	207	754	<LoD	<LoD
RA-2 wk. 41	317	387	91.0	<LoD
RA-2 wk. 41	118	170	<LoD	<LoD
RA-2 wk. 42	303	382	<LoD	<LoD
Sandefjord wk. 36	5 000	1 800	104	511
Sandefjord wk. 37	3 230	1 465	130	<LoD
Sandefjord wk. 40	744	922	<LoD	<LoD
Sandefjord wk. 42	421	209	<LoD	<LoD
TAU wk. 39	530	174	99.0	<LoD
TAU wk. 40	225	122	111	<LoD
TAU wk. 41	410	359	72.0	<LoD
TAU wk. 42	266	116	<LoD	<LoD
VEAS wk. 36	12 258	1 236	67.0	403
VEAS wk. 37	2 700	7 000	250	<LoD
VEAS wk. 38	2 305	518	75.0	<LoD
VEAS wk. 39	489	1 279	166	<LoD

Table A3. Concentrations of *organophosphate flame retardants (OPFR)* in sludge from the 8 WWTPs

Sludge (ng/g)	TIBP	TBP	TCEP	TCPP	DBPhP	DPhBP	TDCP	TBEP	TPhP	EHDPP	ToCP	TCP
Arendal wk. 30	116	188	48.0	1 040	5.60	<LoD	60.0	6 800	72.0	2 800	<LoD	18.0
Arendal wk. 37	38.8	26.4	48.0	1 160	3.52	<LoD	52.0	1 160	26.8	480	1.08	23.2
Arendal wk. 38	68.0	44.0	68.0	1 240	3.80	<LoD	200	2 400	220	2 960	5.20	640
Arendal wk. 39	48.0	48.0	84.0	3 960	4.00	<LoD	388	2 520	288	4 400	7.60	1 000
Bekkelaget wk. 36	64.0	92.0	132	6 400	2.60	<LoD	560	18 000	480	3 680	6.40	520
Bekkelaget wk. 37	100	104	200	6 800	2.32	<LoD	640	18 800	232	3 600	6.40	440
Bekkelaget wk. 40	248	156	48.0	1 720	3.64	<LoD	18.8	840	96.0	44.0	<LoD	14.8
Bekkelaget wk. 41	252	140	120	7 200	16.8	<LoD	312	8 800	292	3 200	10.0	560
Frevar wk. 36	112	92.0	232	2 560	2.72	<LoD	136	1 600	76.0	960	4.40	400
Frevar wk. 37	84.0	164	208	2 120	4.80	<LoD	328	5 600	108	2 120	<LoD	4 800
Frevar wk. 38	132	76.0	160	2 320	4.40	<LoD	192	1 600	228	1 400	6.80	12 000
Frevar wk. 39	156	68.0	56.0	1 200	1.92	<LoD	132	880	112	1 160	5.60	5 200
HIAS wk. 37	88.0	96.0	212	4 400	2.32	<LoD	480	9 200	132	3 240	10.4	880
HIAS wk. 38	80.0	64.0	152	3 280	2.04	<LoD	332	8 000	144	2 480	7.60	1 120
HIAS wk. 42	152	108	100	2 680	3.52	<LoD	280	4 000	156	1 920	15.6	1 320
HIAS wk. 43	112	112	136	3 120	4.40	<LoD	340	8 000	180	2 320	8.00	1 480
RA-2 wk. 40	84.0	144	30.4	1 000	1.96	<LoD	8.80	600	72.0	25.6	<LoD	10.8
RA-2 wk. 41	180	112	92.0	4 800	12.8	<LoD	332	10 000	232	2 680	7.20	440
RA-2 wk. 41	72.0	164	52.0	2 080	2.72	<LoD	12.0	236	104	21.2	2.00	20.8
RA-2 wk. 42	120	180	172	2 600	5.60	<LoD	28.4	800	104	56.0	0.84	11.6
Sandefjord wk. 36	88.0	84.0	196	5 600	10.4	<LoD	600	2 560	284	2 840	8.00	840
Sandefjord wk. 37	104	116	276	4 800	20.8	<LoD	560	4 400	6 000	3 200	20.8	1 160
Sandefjord wk. 40	132	88.0	100	6 000	14.0	<LoD	272	1 520	364	2 120	13.2	1 200
Sandefjord wk. 42	108	104	52.0	3 720	11.6	<LoD	248	2 480	320	2 080	5.20	1 000
TAU wk. 39	64.0	56.0	240	5 600	1.08	<LoD	180	1 600	132	880	3.20	156
TAU wk. 40	356	216	148	1 040	1.40	<LoD	244	23 600	248	2 000	<LoD	68.0
TAU wk. 41	112	60.0	108	560	0.80	<LoD	112	1 280	112	880	1.52	156
TAU wk. 42	84.0	72.0	124	1 600	3.44	<LoD	368	3 320	400	2 280	4.00	480
VEAS wk. 36	44.0	68.0	92.0	2 240	6.40	<LoD	40.0	6 000	52.0	2 360	9.60	88.0
VEAS wk. 37	148	304	244	5 600	7.20	<LoD	1 040	9 200	480	12 800	10.4	272
VEAS wk. 38	52.0	64.0	144	1 520	6.80	<LoD	52.0	5 200	100	2 600	8.00	100
VEAS wk. 39	92.0	72.0	136	2 400	6.40	<LoD	56.0	4 400	88.0	3 480	4.80	120

Table A4. Concentrations of *organophosphate flame retardants (OPFR)* in effluent water from the 8 WWTPs

Effluent (ng/g)	TIBP	TBP	TCEP	TCPP	DBPhP	DPhBP	TDCP	TBEP	TPhP	EHDPP	TEHP	ToCP	TCP
Arendal wk. 30	304	<LoD	10.0	36.0	<LoD	<LoD	16.0	18.4	1.40	3.16	1.12	<LoD	0.76
Arendal wk. 37	252	148	64.0	132	1.40	<LoD	5.20	35 200	400	14.8	<LoD	9.20	48.0
Arendal wk. 38	152	<LoD	6.40	18.4	<LoD	<LoD	<LoD	14.4	1.24	2.16	<LoD	<LoD	0.48
Arendal wk. 39	176	5.60	6.40	96.0	<LoD	<LoD	176	14.8	3.76	18.8	1.24	<LoD	1.40
Bekkelaget wk. 36	144	60.0	164	1 120	0.64	<LoD	136	88.0	5.60	1.04	<LoD	<LoD	2.24
Bekkelaget wk. 37	196	168	272	1 080	1.36	<LoD	228	60 000	392	30.4	<LoD	1.12	38.8
Bekkelaget wk. 40	192	316	228	720	1.92	<LoD	196	36 800	336	20.8	<LoD	2.60	36.0
Bekkelaget wk. 41	124	60.0	176	1 000	0.68	<LoD	156	2 360	11.2	2.32	<LoD	<LoD	0.60
Frevar wk. 36	136	32.0	48.0	356	0.28	<LoD	120	35.2	26.8	48.0	1.84	<LoD	640
Frevar wk. 37	124	23.6	13.6	244	0.36	<LoD	140	12.0	28.4	28.8	1.40	<LoD	392
Frevar wk. 38	112	34.0	48.0	600	0.52	<LoD	128	<LoD	28.4	29.6	<LoD	2.68	116
Frevar wk. 39	148	1.96	3.64	296	<LoD	<LoD	100	8.00	23.6	19.2	<LoD	<LoD	80.0
HIAS wk. 37	128	48.0	108	880	0.36	<LoD	276	56.0	6.40	2.56	1.00	<LoD	1.52
HIAS wk. 38	104	15.2	8.40	96.0	<LoD	<LoD	100	260	8.80	6.40	<LoD	0.44	4.00
HIAS wk. 42	224	48.0	124	720	0.36	<LoD	228	600	2.04	2.96	4.40	<LoD	0.56
HIAS wk. 43	284	32.8	60.0	600	0.32	<LoD	212	1 040	2.04	9.20	<LoD	0.40	1.16
RA-2 wk. 40	164	<LoD	24.0	760	<LoD	<LoD	88.0	152	72.0	3.08	<LoD	0.36	6.40
RA-2 wk. 41	224	<LoD	36.0	640	<LoD	<LoD	104	220	4.80	4.80	<LoD	0.68	1.36
RA-2 wk. 41	220	<LoD	96.0	1 160	<LoD	<LoD	148	84.0	4.40	4.40	<LoD	0.40	0.76
RA-2 wk. 42	292	15.2	88.0	1 160	3.76	<LoD	152	164	4.00	5.60	<LoD	1.40	2.16
Sandefjord wk. 36	180	16.8	40.0	124	0.92	<LoD	26.0	30.0	2 000	2.44	<LoD	14.4	204
Sandefjord wk. 37	172	<LoD	22.0	14.4	<LoD	<LoD	28.4	3.56	1.04	2.48	4.80	<LoD	0.96
Sandefjord wk. 40	224	12.4	52.0	96.0	<LoD	<LoD	29.2	24.4	0.68	7.20	5.60	<LoD	0.60
Sandefjord wk. 42	308	140	56.0	148	0.80	<LoD	18.0	39 600	232	19.2	<LoD	0.60	28.0
TAU wk. 39	140	120	144	680	0.36	<LoD	92.0	19.2	92.0	9.60	<LoD	<LoD	2.04
TAU wk. 40	228	80.0	112	480	0.76	<LoD	92.0	28.4	40.0	10.0	0.92	2.76	4.00
TAU wk. 41	136	64.0	60.0	252	0.32	<LoD	68.0	480	44.0	10.0	<LoD	0.32	1.12
TAU wk. 42	136	84.0	152	480	0.36	<LoD	116	4 800	31.2	6.40	<LoD	<LoD	1.32
VEAS wk. 36	256	76.0	88.0	920	4.00	<LoD	64.0	640	11.2	3.80	0.88	0.60	3.36
VEAS wk. 37	132	88.0	148	960	3.88	<LoD	104	2 160	16.4	8.40	<LoD	<LoD	1.44
VEAS wk. 38	108	72.0	124	880	3.56	<LoD	104	2 920	22.0	8.80	<LoD	<LoD	1.72
VEAS wk. 39	152	76.0	124	800	3.64	<LoD	104	3 080	16.4	8.00	<LoD	<LoD	0.72

Table A5. Concentrations of s*ynthetic musk compounds* in sludge and effluent samples from the 8 WWTPs

	Sludge (ng/g)		Effluent (ng/L)	
	HHCB	AHTN	HHCB	AHTN
Arendal wk. 30	15529.5	1127.4	935	62.5
Arendal wk. 37	10488.9	997.3	827	56.6
Arendal wk. 38	12807.1	994.8	900	61.3
Arendal wk. 39	19689.6	1115.5	924	60.7
Bekkelaget wk. 36	17399.5	819.3	719	66.7
Bekkelaget wk. 37	18645.3	1066.8	1 052	78.6
Bekkelaget wk. 40	9622.7	884.9	1 029	81.5
Bekkelaget wk. 41	9562.9	913.0	1 259	87.0
Frevar wk. 36	9043.3	587.6	778	57.8
Frevar wk. 37	26202.6	751.2	494	51.6
Frevar wk. 38	7671.8	519.2	470	44.5
Frevar wk. 39	5124.5	460.3	698	51.2
HIAS wk. 37	19373.3	1085.8	1 206	73.0
HIAS wk. 38	15726.3	1002.3	1 026	73.2
HIAS wk. 42	13290.1	1087.6	773	60.8
HIAS wk. 43	14313.8	1117.5	1 206	71.6
RA-2 wk. 40	1775.8	156.7	862	69.9
RA-2 wk. 41	1398.3	125.5	648	73.0
RA-2 wk. 41	1885.3	177.0	599	54.3
RA-2 wk. 42	422.9	202.5	650	58.7
Sandefjord wk. 36	16189.8	1006.0	886	63.4
Sandefjord wk. 37	17078.3	1333.8	846	58.6
Sandefjord wk. 40	9536.1	933.5	679	50.4
Sandefjord wk. 42	9802.3	804.6	565	47.8
TAU wk. 39	2566.0	202.8	390	32.7
TAU wk. 40	2232.7	244.0	267	35.0
TAU wk. 41	2203.6	242.6	424	38.7
TAU wk. 42	6228.2	528.8	414	37.2
VEAS wk. 36	7448.0	685.2	440	50.3
VEAS wk. 37	7778.4	530.5	616	64.4
VEAS wk. 38	9402.3	794.6	673	57.5
VEAS wk. 39	8058.1	737.8	412	50.5

Table A6. Concentrations of b*isphenol A and tetrabromobisphenol A* in sludge and effluent samples from the 8 WWTPs

	Sludge (ng/g)		Effluent (ng/L)	
	BPA	TBBPA	BPA	TBBPA
Arendal wk. 30	1 122	34.2	2 960	<LoD
Arendal wk. 37	402	6.30	2 509	<LoD
Arendal wk. 38	750	11.1	2 787	<LoD
Arendal wk. 39	1 141	22.6	2 861	<LoD
Bekkelaget wk. 36	792	<LoD	<LoD	<LoD
Bekkelaget wk. 37	839	6.40	65.8	<LoD
Bekkelaget wk. 40	950	2.90	91.4	<LoD
Bekkelaget wk. 41	949	<LoD	70.8	<LoD
Frevar wk. 36	800	10.9	1 203	<LoD
Frevar wk. 37	908	15.2	2 518	<LoD
Frevar wk. 38	835	10.6	295	<LoD
Frevar wk. 39	206	3.80	577	<LoD
HIAS wk. 37	1 700	<LoD	359	<LoD
HIAS wk. 38	1 160	4.80	230	<LoD
HIAS wk. 42	1 405	<LoD	115	<LoD
HIAS wk. 43	1 310	<LoD	217	<LoD
RA-2 wk. 40	234	<LoD	40.2	<LoD
RA-2 wk. 41	136	<LoD	63.7	<LoD
RA-2 wk. 41	67.5	<LoD	61.4	<LoD
RA-2 wk. 42	140	<LoD	53.8	<LoD
Sandefjord wk. 36	448	24.4	719	<LoD
Sandefjord wk. 37	464	20.0	978	<LoD
Sandefjord wk. 40	387	14.6	394	<LoD
Sandefjord wk. 42	361	13.8	403	<LoD
TAU wk. 39	356	7.40	2 317	<LoD
TAU wk. 40	201	54.3	2 170	<LoD
TAU wk. 41	312	30.4	1 461	<LoD
TAU wk. 42	370	129	3 398	<LoD
VEAS wk. 36	76.1	<LoD	112	<LoD
VEAS wk. 37	143	<LoD	103	<LoD
VEAS wk. 38	99.3	1.90	103	<LoD
VEAS wk. 39	103	2.40	59.0	<LoD

Table A7. Concentrations of h*exabromocyclododecane (HBCD)* in sludge and effluent samples from the 8 WWTPs

	Sludge (ng/g)			Effluent (ng/L)		
	α-HBCD	β-HBCD	γ-HBCD	α-HBCD	β-HBCD	γ-HBCD
Arendal wk. 30	0.76	<LoD	2.75	<LoD	<LoD	<LoD
Arendal wk. 37	0.74	<LoD	0.89	<LoD	<LoD	<LoD
Arendal wk. 38	0.53	0.15	0.99	<LoD	<LoD	<LoD
Arendal wk. 39	3.34	<LoD	5.21	<LoD	<LoD	<LoD
Bekkelaget wk. 36	1.17	5.33	4.18	<LoD	<LoD	<LoD
Bekkelaget wk. 37	1.59	0.29	9.49	<LoD	<LoD	<LoD
Bekkelaget wk. 40	1.13	<LoD	8.50	<LoD	<LoD	<LoD
Bekkelaget wk. 41	1.85	<LoD	11.3	<LoD	<LoD	<LoD
Frevar wk. 36	0.24	<LoD	0.43	<LoD	<LoD	<LoD
Frevar wk. 37	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
Frevar wk. 38	0.79	<LoD	0.47	<LoD	<LoD	<LoD
Frevar wk. 39	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
HIAS wk. 37	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
HIAS wk. 38	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
HIAS wk. 42	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
HIAS wk. 43	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
RA-2 wk. 40	0.77	<LoD	1.82	<LoD	<LoD	<LoD
RA-2 wk. 41	1.32	<LoD	5.45	<LoD	<LoD	<LoD
RA-2 wk. 41	1.07	<LoD	8.27	<LoD	<LoD	<LoD
RA-2 wk. 42	3.22	<LoD	39.5	<LoD	<LoD	<LoD
Sandefjord wk. 36	1.85	<LoD	15.3	<LoD	<LoD	<LoD
Sandefjord wk. 37	1.09	<LoD	7.58	<LoD	<LoD	<LoD
Sandefjord wk. 40	70.6	6.27	56.9	<LoD	<LoD	<LoD
Sandefjord wk. 42	0.66	<LoD	19.6	<LoD	<LoD	<LoD
TAU wk. 39	3.89	3.27	4.38	<LoD	<LoD	<LoD
TAU wk. 40	2.50	0.57	17.5	1.01	<LoD	<LoD
TAU wk. 41	3.67	<LoD	4.63	<LoD	<LoD	<LoD
TAU wk. 42	6.93	1.71	6.57	<LoD	<LoD	<LoD
VEAS wk. 36	1.54	0.29	4.15	<LoD	<LoD	<LoD
VEAS wk. 37	2.53	<LoD	3.20	<LoD	<LoD	<LoD
VEAS wk. 38	1.73	0.28	5.08	<LoD	<LoD	<LoD
VEAS wk. 39	0.83	<LoD	8.09	<LoD	<LoD	<LoD

Table A8. Concentrations of p*olybrominated diphenyl ether (PBDE)* in sludge samples from the 8 WWTPs

Sludge (ng/g)	2,4,4',-TriBDE	2,2',4,4'-TetBDE	2,3',4,4'-TetBDE	2,2',4,5' + 2,3',4',6-TetBDE	3,3',4,4'-TetBDE	2,2',3,4,4'-PenBDE	2,2',4,4',5-PenBDE	2,2',4,4',6-PenBDE	2,3',4,4',6-PenBDE
Arendal wk. 30	0.29	15.1	<LoD	0.66	<LoD	1.41	17.5	2,10	<LoD
Arendal wk. 37	0.25	12.9	<LoD	0.61	<LoD	0.72	13.8	1,87	<LoD
Arendal wk. 38	0.29	16.4	<LoD	1.43	<LoD	<LoD	18.3	2,94	0,42
Arendal wk. 39	0.26	14.7	<LoD	1.10	<LoD	<LoD	17.4	2,65	0,40
Bekkelaget wk. 36	0.36	15.0	<LoD	4.18	<LoD	0.48	12.8	2,77	<LoD
Bekkelaget wk. 37	0.30	14.9	<LoD	4.12	<LoD	0.69	13.4	2,07	<LoD
Bekkelaget wk. 40	0.27	15.8	0.22	4.72	<LoD	<LoD	13.6	2,70	0,35
Bekkelaget wk. 41	0.29	16.0	0.32	4.27	<LoD	0.56	13.9	2,58	<LoD
Frevar wk. 36	0.22	9.03	0.21	0.43	<LoD	0.43	9.52	1,43	<LoD
Frevar wk. 37	0.20	10.4	<LoD	0.54	<LoD	0.64	11.2	1,47	<LoD
Frevar wk. 38	0.18	9.82	<LoD	0.77	<LoD	0.29	11.3	1,69	<LoD
Frevar wk. 39	0.15	8.23	<LoD	0.80	<LoD	0.37	8.66	1,17	0,19
HIAS wk. 37	0.38	18.5	<LoD	0.65	<LoD	0.49	20.6	2,73	<LoD
HIAS wk. 38	0.27	14.9	0.27	1.25	<LoD	0.48	16.5	2,51	<LoD
HIAS wk. 42	0.29	17.1	<LoD	1.00	<LoD	<LoD	18.2	2,52	<LoD
HIAS wk. 43	0.29	16.4	<LoD	1.38	<LoD	<LoD	16.8	2,62	0,35
RA-2 wk. 40	0.07	2.64	<LoD	0.08	<LoD	0.17	2.86	0,41	<LoD
RA-2 wk. 41	0.05	2.30	<LoD	0.09	<LoD	0.16	2.50	0,34	<LoD
RA-2 wk. 41	0.06	2.55	<LoD	0.10	<LoD	0.22	2.94	0,39	<LoD
RA-2 wk. 42	0.07	3.56	<LoD	0.13	<LoD	0.22	3.90	0,52	<LoD
Sandefjord wk. 36	0.24	12.4	<LoD	0.45	<LoD	0.95	13.7	1,53	<LoD
Sandefjord wk. 37	0.26	14.9	<LoD	1.06	<LoD	<LoD	14.7	1,78	<LoD
Sandefjord wk. 40	0.20	13.6	<LoD	0.96	<LoD	<LoD	16.6	2,64	0,26
Sandefjord wk. 42	0.26	14.4	<LoD	1.01	<LoD	<LoD	16.5	2,32	0,34
TAU wk. 39	0.07	3.70	0.08	0.12	<LoD	0.18	3.77	0,59	<LoD
TAU wk. 40	0.05	2.62	<LoD	0.12	<LoD	0.16	3.09	0,41	<LoD
TAU wk. 41	0.09	5.14	<LoD	0.27	<LoD	0.33	6.08	0,83	<LoD
TAU wk. 42	0.14	10.2	0.11	0.33	<LoD	0.54	11.7	1,61	<LoD
VEAS wk. 36	0.17	10.4	<LoD	0.36	<LoD	0.86	12.6	1,38	<LoD
VEAS wk. 37	0.17	9.58	<LoD	0.42	<LoD	0.89	11.9	1,42	<LoD
VEAS wk. 38	0.17	10.3	<LoD	0.65	<LoD	<LoD	12.1	1,85	<LoD
VEAS wk. 39	0.17	9.29	<LoD	0.52	<LoD	<LoD	10.3	1,60	<LoD

Table A9. Concentrations of p*olybrominated diphenyl ether (PBDE)* in sludge samples from the 8 WWTPs

Sludge (ng/g)	2,2',3,4,4',5'-HexBDE	2,2',4,4',5,5'-HexBDE	2,2',4,4',5,6'-HexBDE	2,2',3,4,4',5',6-HepBDE	2,2',3,3',4,4',5,6'-OctBDE	2,2',3,3',4,4',5,5',6-NonBDE	DecaBDE
Arendal wk. 30	0.36	2.07	0.79	1.49	5.12	23.7	1 521
Arendal wk. 37	0.10	1.59	0.86	0.90	3.68	25.5	1 143
Arendal wk. 38	0.38	2.22	1.02	1.39	3.32	15.5	1 333
Arendal wk. 39	0.37	2.13	1.02	1.20	2.75	18.2	1 710
Bekkelaget wk. 36	<LoD	2.19	1.11	2.81	6.57	19.5	328
Bekkelaget wk. 37	0.17	1.81	0.80	1.25	6.57	13.0	352
Bekkelaget wk. 40	0.26	1.76	0.97	1.25	3.43	7.09	287
Bekkelaget wk. 41	0.24	1.74	1.06	1.17	3.73	8.11	277
Frevar wk. 36	0.37	2.05	0.65	1.41	21.9	15.1	204
Frevar wk. 37	0.39	2.23	0.61	1.52	14.1	9.40	181
Frevar wk. 38	0.34	2.12	0.78	1.38	6.40	4.83	155
Frevar wk. 39	0.22	1.35	0.59	0.82	2.43	3.02	157
HIAS wk. 37	0.38	2.36	0.84	1.46	14.5	18.2	518
HIAS wk. 38	0.26	1.82	0.93	1.20	9.68	13.7	300
HIAS wk. 42	0.27	1.97	0.96	1.24	18.0	14.6	276
HIAS wk. 43	0.30	2.03	0.95	1.16	6.00	13.3	430
RA-2 wk. 40	<LoD	0.30	0.19	0.15	<LoD	4.84	81.8
RA-2 wk. 41	0.04	0.27	0.15	0.11	0.13	1.63	43.9
RA-2 wk. 41	0.05	0.35	0.17	0.17	0.20	1.88	55.3
RA-2 wk. 42	0.05	0.43	0.19	0.20	0.21	1.72	61.3
Sandefjord wk. 36	0.11	1.68	0.68	1.52	3.78	13.3	367
Sandefjord wk. 37	0.33	1.78	0.55	2.03	2.05	5.72	380
Sandefjord wk. 40	0.31	2.62	0.89	1.36	1.78	6.03	364
Sandefjord wk. 42	0.23	1.92	0.82	1.29	1.23	5.92	371
TAU wk. 39	<LoD	0.43	0.25	0.19	<LoD	3.51	99.2
TAU wk. 40	<LoD	0.36	0.19	0.18	0.10	2.04	113
TAU wk. 41	0.12	0.66	0.34	0.32	0.40	4.18	203
TAU wk. 42	0.14	1.08	0.72	0.84	0.62	5.11	195
VEAS wk. 36	0.20	1.35	0.68	1.02	6.47	14.3	350
VEAS wk. 37	0.19	1.42	0.48	1.07	8.61	10.1	255
VEAS wk. 38	0.20	1.62	0.60	1.09	4.28	5.40	196
VEAS wk. 39	0.20	1.19	0.44	0.92	3.54	5.24	201

Table A10. Concentrations of p*olybrominated diphenyl ether (PBDE)* in effluent samples from the 8 WWTPs

Effluent (ng/L)	2,4,4'-TriBDE	2,2',4,4'-TetBDE	2,3',4,4'-TetBDE	2,2',4,5' + 2,3',4'6-TetBDE	3,3',4,4'-TetBDE	2,2',3,4,4'-PenBDE	2,2',4,4',5-PenBDE	2,2',4,4',6-PenBDE	2,3',4,4',6-PenBDE
Arendal wk. 30	0,02	0,67	<LoD	0,04	<LoD	<LoD	0,28	0,08	<LoD
Arendal wk. 37	0,02	0,33	<LoD	0,43	<LoD	<LoD	0,14	0,05	<LoD
Arendal wk. 38	0,02	0,45	<LoD	<LoD	<LoD	<LoD	0,16	0,03	<LoD
Arendal wk. 39	<LoD	0,33	<LoD	<LoD	<LoD	<LoD	0,28	0,08	<LoD
Bekkelaget wk. 36	0,03	0,86	<LoD	0,03	<LoD	<LoD	0,2	0,06	<LoD
Bekkelaget wk. 37	<LoD	0,53	<LoD	0,05	<LoD	<LoD	0,11	0,03	<LoD
Bekkelaget wk. 40	<LoD	0,33	<LoD	<LoD	<LoD	<LoD	0,22	0,04	<LoD
Bekkelaget wk. 41	<LoD	0,37	<LoD	<LoD	<LoD	<LoD	0,16	0,03	<LoD
Frevar wk. 36	0,04	0,92	<LoD	0,03	<LoD	0,02	0,65	0,13	<LoD
Frevar wk. 37	0,04	1,17	<LoD	0,07	<LoD	0,02	0,91	0,23	<LoD
Frevar wk. 38	<LoD	0,49	<LoD	0,01	<LoD	<LoD	0,43	0,08	<LoD
Frevar wk. 39	<LoD	0,73	0,04	<LoD	<LoD	<LoD	0,61	0,16	<LoD
HIAS wk. 37	<LoD	0,3	<LoD	<LoD	<LoD	<LoD	0,12	0,02	<LoD
HIAS wk. 38	0,05	1,21	<LoD	0,06	<LoD	<LoD	0,29	0,09	<LoD
HIAS wk. 42	<LoD	0,6	<LoD	<LoD	<LoD	<LoD	0,18	0,08	<LoD
HIAS wk. 43	<LoD	0,43	<LoD	<LoD	<LoD	<LoD	0,19	0,05	<LoD
RA-2 wk. 40	0,04	0,69	<LoD	0,09	<LoD	<LoD	0,18	0,07	<LoD
RA-2 wk. 41	<LoD	0,84	<LoD	<LoD	<LoD	<LoD	0,31	0,08	<LoD
RA-2 wk. 41	0,07	0,39	<LoD	<LoD	<LoD	<LoD	0,13	0,05	<LoD
RA-2 wk. 42	0,08	0,48	<LoD	0,06	<LoD	<LoD	0,17	0,04	<LoD
Sandefjord wk. 36	0,02	1	<LoD	0,04	<LoD	<LoD	0,35	0,1	<LoD
Sandefjord wk. 37	<LoD	0,6	<LoD	<LoD	<LoD	<LoD	0,22	0,06	<LoD
Sandefjord wk. 40	<LoD	0,55	<LoD	<LoD	<LoD	<LoD	0,39	0,08	<LoD
Sandefjord wk. 42	0,06	0,38	<LoD	0,07	<LoD	<LoD	0,12	0,04	<LoD
TAU wk. 39	0,01	0,34	<LoD	<LoD	<LoD	<LoD	0,11	0,04	<LoD
TAU wk. 40	<LoD	0,45	<LoD	<LoD	<LoD	<LoD	0,39	0,1	<LoD
TAU wk. 41	0,05	0,43	<LoD	<LoD	<LoD	<LoD	0,24	0,07	<LoD
TAU wk. 42	<LoD	0,37	<LoD	<LoD	<LoD	<LoD	0,11	0,07	<LoD
VEAS wk. 36	0,03	0,61	<LoD	0,03	<LoD	<LoD	0,2	0,05	<LoD
VEAS wk. 37	0,04	0,71	<LoD	0,09	<LoD	<LoD	0,26	0,07	<LoD
VEAS wk. 38	<LoD	0,29	<LoD	<LoD	<LoD	<LoD	0,16	0,03	<LoD
VEAS wk. 39	0,02	0,45	<LoD	<LoD	<LoD	<LoD	0,2	0,04	<LoD

Table A11. Concentrations of p*olybrominated diphenyl ether (PBDE)* in effluent samples from the 8 WWTPs

Effluent (ng/L)	2,2',3,4,4',5'-HexBDE	2,2',4,4',5,5'-HexBDE	2,2',4,4',5,6'-HexBDE	2,2',3,4,4',5',6-HepBDE	2,2',3,3',4,4',5,6'-OctBDE	2,2',3,3',4,4',5,5',6-NonBDE	DecaBDE
Arendal wk. 30	<LoD	<LoD	0.02	<LoD	<LoD	0.19	6.15
Arendal wk. 37	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	4.86
Arendal wk. 38	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	1.44
Arendal wk. 39	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	8.16
Bekkelaget wk. 36	<LoD	0.03	<LoD	<LoD	<LoD	<LoD	<LoD
Bekkelaget wk. 37	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
Bekkelaget wk. 40	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	3.13
Bekkelaget wk. 41	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	2.09
Frevar wk. 36	0.03	0.10	0.04	0.06	<LoD	0.37	2.62
Frevar wk. 37	<LoD	0.06	0.04	0.04	<LoD	0.13	2.37
Frevar wk. 38	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	1.77
Frevar wk. 39	<LoD	<LoD	0.07	0.19	<LoD	<LoD	5.62
HIAS wk. 37	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
HIAS wk. 38	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
HIAS wk. 42	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	1.17
HIAS wk. 43	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	1.24
RA-2 wk. 40	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	11.2
RA-2 wk. 41	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	5.19
RA-2 wk. 41	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	1.31
RA-2 wk. 42	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
Sandefjord wk. 36	<LoD	0.04	<LoD	<LoD	<LoD	<LoD	1.26
Sandefjord wk. 37	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	1.16
Sandefjord wk. 40	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	3.73
Sandefjord wk. 42	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	0.97
TAU wk. 39	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	1.19
TAU wk. 40	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	5.95
TAU wk. 41	<LoD	0.20	0.10	0.10	<LoD	<LoD	2.28
TAU wk. 42	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
VEAS wk. 36	<LoD	0.02	0.01	0.03	<LoD	<LoD	0.53
VEAS wk. 37	<LoD	0.07	0.06	<LoD	<LoD	<LoD	0.84
VEAS wk. 38	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	0.90
VEAS wk. 39	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	1.39

Table A12. Concentrations of p*erfluorinated compounds (PFCs)* in sludge samples from the 8 WWTPs

Sludge (ng/g)	6:2 FTS	8:2 FTS	PFOS	PFOA	4:2 FTOH	6:2 FTOH	8:2 FTOH	10:2 FTOH	N-Et FOSA	N-Me FOSA	N-Me FOSE	N-Et FOSE
Arendal wk. 30	<LoD	<LoD	3.36	<LoD	<LoD	<LoD	8.22	6.12	<LoD	<LoD	6.58	2.17
Arendal wk. 37	<LoD	<LoD	4.53	<LoD	<LoD	2.96	147	60.2	<LoD	<LoD	20.3	9.75
Arendal wk. 38	<LoD	<LoD	3.81	1.19	3.78	2.38	54.7	23.9	4.69	4.66	19.2	10.4
Arendal wk. 39	<LoD	<LoD	4.53	<LoD	<LoD	<LoD	10.1	7.08	<LoD	<LoD	8.41	3.58
Bekkelaget wk. 36	<LoD	<LoD	8.90	1.28	<LoD	<LoD	5.93	6.11	<LoD	<LoD	2.60	2.02
Bekkelaget wk. 37	<LoD	<LoD	9.60	1.44	<LoD	1.75	7.91	9.91	<LoD	<LoD	5.73	3.25
Bekkelaget wk. 40	<LoD	<LoD	9.46	<LoD	<LoD	1.41	8.26	5.74	<LoD	<LoD	2.39	1.53
Bekkelaget wk. 41	<LoD	<LoD	9.33	1.28	<LoD	1.37	13.0	8.55	<LoD	<LoD	3.28	2.65
Frevar wk. 36	<LoD	<LoD	4.96	<LoD	<LoD	<LoD	2.63	2.69	<LoD	<LoD	<LoD	<LoD
Frevar wk. 37	<LoD	<LoD	5.30	<LoD	7.66	12.1	11.2	11.2	7.69	7.32	11.0	8.09
Frevar wk. 38	<LoD	<LoD	4.01	<LoD	<LoD	<LoD	2.94	2.10	<LoD	<LoD	1.77	0.98
Frevar wk. 39	<LoD	<LoD	3.86	<LoD	<LoD	<LoD	2.25	1.62	<LoD	<LoD	<LoD	<LoD
HIAS wk. 37	<LoD	<LoD	3.60	4.75	<LoD	1.39	5.97	4.02	0.70	0.76	4.59	2.13
HIAS wk. 38	<LoD	<LoD	2.61	2.64	<LoD	<LoD	8.73	10.2	<LoD	<LoD	5.43	2.41
HIAS wk. 42	<LoD	<LoD	3.04	2.35	<LoD	1.09	5.37	3.62	0.62	0.68	4.13	1.96
HIAS wk. 43	0.80	<LoD	2.85	2.54	<LoD	0.73	7.13	7.58	<LoD	<LoD	5.44	<LoD
RA-2 wk. 40	<LoD	<LoD	6.90	<LoD	<LoD	<LoD	16.6	5.62	<LoD	<LoD	1.51	<LoD
RA-2 wk. 41	<LoD	<LoD	5.19	0.50	<LoD	1.84	18.6	6.45	<LoD	<LoD	1.48	1.00
RA-2 wk. 41	1.01	<LoD	4.38	<LoD	<LoD	1.67	9.60	6.67	<LoD	<LoD	2.78	1.85
RA-2 wk. 42	<LoD	<LoD	5.43	<LoD	<LoD	0.53	24.5	<LoD	<LoD	<LoD	2.28	1.43
Sandefjord wk. 36	1.00	<LoD	4.33	<LoD	<LoD	1.84	5.86	3.10	<LoD	<LoD	3.88	2.07
Sandefjord wk. 37	0.82	<LoD	4.59	<LoD	<LoD	1.54	6.59	4.44	0.75	0.84	5.07	2.52
Sandefjord wk. 40	<LoD	<LoD	5.76	<LoD	<LoD	<LoD	14.8	14.4	<LoD	<LoD	2.89	2.28
Sandefjord wk. 42	0.65	<LoD	4.94	<LoD	7.12	12.3	10.6	10.4	7.16	6.81	10.2	7.52
TAU wk. 39	0.51	<LoD	4.03	0.86	<LoD	2.82	26.0	13.7	<LoD	<LoD	6.54	5.74
TAU wk. 40	<LoD	<LoD	3.92	<LoD	<LoD	1.20	2.75	<LoD	<LoD	<LoD	<LoD	<LoD
TAU wk. 41	<LoD	<LoD	6.23	<LoD	3.81	4.40	7.20	4.63	4.15	3.91	6.84	6.88
TAU wk. 42	<LoD	<LoD	4.43	0.55	<LoD	1.58	6.60	8.27	<LoD	<LoD	4.78	2.75
VEAS wk. 36	1.09	<LoD	4.97	0.98	<LoD	1.48	25.7	15.1	<LoD	<LoD	5.31	3.03
VEAS wk. 37	1.10	<LoD	4.34	1.48	<LoD	0.86	16.8	13.2	<LoD	<LoD	4.58	1.95
VEAS wk. 38	0.97	<LoD	5.24	<LoD	<LoD	0.70	6.55	7.79	<LoD	<LoD	5.08	3.84
VEAS wk. 39	1.07	<LoD	4.48	1.10	<LoD	2.17	29.4	17.1	<LoD	<LoD	7.23	6.15

Table A12. Concentrations of *perfluorinated compounds (PFCs)* in effluent samples from the 8 WWTPs

Effluent (ng/L)	6:2 FTS	8:2 FTS	PFOS	PFOA	4:2 FTOH	6:2 FTOH	8:2 FTOH	10:2 FTOH	N-Et FOSA	N-Me FOSA	N-Me FOSE	N-Et FOSE
Arendal wk. 30	3.55	<LoD	10.1	<LoD	<LoD	1.77	1.85	<LoD	<LoD	<LoD	0.94	<LoD
Arendal wk. 37	2.58	<LoD	6.15	8.44	<LoD	3.30	3.81	<LoD	1.31	1.33	1.89	<LoD
Arendal wk. 38	3.51	<LoD	3.99	5.95	<LoD	<LoD	1.80	<LoD	<LoD	<LoD	<LoD	<LoD
Arendal wk. 39	4.18	<LoD	10.7	13.2	-	-	-	-	-	-	-	-
Bekkelaget wk. 36	6.76	<LoD	5.94	<LoD	<LoD	<LoD	1.29	<LoD	<LoD	<LoD	<LoD	<LoD
Bekkelaget wk. 37	9.62	<LoD	5.73	13.1	<LoD	<LoD	1.57	<LoD	<LoD	<LoD	0.63	<LoD
Bekkelaget wk. 40	3.25	<LoD	5.63	13.1	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
Bekkelaget wk. 41	3.77	<LoD	6.12	11.3	12.1	11.8	9.80	0.03	8.97	8.75	11.8	8.69
Frevar wk. 36	806	<LoD	10.9	20.2	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
Frevar wk. 37	31.9	<LoD	16.5	15.0	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
Frevar wk. 38	13.5	<LoD	12.1	7.86	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
Frevar wk. 39	5.41	<LoD	10.9	16.4	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
HIAS wk. 37	8.21	<LoD	2.05	13.5	<LoD	<LoD	20.0	<LoD	<LoD	<LoD	<LoD	<LoD
HIAS wk. 38	8.17	<LoD	2.04	15.2	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
HIAS wk. 42	4.98	<LoD	1.55	7.46	<LoD	<LoD	<LoD	<LoD	<LoD	1.18	1.27	<LoD
HIAS wk. 43	4.73	<LoD	1.55	4.57	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
RA-2 wk. 40	4.64	<LoD	40.2	12.2	-	-	-	-	-	-	-	-
RA-2 wk. 41	4.78	<LoD	42.1	15.1	<LoD	<LoD	1.78	<LoD	<LoD	0.77	1.34	<LoD
RA-2 wk. 41	4.51	<LoD	19.0	10.8	<LoD	<LoD	1.74	<LoD	<LoD	0.63	0.75	<LoD
RA-2 wk. 42	<LoD	<LoD	19.4	9.06	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
Sandefjord wk. 36	19.0	<LoD	6.12	6.96	<LoD	2.20	2.68	<LoD	1.35	1.39	1.91	<LoD
Sandefjord wk. 37	19.3	<LoD	3.53	3.80	<LoD	<LoD	1.63	<LoD	<LoD	<LoD	<LoD	<LoD
Sandefjord wk. 40	7.88	<LoD	5.47	6.31	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
Sandefjord wk. 42	18.7	<LoD	2.57	4.28	<LoD	<LoD	1.68	<LoD	<LoD	<LoD	<LoD	<LoD
TAU wk. 39	5.21	<LoD	7.30	<LoD	<LoD	1.69	2.57	<LoD	<LoD	<LoD	0.98	<LoD
TAU wk. 40	5.76	<LoD	7.48	<LoD	<LoD	<LoD	1.68	<LoD	<LoD	<LoD	<LoD	<LoD
TAU wk. 41	5.41	<LoD	10.3	11.7	<LoD	1.46	2.24	<LoD	<LoD	0.56	0.79	<LoD
TAU wk. 42	14.6	<LoD	4.73	9.93	<LoD	<LoD	1.72	<LoD	<LoD	<LoD	<LoD	<LoD
VEAS wk. 36	10.6	<LoD	6.16	7.55	<LoD	4.60	4.26	<LoD	1.81	1.93	2.68	1.94
VEAS wk. 37	9.33	<LoD	7.03	18.3	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD
VEAS wk. 38	14.7	<LoD	6.11	7.84	-	-	-	-	<LoD	<LoD	<LoD	<LoD
VEAS wk. 39	11.2	<LoD	6.36	<LoD	<LoD	<LoD	<LoD	<LoD	<LoD	0.37	<LoD	<LoD

Where – denotes not analysed

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Om Statlig program for forurensningsovervåking

Statlig program for forurensningsovervåking omfatter overvåking av forurensningsforholdene i luft og nedbør, skog, vassdrag, fjorder og havområder. Overvåkingsprogrammet dekker langsiktige
undersøkelser av:

- overgjødsling
- forsuring (sur nedbør)
- ozon (ved bakken og i stratosfæren)
- klimagasser
- miljøgifter

Overvåkingsprogrammet skal gi informasjon om tilstanden og utviklingen
av forurensningssituasjonen, og påvise eventuell uheldig utvikling på et
tidlig tidspunkt. Programmet skal dekke myndighetenes informasjonsbehov om forurensningsforholdene, registrere virkningen av iverksatte tiltak for å redusere forurensningen, og danne grunnlag for vurdering av nye tiltak.
Klima- og forurensningsdirektoratet er ansvarlig for gjennomføringen av overvåkingsprogrammet.

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